# TREATMENT WETLANDS

## SECOND EDITION



ROBERT H. KADLEC

SCOTT D. WALLACE





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# Preface

Since the first edition of this book, treatment wetland technology has advanced on all fronts. Considerably more is known today about how treatment wetlands function. Over the last decade, wetland technology has evolved into new reactor configurations, a much broader range of treatment applications, and a dramatically expanded presence worldwide.

This growing knowledge base leads to an increased appreciation of just how complex treatment wetlands are. Because treatment wetlands are strongly influenced by a variety of internal and external ecological cycles, the assumptions that simplify the analysis of conventional reactors in the environmental engineering field can no longer be justified. As wetland technology continues to evolve, much effort is being applied to understand both short-term and long-term variability cycles within treatment wetlands. Because treatment variability strongly influences wetland design, factors that influence performance—especially the role of internal biogeochemical cycles—become paramount in understanding how treatment wetlands function. This knowledge can then be applied to make informed decisions regarding wetland design.

## WHAT IS A WETLAND?

The meaning of the word *wetland* has been severely stretched in the treatment wetland literature. We would generally insist that wetlands have plants, water, and some kind of media. Without plants they are soil, sand, or gravel filters, or ponds. In fact, planted gravel filters— meaning all subsurface flow wetlands—have no natural wetland analog. Similarly, it is not unusual to hear discussion of "treatment wetlands" that do not have plants. We have tried to use commonly accepted terminology for systems that are generally regarded as within the scope of the treatment wetland field. We have also distinguished systems based on their hydrology, which may be horizontal flow, pulse-feed vertical, fill-and-drain, or recirculating.

## SCALING FACTORS

Treatment wetlands are in-the-ground, outdoor systems. With other visions guiding them, treatment wetland researchers sometimes find wetlands to be potted plants, pots filled with gravel and no plants, sections of pipe, flasks, test tubes, and all manner of tubs, tanks, and troughs, sometimes with recently inserted propagules. Indoor systems do not experience wind, sun, birds, and animals. When the size is too small, the system is subject to severe edge effects. Although comparative results from small lab systems are useful, there is often the unstated assumption that they would represent the treatment performance of a full-scale wetland. We have tried to be reasonably careful by drawing attention to scale with the terms *microcosm*, *mesocosm*, and *pilot project.*

## SHORT-TERM STUDIES

We find too many studies are based on infant or juvenile ecosystems, which have not had time to mature into the full suite of components that occur in fully developed wetlands. We also find too many studies focus on short-term events. This, we believe, is like interpreting the meal-time hamburger intake rate of teenage boys and girls as their sustainable caloric intake.

For instance, the development of bed clogging in HSSF wetlands has not been studied in a systematic way in the academic community. Recent knowledge of bed clogging has come from the hydraulic failure of full-scale systems (often at a high price) because clogging phenomena takes longer to develop than the tenure of a typical graduate student. As a result, the long-term viability, and maintenance requirements, of HSSF wetlands is still unknown, despite the fact that thousands of systems have been constructed worldwide.

It is fortunate that there are now numerous full-scale projects to balance the data scales.

## WHAT'S NEW?

Of course, there is much more information available now than in 1995 when the previous analyses were completed. The doubling time of the available data is on the order of two or three years, because old systems continue to return new information as more and more systems come on line in more and more application areas. As a consequence, about 90% of the data used in support of this book was not available at the time of the first edition. It has been reassuring to find that most of the data and interpretations of the first edition have stood up well to the test of time, but not surprising to find that some numerical interpretation had to be updated.

Data analysis in the first edition was predicated on the plug flow assumption, despite the known fact that virtually no treatment wetland actually tested out as plug flow. It is now understood that while this may provide acceptable interpolation on existing performance ranges, it can lead to very bad extrapolations that should not be used in design. Further, it has been recognized that weathering of the mixtures that comprise many of the standard wastewater parameters will also invalidate the plug flow assumption. Accordingly, a mixing parameter has been added to the mathematical representation of wetland behavior.

## DESIGN TOOLS

With the advent and proliferation of desktop computing, expectations for calculational detail have risen markedly in the last 15 years. It is no longer necessary to be given a single equation, arranged to be solved for the single variable of interest. This second edition is predicated on the extensive use of spreadsheets, and the large array of iterative and optimization tools that go with them. The scientific design team for a constructed wetland must include that capability, or else be constrained to simple scale-up or scale-down for a repetitive design.

In the first edition, central tendency rate coefficients were presented, along with tables detailing the values for individual systems. Several investigators soon found that their results did not match the central tendencies, and incorrectly concluded that something must be wrong. In this edition, we have therefore opted to present the distributions of rate coefficients across numerous wetlands of all types, so that new results may be placed in that spectrum, and designs may be selected with different positions across the intersystem landscape.

The scatter of wetland outlet concentrations around an often-seasonal trend is another type of variability to be accounted. The first edition utilized maximum monthly deviations across the year. Here, the annual pattern is accounted separately, based on system performances, and various percentiles of the exceedance distribution are presented as a necessary part of design.

Among the differences between the new and the old data interpretations, the narrowing of the gaps between surface and subsurface flow system performance and cost are perhaps the most intriguing. Based on new and greatly expanded data analysis, subsurface flow wetlands do not enjoy much of a performance margin on a per unit area basis, and may be less effective than surface flow systems for some contaminants. However, the cost differential is much less than previously thought, when comparable-sized wetlands are evaluated, but still remains about a three to one capital advantage for surface flow. Therefore, nuisance and health hazard avoidance rules the selection of wetland type.

## TECHNOLOGY SELECTION

In the early years of constructed wetland technology, and to some extent continuing today, there was a tendency to consider wetlands as stand-alone devices, usually accompanied by pretreatment. It is now understood that series and parallel natural system networks, perhaps involving recirculation, are sometimes better choices. Combinations of vertical flow, horizontal subsurface flow, ponds, and free water surface wetlands are increasingly being used.

## THIS BOOK

This book has been updated to reflect the dramatic advances in wetland technology over the last 12 years. The authors of this second edition come from different backgrounds, and work in different aspects of the treatment wetland field. By combining our knowledge and experience, we have endeavored to present a broad range of information regarding the science, hydrology, hydraulics, reactor theory, applied design, implementation, cost, and O&M of treatment wetland systems.

The format of the second edition reflects a dual approach. Part I is organized in a manner that allows the reader to explore the internal mechanisms by which treatment wetlands operate. Existing projects and operating results from real-world treatment wetlands are utilized extensively. Internal mechanisms, their influence on treatment performance, and their effect on system variability are explored in detail in Part I.

Part II is organized to allow the reader to examine how performance data is analyzed and applied to the design process. Like the first edition, this book adopts a performance-based approach to design, in addition to presenting design tools such as loading charts and scaling factors. Continuing with the theme of practical implementation, Part II also summarizes current knowledge that is key to getting wetland projects built, including construction methods, cost information, and operation & maintenance (O&M) requirements.

We have not repeated the natural wetland fundamentals that are contained in the first edition, nor have we reiterated databases or case histories contained therein. All other topics have been nearly totally rewritten, as required by the vastly increased data sources and understanding that have developed in the many years since the first edition.

However, as much as things have changed, some things remain the same. The predictions made in the first edition about rapid evolution of treatment wetlands have certainly proven true. We expect that, if anything, this rate of change will continue to increase after the publication of this second edition, which might have been more properly called *Treatment Wetlands II*.

**Robert H. Kadlec**
**Scott D. Wallace**

# Acknowledgments

The authors want to acknowledge our families and friends who supported us while writing this book. For Bob Kadlec, the extreme patience of his wife Kelli was a paramount virtue, as she put up with over a year's worth of working weekends and the virtual loss of a spouse. Scott Wallace would like to thank his coworkers at North American Wetland Engineering, who stepped forward and handled all the challenges of managing projects and running an engineering company so he could have the freedom to write this book. It is a pleasure to work with such a group of excellent people.

A tremendous amount of effort was given by Jan Vymazal, who helped immensely in the preparation of Part I of this book. His broad understanding of treatment wetlands is evidenced in his many authored and edited volumes, and we are very grateful for his assistance.

Jaime Nivala was instrumental in the completion of this book. She carefully reviewed every chapter, figure, and table; her abilities as both coordinating editor and environmental engineer were invaluable to us. Jaime did an excellent job of managing the myriad details of producing a book of this scope, and her organizational skills made the writing process much easier. This book could not exist in its current form without her extraordinary efforts.

We also want to thank Sue Knapp, who injected a breath of life into the cover design and all of the engineering drawings and hand sketches that are now the final artwork in this book.

This book expands upon many concepts advanced in the first edition, for which Robert Knight bears a full share of credit. He was a major architect of the foundation for this work.

The authors wish to acknowledge the efforts of the hundreds of engineers and scientists who have had the courage to create, innovate, and ultimately develop treatment wetlands as a viable technology to solve many environmental problems. The friendly and open communication between colleagues at international conferences has made this field a pleasure to work in, and the "lessons learned" have greatly contributed to the rapid evolution of treatment wetlands.

We are very appreciative of those projects that have shared data with us. Without the data assembled from these diverse resources, this book could not exist. The list is long, and these hundreds of project owners are owed heartfelt thanks for their generosity.

**Robert H. Kadlec**
**Scott D. Wallace**

# Authors

**Robert H. Kadlec** holds B.S., M.S., and Ph.D. degrees in chemical engineering from the University of Wisconsin and University of Michigan, 1958–1962. That era saw the culmination of the "unit operations" approach to chemical processing, and the transition to the use of principles of transport phenomena to describe transfer and reaction rates in a wide variety of chemical and biochemical processes. Those techniques and analytical tools are also the foundation of today's environmental engineering. Bob began applying engineering analysis to wetland processes in 1970, with the goal of managing wetlands for water quality improvement. The result was the Houghton Lake natural wetland treatment system, which is still operating successfully.

Research on that natural wetland, and on the ensuing 30 years of its operation for engineered treatment, formed the early framework for Dr. Kadlec's development of wetland process characterization. The technology has grown tremendously, and so has Bob's involvement in treatment wetland projects. He has participated in over 250 projects, ranging from simple feasibility studies to comprehensive university research projects. Early university studies focused primarily on wetland hydrology and water chemistry. In the course of many projects, a good deal of knowledge of practical ecology was imparted by his colleagues.

He has worked on treatment wetlands in many states and several other countries, participating in the design of over a hundred treatment wetlands. Major and long-running projects have included Houghton Lake, Michigan; Incline Village, Nevada; Hillsdale, Michigan; Columbia, Missouri; and the Everglades Stormwater Treatment Areas. He is past chairman of the International Water Association (IWA) Specialist Group on the Use of Macrophytes in Water Pollution Control. He has authored or coauthored over 130 publications on treatment wetlands, in addition to dozens of project reports. He was a proposer and developer of the U.S. EPA North American Treatment Wetland Database.

Dr. Kadlec retired from his teaching duties in 1993, and is currently doing business as Wetland Management Services, providing specialty consulting services to a wide range of governmental and private organizations. His contributions to this book are an effort to consolidate over three decades of research and practical experience.

**Scott D. Wallace** began his career as a wastewater treatment plant operator, and also worked as a field technician and analytical chemist. He earned a B.S. in civil engineering (1986) and an M.S. in environmental engineering (1989), both from the University of Iowa. Scott has worked full-time as a consulting engineer since 1988, and has been employed at CH2M HILL, Shive-Hattery Engineers and Architects, and HDR Engineering. Scott began designing treatment wetlands in 1992, beginning with the Indian Creek Nature Center, one of the first cold-climate subsurface flow wetlands in the United States.

In 1997, he cofounded North American Wetland Engineering (NAWE), a consulting firm focused on the development and application of treatment wetlands. Since then, he has designed over 200 treatment systems, the majority of which involve wetlands. NAWE was acquired by Jacques Whitford in 2007, and Scott currently works as a principal in the Water Resources Sector for Jacques Whitford. He consults on a wide variety of projects in the United States and internationally.

Scott has been active in research and development, and holds 5 patents on wastewater treatment systems, including aerated subsurface flow wetlands, a technology briefly discussed in this book. He is a registered professional engineer in 22 states, and has written numerous technical papers on treatment wetlands. Scott was the principal investigator for *Small-Scale Constructed Wetland Treatment Systems; Feasibility, Design Criteria, and O&M Requirements,* a design manual published by the Water Environment Research Foundation (WERF) in 2006. He is an active member of the IWA Specialist Group on the Use of Macrophytes in Water Pollution Control.

# *Part I*

## *Technical Underpinnings*



# 1 Introduction to Treatment Wetlands

Since the first edition of this book, treatment wetland technology has advanced on all fronts. Considerably more is known today about how treatment wetlands function. Over the last decade, wetland technology has evolved into new system configurations, a much broader range of treatment applications, and a dramatically expanded presence worldwide.

This growing knowledge base leads to an increased appreciation of just how complex treatment wetlands are. Because treatment wetlands are strongly influenced by a variety of biological processes and by biogeochemical cycles, the assumptions that simplify the analysis of conventional treatment reactors in the environmental engineering field may no longer apply. As wetland technologies continue to evolve, countless effort is being applied to better understand the short-term and long-term treatment and variability cycles in these systems. Because treatment variability strongly influences wetland design, the factors that influence performance, such as hydraulics and internal biogeochemical cycling, become paramount in understanding how treatment wetlands function. This knowledge can then be applied to make informed decisions regarding wetland design.

The format of this second edition reflects this approach. Part I is organized in a manner that allows the reader to explore the internal mechanisms by which treatment wetlands operate. Operating results from existing treatment wetland projects are extensively analyzed. Internal mechanisms, their influence on treatment performance, and their effect on system variability are explored in detail in this part.

Part II is organized to allow the reader to examine how performance data is analyzed and applied to the design process. Like the first edition, this book adopts a performance-based approach to design. Additionally, loading charts and scaling factors are also presented. Continuing with the theme of practical implementation, Part II also summarizes current knowledge that is key to getting wetland projects built, including construction methods, cost information, and operation and maintenance (O&M) requirements.

This book has been updated to reflect the dramatic advances in wetland technology over the last 13 years. The authors of this second edition come from very different backgrounds, and work in different aspects of the treatment wetland field. By combining our knowledge and experience, we have endeavored to present a broad range of information regarding the science, hydrology, hydraulics, reactor theory, applied design, construction, cost, and O&M of treatment wetland systems.

The focus of this book is almost entirely upon constructed wetlands rather than the use of natural wetlands. Although there are definitely circumstances in which it is logical and legal to utilize existing wetlands, it is far more common, at this point in wetland history, that a treatment wetland will be built on an existing upland site. The principles of operation and performance forecasting are not different.

## 1.1 WETLAND CHARACTERISTICS

Wetlands are land areas that are wet during part or all of the year because of their location in the landscape. Historically, wetlands were called *swamps, marshes, bogs, fens*, or *sloughs*, depending on existing plant and water conditions, and on geographic setting. Wetlands are frequently transitional between uplands (terrestrial systems) and continuously or deeply flooded (aquatic) systems. Wetlands are also found at topographic lows (depressions) or in areas with high slopes and low permeability soils (seepage slopes). In other cases, wetlands may be found at topographic highs or between stream drainages when land is flat and poorly drained (sometimes termed *blanket bogs* or *pocosins* in North America). In all cases, the unifying principle is that wetlands are wet long enough to exclude plant species that cannot grow in saturated soils and to alter soil properties because of the chemical, physical, and biological changes that occur during flooding.

There exists a wealth of published information about general wetland science. The reader may consult any of several texts, prominently including:

*Biogeochemistry of Wetlands: Science and Applications*, K.R. Reddy and R. DeLaune, 2008. CRC Lewis Publishers; Boca Raton, Florida.

*Wetland Ecology Principles and Conservation*, P.A. Keddy, 2000. Cambridge University Press; Cambridge, United Kingdom.

*Wetlands*, W.J. Mitsch and J.G. Gosselink, 2007. Fourth Edition, Wiley Publishers; New York, 600 pp.

*Wetland Plants: Biology and Ecology*, J.K. Cronk and M.S. Fennessey (Eds.), 2001. Lewis Publishers; Boca Raton, Florida, 462 pp.

*Wetland Soils*, J.L. Richardson and M.J. Vepraskas (Eds.), 2001. Lewis Publishers; Boca Raton, Florida, 417 pp.

Additionally, there are many compilations of research results for specific regional situations. These are for diverse locations, including, for example:

- **Czech Republic:**
  *Freshwater Wetlands and Their Sustainable Future: A Case Study of the Trebon Basin Biosphere Reserve, Czech Republic*, J. Kvet, J. Jenik, and L. Soukupova (Eds.), 2002. Parthenon Publishing; New York, 495 pp.
- **Canada:**
  *Prairie Wetland Ecology,* H.R. Murkin, A.G. van der Valk, and W.R. Clark (Eds.), 2000. Iowa State University Press; Ames, Iowa, 413 pp.
- **Florida:**
  *Phosphorus Biogeochemistry in Sub-Tropical Ecosystems*, K.R. Reddy, G.A. O'Connor, and C.L. Schelske (Eds.), 1999. Lewis Publishers; Boca Raton, Florida, 707 pp.

When combined with periodicals focused on wetlands, such as the journal *Wetlands*, these form a formidable collection of scientific works that explore many facets of wetland behavior and character.

Wetlands have properties that make them unique among the major ecosystem groups on Earth. Ample water is important for most forms of biological productivity, and wetland plants are adapted to take advantage of this abundant supply of water while overcoming the periodic shortage of other essential chemical elements, such as oxygen. Because of this, wetlands are among the most biologically productive ecosystems on the planet (Figure 1.1). As such, they are frequently inhabited by jungle-like growths of plants and are home to a multitude of animals including mammals, birds, reptiles, amphibians, and fish that are uncommon in other ecosystems.



**FIGURE 1.1** A wide variety of birds and animals use treatment wetlands. (Photo courtesy R. Knight.)

In addition, because wetlands have a higher rate of biological activity than most ecosystems, they can transform many of the common pollutants that occur in conventional wastewaters into harmless byproducts or essential nutrients that can be used for additional biological productivity. These transformations are accomplished by virtue of the wetland's land area, with its inherent natural environmental energies of sun, wind, soil, plants, and animals. These pollutant transformations can be obtained for the relatively low cost of earthwork, piping, pumping, and a few structures. Wetlands are one of the least expensive treatment systems to operate and maintain. Because of the natural environmental energies at work in a wetland treatment system, minimal fossil fuel energy and chemicals are typically necessary to meet treatment objectives (Table 1.1).

**TABLE 1.1**
**Energy Requirements for Different Types of Wastewater Treatment Plants**

| System | Hydraulic Load ($m^3/d$) | Energy Utilization ($kW \cdot h/m^3$) | Reference |
|---|---|---|---|
| Surface Flow Wetlands | — | <0.1 | Brix (1999) |
| Subsurface Flow Wetlands | — | <0.1 | Brix (1999) |
| Facultative Lagoon + Rapid Infiltration | 3,786 | 0.11 | Campbell and Ogden (1999) |
| Facultative Lagoon + Overland Flow | 3,786 | 0.16 | Crites *et al.* (2006) |
| Aerated Subsurface Flow Wetlands | 5,500 | 0.16 | Wallace *et al.* (2006) |
| Tidal Flow (Fill-and-Drain) Wetlands | 1,000 | 0.18 | Maciolek and Austin (2006) |
| Carrousel Oxidation Ditch | 3,786 | 0.51 | U.S. EPA (1996) |
| Trickling Filter + Nitrogen Removal | 3,786 | 0.61 | Crites *et al.* (2006) |
| Activated Sludge + Nitrification | 3,786 | 0.76 | Campbell and Ogden (1999) |
| Extended Aeration Package Plant | 3,786 | 1.06 | U.S. EPA (1996) |
| Sequencing Batch Reactor | 303 | 1.13 | U.S. EPA (1996) |
| Living Machine | 3,786 | 1.51 | U.S. EPA (1996) |

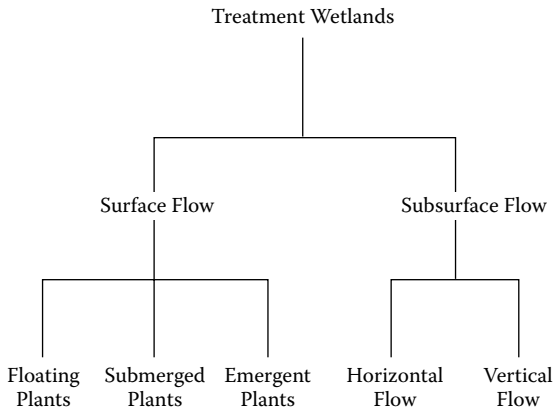



**FIGURE 1.2** Treatment wetland types.

## 1.2 TYPES OF TREATMENT WETLANDS

Modern treatment wetlands are man-made systems that have been designed to emphasize specific characteristics of wetland ecosystems for improved treatment capacity. Treatment wetlands can be constructed in a variety of hydrologic modes. The basic types of constructed wetland systems are shown in Figure 1.2. At the current stage of technology development, three types of wetlands are in widespread use:

- Free water surface (FWS) wetlands have areas of open water and are similar in appearance to natural marshes.
- Horizontal subsurface flow (HSSF) wetlands, which typically employ a gravel bed planted with wetland vegetation. The water, kept below the surface of the bed, flows horizontally from the inlet to the outlet.
- Vertical flow (VF) wetlands distribute water across the surface of a sand or gravel bed planted with wetland vegetation. The water is treated as it percolates through the plant root zone. Biosolids dewatering wetlands can be thought of as a type of VF wetland system.

Each of these major categories employs variants of the layout, media, plants, and flow patterns. For example, FWS wetlands can be operated in intermittent flow, with a fill and drain mode, such as noted by Poach and Hunt (2007), even for continuous municipal discharges. The same operational strategy has also been advocated for HSSF systems (Behrends et al., 2001), and for VF systems (Sun *et al.*, 1999). VF wetlands may be operated in continuous downflow, as is the case for anaerobic mine water wetlands, or they may be operated with intermittent dosing, or they may be operated with intermittent or continuous sprinkling. Event-driven systems, such as stormwater wetlands, experience inflows that are erratic as well as intermittent.

### FWS Wetlands

These wetlands contain areas of open water, floating vegetation, and emergent plants, either by design or as an unavoidable consequence of the design configuration. Depending upon local regulations and soil conditions, berms, dikes, and liners can be used to control flow and infiltration. As the wastewater flows through the wetland, it is treated by the processes of sedimentation, filtration, oxidation, reduction, adsorption, and precipitation. The components in a typical FWS wetland are shown in Figure 1.3.

Because FWS constructed wetlands closely mimic natural wetlands, it should be no surprise that they attract a wide variety of wildlife, namely insects, mollusks, fish, amphibians, reptiles, birds, and mammals (NADB database, 1993; Kadlec and Knight, 1996). Because of the potential for human exposure to pathogens, FWS wetlands are rarely used for secondary treatment (U.S. EPA, 2000c). The most common application for FWS wetlands is for advanced treatment of effluent from secondary or tertiary treatment processes (e.g., lagoons, trickling filters, activated sludge systems, etc.). A typical application of a treatment train with a FWS wetland is shown in Figure 1.4.

FWS wetlands are suitable in all climates, including the far north. However, ice formation can hydraulically preclude winter operation, and the rates of some removal processes are lower for cold water temperatures, notably nitrogen conversion processes. When ice covers the open water, the transfer of oxygen from the atmosphere is reduced, decreasing oxygen-dependent treatment processes. Other processes, such as TSS removal, are more effective under the ice than in summer

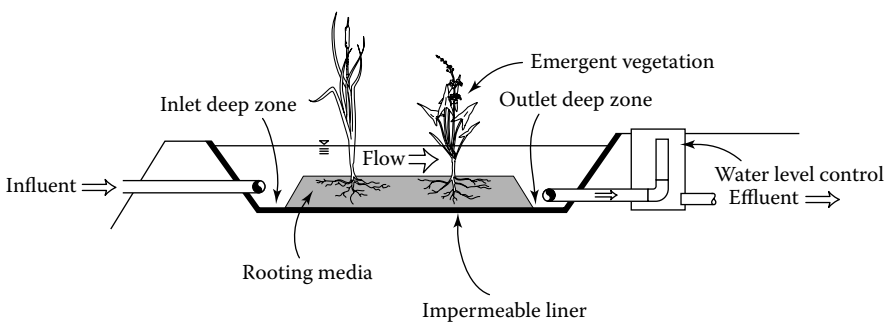



**FIGURE 1.3** Basic elements of a FWS wetland.

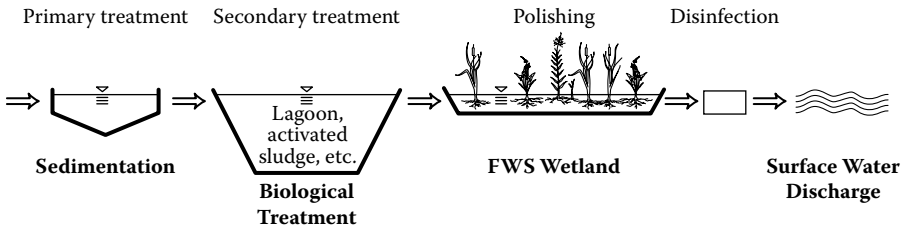



**FIGURE 1.4** Typical application of a FWS wetland for municipal wastewater treatment. (From Wallace and Knight (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements*. Final Report, Project 01-CTS-5, Water Environment Research Foundation (WERF): Alexandria, Virginia. Reprinted with permission.)

conditions. It is generally more efficient to store water during winter and treat it during the warm part of the year.

FWS wetlands are the nearly exclusive choice for the treatment of urban, agricultural, and industrial stormwaters, because of their ability to deal with pulse flows and changing water levels. They are a frequent choice for treatment of mine waters, and for groundwater remediation and leachate treatment.

FWS systems can provide significant ancillary benefits, primarily in the form of human uses and wildlife habitat. Treatment marshes are not inexpensive, but are usually capital cost-competitive with alternative technologies. Operating costs are typically quite low compared to alternatives.

## HSSF Wetlands

HSSF wetlands consist of gravel or soil beds planted with wetland vegetation. They are typically designed to treat primary effluent prior to either soil dispersal or surface water discharge. The wastewater is intended to stay beneath the surface of the media and flows in and around the roots and rhizomes of the plants. Because the water is not exposed during the treatment process, the risk associated with human or wildlife exposure to pathogenic organisms is minimized. Properly operated HSSF wetlands do not provide suitable habitat for mosquitoes.

HSSF wetland systems are generally more expensive than FWS wetlands, although maintenance costs remain low compared to alternatives. They are commonly used for secondary treatment for single-family homes or small cluster systems (Wallace and Knight, 2006), or for small communities (Cooper *et al.*, 1996). However, there are many other applications to specialty wastewaters from industry. In general, HSSF wetlands have been utilized for smaller flow rates than FWS wetlands, probably because of cost and space considerations. HSSF wetlands are typically comprised of inlet piping, a clay or synthetic liner, filter media, emergent vegetation, berms, and outlet piping with water level control. A schematic of a conventional HSSF wetland for warm climates is depicted in Figure 1.5. A typical application of a HSSF wetland is shown in Figure 1.6.

These systems are capable of operation under colder conditions than FWS systems, because of the ability to insulate the top. A key operational consideration is the propensity for clogging of the media. HSSF wetlands do not provide the same opportunities for ancillary benefits that FWS systems do.

## VF Wetlands

Several variations of VF wetlands exist. The most common type, used most often in Europe, employs surface flooding (pulse loading) of the bed in a single-pass configuration (Figure 1.7) (ÖNORM B 2505, 1997). Such systems are roughly analogous to the dosing scheme used in intermittent sand filters. VF wetlands in North America have been

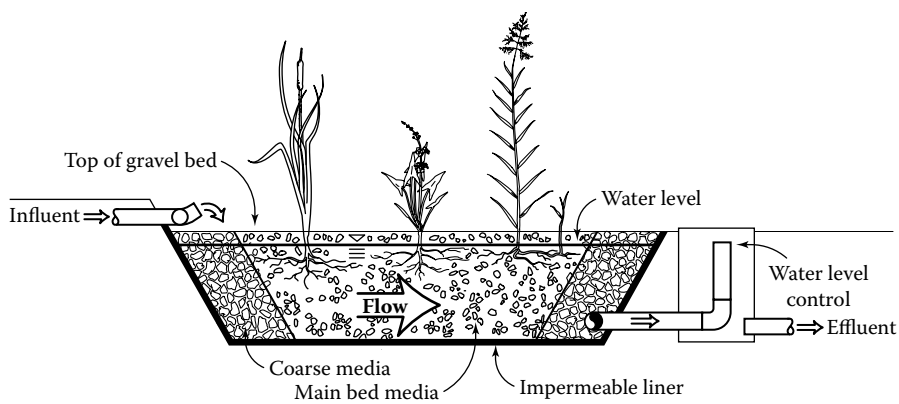



**FIGURE 1.5** HSSF wetland schematic. (From Wallace and Knight (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements*. Final Report, Project 01-CTS-5, Water Environment Research Foundation (WERF): Alexandria, Virginia. Reprinted with permission.)

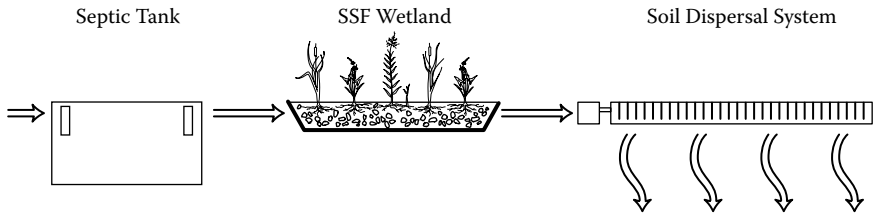



**FIGURE 1.6** Application of a HSSF wetland to domestic wastewater treatment. (From Wallace and Knight (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements*. Final Report, Project 01-CTS-5, Water Environment Research Foundation (WERF): Alexandria, Virginia. Reprinted with permission.)

designed as vegetated recirculating gravel filters (Lemon *et al*., 1996). Upflow systems have been suggested to minimize oxygen transfer and promote reductive dehalogenation (Kassenga *et al*., 2004), and fill-and-drain (tidal flow) systems have been implemented, mainly in North America, to treat high-strength wastes and to oxidize ammonia (Behrends *et al*., 1996b; Austin and Lohan, 2005).

HSSF wetlands have a limited capacity to oxidize ammonia, because of limited oxygen transfer. VF wetlands were developed in Europe to provide higher levels of oxygen transfer, thus producing a nitrified effluent. The technology, initiated by Dr. Kathe Seidel in the early 1960s, became part of the Max Planck Institute Process (MPIP) (Brix, 1994d). These systems may be combined with HSSF or FWS wetlands to create nitrification-denitrification treatment trains (Figure 1.8; Cooper *et al*., 1999).

The ability of VF wetlands to oxidize ammonia has resulted in their use in applications with higher ammonia than municipal or domestic wastewater. Landfill leachates and food processing wastewaters can have ammonia levels in the hundreds of milligrams per liter, and the key to reduction is the ability to nitrify. Successful VF wetlands therefore have formed part of the treatment process for those wastes (Burgoon *et al*., 1999; Kadlec, 2003c).

Another variation of VF wetlands relies upon exactly the opposite process: the use of overlying water to block oxygen transport, in order to create anaerobic conditions in the bottom bed sediments. A surface water pool on top of organics and limestone creates downflow into a zone with reducing conditions that fosters appropriate sulfur chemistry to immobilize metals (Younger *et al*., 2002).

Very concentrated wastewaters can be treated in VF systems. Unsettled raw sewage is added to VF wetlands in a French version of the technology (Molle *et al*., 2005a), and sludge from activated sludge plants may be dewatered in VF systems (Nielsen, 2004).

Biosolids dewatering wetlands consist of an enclosed basin with alternating filter layers which trap organic biosolids on the surface of the wetland bed. Biosolids are applied to the surface of the wetland bed, and water percolates vertically down through the wetland bed primarily through mechanisms of unsaturated flow. Sludge dewatering systems target water removal and consolidation, rather than the elimination of dissolved constituents. Sludge dewatering beds consist of an enclosed basin with a sand layer underlain by drainage pipes. The sand bed is planted with emergent wetland plants (typically *Phragmites*), and fed throughout the year in intervals with up to 20 cm of stabilized sewage sludge per loading (Barjenbruch *et al*., 2002). Solids content is typically 35–40% after dewatering (DeMaeseneer, 1997). Higher solids contents may be achieved, but this usually requires sacrificing the plants to drought stress (Nielsen, 1990). Freezing

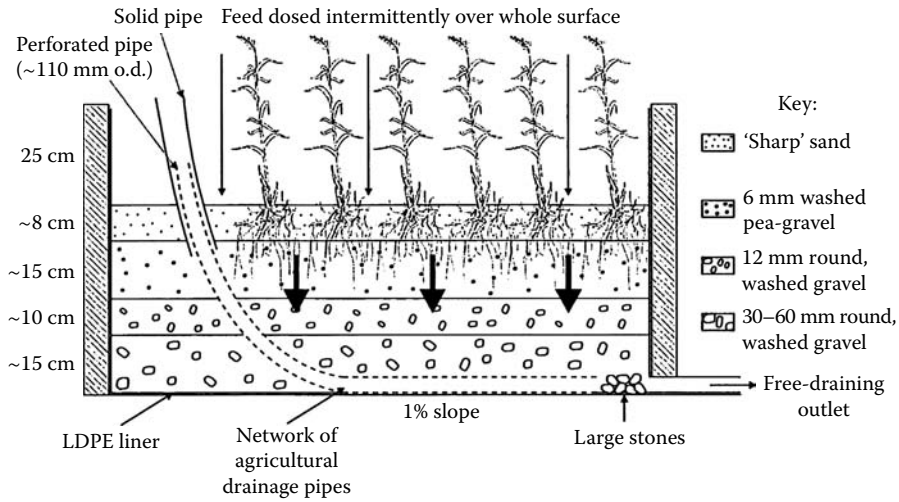



**FIGURE 1.7** Typical arrangement of a VF constructed wetland. (From P.F. Cooper *et al*. (1996) *Reed Beds and Constructed Wetlands for Wastewater Treatment*. WRc Publications. Swindon, United Kingdom. Reprinted with permission.)

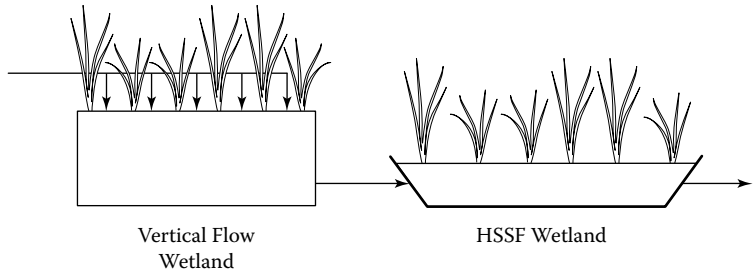



**FIGURE 1.8** A hybrid wetland system (VF + HSSF).

conditions enhance performance since ice crystals lyse the cell walls of the bacteria in the biosolids (Reed *et al*., 1995).

## 1.3 WETLANDS AS A TREATMENT TECHNOLOGY

Wastewater from human dwellings and activities has been a primary target of many treatment technologies, including treatment wetlands. Most of the early applications were for domestic and municipal wastewater, and that sector of the technology continues to grow at a rapid pace in many places. There are exceptions, such as Denmark, where some types of treatment wetlands have been implemented in essentially all locations where they make sense. However, there are a growing number of applications dealing with animal and industrial waters, urban and agricultural stormwaters, mine waters, groundwater remediation, and other applications. This diversification adds a second dimension to the growth of the use of treatment wetlands. It is useful to position the role of wetlands in the milieu of wastewater treatment in general.

### Municipal Wastewater Treatment

Conventional municipal wastewater treatment is accomplished by physical, chemical, and biological processes. Many of these processes are general in nature and can function within a variety of treatment schemes. A review of these technologies is valuable for planning and designing a wetland treatment project for at least two reasons. First, pretreatment with conventional processes is usually advisable before discharge into a wetland because of the potential solids or oxygen demand overload that might create nuisance conditions within a wetland receiving raw or inadequately treated wastewaters. The wetland designer should be aware of the types of conventional processes that can be used to accomplish this pretreatment.

Secondly, a wetland treatment technology may not be the most cost-effective, environmentally sensitive, or technically reliable process for a given wastewater or project location. Conventional treatment technologies should be compared with wetlands and other land treatment technologies before final project planning and design is begun. A knowledge of conventional wastewater treatment methods, as well as the other natural land technologies, is essential to make a sound evaluation of the most appropriate treatment technology or combination of technologies for a given application. For detailed references on conventional municipal and domestic wastewater treatment technologies, see Metcalf and Eddy Inc. (1998) and Crites and Tchobanoglous (1998).

### Primary Treatment

Primary treatment is considered "the first line of defense" in wastewater treatment (Water Environment Federation, 1988) because it sets the stage for the majority of biological treatment technologies that follow. Primary treatment consists of screening, grit removal, and primary sedimentation. Screening and grit removal may be referred to as *preliminary treatment* because they remove larger solids from the wastewater and the heavier mineral solids that might otherwise erode mechanical equipment downstream in the treatment facility.

Grit in raw wastewater primarily consists of inorganic and organic solids that enter the collection system and include materials such as sand, gravel, seeds, coffee grounds, and other minimally decomposable organic solids. Because grit is more settleable than more highly decomposable organic solids, it is removed in the front end of the treatment plant to protect mechanical equipment from abrasion and prevent sedimentation in pipelines and basins. Primary sedimentation is used to initially reduce the high concentration of total suspended solids (TSS) present in raw wastewater. Sedimentation is accomplished by creating quiescent flow conditions within a fairly deep (typically 3 to 5 m) pond or concrete vessel known as a primary clarifier. Settled solids are removed as sludge for further treatment, dewatering, and disposal, while the supernatant is removed via weirs to undergo additional treatment or discharge.

### Secondary Treatment

Secondary treatment is the minimal level of municipal and industrial treatment that is required in the United States before discharge to most surface receiving waters. Secondary treatment requires a treatment level that will produce concentrations of five-day biochemical oxygen demand ($BOD_5$) and TSS less than 30 milligrams per liter (mg/L) and in addition, a minimum percent reduction of 85%.

Secondary treatment generally consists of the removal of additional wastewater solids and dissolved organic matter through microbial uptake and growth. Thus, secondary treatment is essentially a biological process in which bacteria and fungi are encouraged to grow in lagoons, mixed tanks, and ponds, or on fixed surfaces. The principal secondary treatment technologies are facultative ponds, aerated lagoons, aeration basins with solids recycling (activated sludge), trickling filters, and rotating biological contactors.

The activated sludge process is highly efficient at removing residual biochemical oxygen demand and suspended solids remaining in primary wastewater and can be adapted to also reduce ammonia nitrogen, total nitrogen, and phosphorus. A second group of secondary treatment technologies relies on attached growth of microbial populations to extract soluble carbon and nutrients from primary effluents. The trickling filter and rotating biological contactor technologies are two variations of this attached growth treatment process. Rotating biological contactors use circular plastic disks (media) mounted on a horizontal shaft and turning at about one to two rotations per minute (rpm) through a shallow wastewater-filled tank. Multi-unit rotating biological contactors are frequently designed in series, parallel, or both. Covers are typically provided over rotating biological contactors to minimize variation in the physical environment of the treatment process.

## Advanced Treatment

Reductions of biochemical oxygen demand, total suspended solids, nitrogen, and phosphorus beyond those typically accomplished by secondary treatment are called tertiary or *advanced treatment*. Three forms of advanced wastewater treatment include nitrification, denitrification, and phosphorus removal. Nitrification can be accomplished in either suspended growth or in attached growth systems. Nitrification is an aerobic process in which bacteria oxidize ammonia to nitrate nitrogen. Standard activated sludge treatment can be modified to accomplish nitrification by increasing solids recycling (sludge age) and increasing the overall hydraulic residence time of the treatment system. Extended aeration activated sludge systems and oxidation ditch designs typically are capable of nitrification. Trickling filters and rotating biological contactor systems can also be designed for nitrification, especially through the use of multi-stage systems.

The total mass of nitrogen in the wastewater is not reduced by nitrification alone, but can be reduced by a second microbial transformation process called denitrification. In denitrification, nitrate nitrogen is microbially transformed into nitrogen gas, which is lost to the atmosphere. This process is anoxic, and occurs to a limited extent in conventional aerated treatment processes such as activated sludge or trickling filter units. Wastewater treatment systems can be designed for denitrification by including an anaerobic process after effluent nitrification. This process has been added in separate units as well as within single vessel units.

Normal microbial growth during secondary treatment results in a sludge with about 1.5–2% total phosphorus on a dry weight basis. Through sludge wasting, the total phosphorus content of the wastewater receiving secondary activated sludge treatment is reduced by about 10–25%. Biological phosphorus removal relies on a "luxury uptake" of phosphorus that occurs in microbial populations during growth in more vigorously aerated conditions. With higher uptake rates and increased sludge wastage, a higher percentage of the dissolved phosphorus can be removed from the wastewater. In addition, by sequencing through an anaerobic reactor before entry into the aerobic reactor, phosphorus content of the microbes is initially reduced, allowing a greater removal efficiency from solution in the second reactor.

Phosphorus removal from wastewaters is also frequently accomplished through several conventional chemical and physical processes. Chemical processes typically use aluminum (alum) or iron (ferric) salts to chemically precipitate dissolved phosphorus and remove it in a solid (sludge) form. Total phosphorus removal efficiency through chemical precipitation can exceed 90% in municipal effluents resulting in final total phosphorus concentrations lower than 0.5 mg/L. Physical phosphorus removal processes include ultrafiltration, reverse osmosis, and ion exchange. The first two physical processes rely on filtration of colloidal and dissolved phosphorus with a membrane, whereas ion exchange relies on the electrical attraction between ionized forms of phosphorus and specific ion exchange resins. All three of these physical processes require extensive pretreatment for suspended solids reduction and all generate a reject waste stream that may require additional chemical treatment for ultimate phosphorus removal.

## Where Do Wetlands Fit?

Constructed wetlands may be used to provide some or all of the functions of secondary treatment and higher. Effluents that have undergone primary treatment may be further treated in constructed wetlands. At the present time, most such municipal wetlands are restricted to small communities and simple pretreatment systems. Common applications are:

- *Secondary treatment for small communities.* For example, Green and Upton (1992) analyzed the costs for HSSF systems in the United Kingdom, and concluded that they were the technology of choice for villages of up to 2,000 population.
- *Add-ons to aging or overloaded conventional secondary plants.* The wetland acts as a buffer to complete the treatment when there are upsets or extreme flow events that create bypass and concentration excursions in the conventional plant outflow.
- *Add-ons to lagoons.* The solids trapping properties of wetlands can compensate for the export of algal debris from facultative ponds, and provide further nutrient removal.

- *Tertiary and higher treatment of compliant secondary discharges*. Changing regulatory requirements can create the need for advanced treatment, which may be provided by constructed wetlands.

### Domestic Wastewater Treatment

Although septic tanks and their accompanying drainfields have often served admirably in the partial treatment and disposal of wastewaters from single households or small dwelling clusters, that technology is limited in its capabilities and by site conditions. These systems do not control nitrogen, and in fact typically send oxidized nitrogen to groundwater. Clay soils, rock, and high groundwater tables may preclude effective infiltration. In such cases, the addition of a subsurface treatment wetland after the septic tank and preceding the drainfield can compensate for substandard infiltration conditions, and provide a greater level of nitrogen control. Treatment can be designed so that the wetland can discharge to surface waters, which is a frequent choice in Europe and developing countries.

### Animal Wastewater Treatment

Constructed treatment wetlands are compatible with typical farm and ranch operations. Types of livestock wastewater being treated by constructed wetlands include dairy manure and milkhouse wash water, runoff from concentrated cattle-feeding operations, poultry manure, swine manure, and catfish pond water. The Livestock Wastewater Treatment Database (LWDB), created in 1998, included 68 sites with a total of 135 separate systems in North America (Knight *et al.*, 2000). The large majority of these were FWS wetlands. The strength of wastewater is higher than for municipal applications, with BOD, TSS, and ammonia often above 100 mg/L. Consequently, in contrast to other FWS applications, the treatment level may be characterized as primary.

### Minewater Treatment

In the 1980s, a large number of treatment wetlands were built to treat acid mine drainage in the United States (Wieder, 1989). Constructed wetlands were in use at more than 300 sites in the United States in 1989, to increase the pH and reduce concentrations of iron and/or manganese at coal mine sites (Kleinman and Hedin, 1989). Conventional treatment of leachates at these sites would include surface grading and recontouring to reduce or divert flows, and chemical buffering and precipitation to improve water quality. Because these processes have relatively high capital and lifecycle costs, there was considerable interest in developing more cost-effective alternatives, and constructed wetlands were a logical choice. Methods of design were rudimentary, and remain so. Recent interest has grown to include metal mine wastewaters and tailings pile leachates (Younger, 2000). Applications include copper, gold, lead, and zinc mines.

### Industrial Wastewater Treatment

A group of industries, characterized by their involvement in food processing, produce wastewaters that are high in biodegradable organic and nitrogen content. These wastewaters are typically quite strong, and routinely undergo some form of preliminary treatment. However, the reduction of nutrients and organics to regulatory levels is increasingly being accomplished by constructed wetlands. Application areas now involve wetlands serving the potato, wine, olive oil, sugar, starch, alcohol, and meat processing industries.

To meet reduced effluent limitations, some pulp and paper mills are being required to provide treatment beyond the secondary level. One goal may be to further reduce $BOD_5$, TSS, nitrogen, phosphorus, color, chlorinated organics (such as adsorbable organic halides or dioxin), and whole effluent toxicity. Constructed and natural wetland treatment systems have been used at pulp and paper mills to provide advanced secondary or tertiary treatment (NCASI, 2004).

Constructed wetlands are also providing advanced secondary and tertiary treatment of process water and stormwater at a growing number of petroleum refineries (Knight *et al.*, 1997). Typical wastewater pollutants at petroleum refineries include $BOD_5$, COD, oil and grease, TSS, $NH_4$-N, phenolics, $H_2S$, trace organics, and heavy metals. Concentrations of many of these pollutants are reduced through source control and preliminary treatments such as sour water stripping, oxidation and neutralization of spent caustics, and cooling tower blowdown treatment. Constructed wetlands are in use to reduce remaining concentrations of these contaminants to advanced treatment levels.

### Leachate and Remediation

Treatment and disposal of liquid leachates is one of the most difficult problems associated with the use of sanitary landfills for disposal of solid waste. Leachates are produced when rainfall and percolated groundwater combine with inorganic and organic degraded waste. The highly variable nature of solid waste, differences in age and decomposition, and the diversity of chemical and biological reactions that take place in landfills result in a wide range of chemical quality of leachates. In unlined landfills, leachates frequently discharge to groundwater or appear as surficial drainage around the base of the landfill. In modern lined landfills, leachates are collected from the lined cells and routed to treatment units, or are trucked off-site to existing treatment plants. The use of constructed wetlands to treat these landfill leachates is a rapidly developing technology, with both subsurface flow (SSF) wetlands and surface flow wetlands (Mulamoottil *et al.*, 1998).

Groundwaters have been contaminated at a very large number of old industrial sites. For instance, the use of chlorinated ethenes was extremely prevalent up to about 30 years ago, at which time their attendant health hazards were recognized, and use discontinued. However, the dense and partially

water-soluble materials had already been dumped to groundwater for many decades, leaving a legacy of contaminated groundwater. At some few sites, it was observed that removal was being achieved by natural wetlands, via the mechanisms of biodegradation and volatilization. It was only a small step to construct wetlands for the same purpose, and multiple sites now use wetland technology. Other common groundwater contaminants include hydrocarbons such as benzene, toluene, and other fuel hydrocarbons; explosives; and nitrates.

Alternatives to wetlands are extremely costly by comparison. Usually, the target chemicals can be more completely removed, but only with the expense of multistep processes involving chemical engineering technology. Those competing processes have a strong tendency to create large operations and maintenance requirements, and the presence of skilled operators. Such alternatives are especially onerous for remediation that is anticipated to last for many decades. The passive wetland alternative, with low or nonexistent replacement costs, presents a much better lifecycle prospect.

### Urban Stormwater Treatment

Stormwater concentrations and loads are cyclic due to periods of dryfall and deposition, followed by the first flush of runoff after rain, followed by exponential decreases in runoff constituent concentrations as storages rinse from the landscape, and finally dry conditions and deposition until the next storm event. Pollutant concentrations and loads generally range from low levels from undeveloped and park lands, to low density residential and commercial, to higher density residential and commercial, and finally to high density commercial and industrial land uses. The use of constructed wetlands, usually with accompanying ponds, is now a routine best management practice (BMP) for controlling the quality of runoff. In the United States, the implementation of wetland stormwater BMPs has been very uneven, with numerous and early applications on both the east and west coasts, but much later and fewer systems elsewhere.

In contrast to other applications, there is basically no pretreatment for urban stormwaters, if the forebay settling basin is considered part of the wetland. At most, there may be a debris screen to catch major floating objects. Expectations are also typically lower than for many other application areas, with moderately good TSS reductions, but much lesser reductions in dissolve constituents. Although most urban stormwater wetlands are small, and do not acquire data, it is clear that their numbers are quite large.

### Field Runoff Treatment

Target contaminants from agricultural fields vary, depending upon the perceived threat to receiving ecosystems. The principal contaminants include suspended solids, nitrate, phosphorus, and agricultural chemicals, but normally not all at the same time. Runoff from row crops and pasture areas may be low or high in mineral solids, depending on farming practices, rainfall intensity, soil types, and topography. Nutrient concentrations and loads from row crops and pastures depend on fertilization practices. As for urban runoff, there is usually no pretreatment prior to the wetland.

There is a potential role for wetlands in reducing solids loading coming from especially row crops. Such "dirt traps" are remarkably effective even at very short detention times (small systems) (U.S. Department of Agriculture, 1991; Braskerud, 2001a). However, short detention does not suffice to reduce dissolved nutrients, because those removals rely on the biogeochemical cycle, which operates at a much slower pace. Constructed wetlands also have the ability to abate the pulses of some pesticides, but not all, that are exported from fields in modern agriculture (Rodgers and Dunn, 1992). Despite such successes, the implementation of end-of-field wetlands has proceeded at a slow pace.

Wetlands are the only economically feasible means of controlling phosphorus in runoff reaching the Florida Everglades. The biggest constructed wetlands in the world are in operation there, with an aggregate area in excess of 20,000 ha. However, nitrogen pollution is a concern for the marine environments, and therefore wetlands are receiving considerable attention in connection with protection and restoration of the Baltic Sea and the Gulf of Mexico (Arheimer and Wittgren, 1994; Hey, 2002).

## 1.4 HISTORICAL PERSPECTIVE

Natural wetlands have been used as convenient wastewater discharge sites for as long as sewage has been collected (at least 100 years in some locations). Examples of old wetland sites in North America include the Great Meadows natural wetland near the Concord River in Lexington, Massachusetts, which began receiving wastewater in 1912; the Brillion Marsh in Wisconsin that has received municipal wastewater discharges since 1923; the Dundas sewage treatment plant, which began discharging to the Cootes Paradise natural wetland near Hamilton, Ontario, in 1919; and a discharge to a natural cypress swamp from the city of Waldo, Florida, since 1939.

Wetlands constructed for the purpose of treating water have a much shorter history. The worldwide spread of this technology originated from research conducted at the Max Planck Institute in West Germany, starting in 1952 (Bastian and Hammer, 1993) and in the western hemisphere during the 1970s. Implementation of wetland technology has been accelerating around the world since 1985, primarily because treatment wetlands, while mechanically simple, are biologically complex systems capable of achieving high levels of treatment. Furthermore, treatment wetlands can be constructed using local materials and local labor, which is a major advantage in developing countries.

Table 1.2 presents an annotated chronology of some of the major conferences leading to the acceptance of the use of natural and constructed wetlands for water quality management. The table lists selected research efforts, full-scale project

**TABLE 1.2**
**Treatment Wetland Technology Conferences**

| Year | Location | Title (Proceedings) |
|---|---|---|
| 1976 | Ann Arbor, Michigan | Freshwater Wetland and Sewage Effluent Disposal (Tilton *et al.*, 1976) |
| 1978 | Tallahassee, Florida | Environmental Quality Through Wetlands Utilization (Drew, 1978) |
| 1978 | Lake Buena Vista, Florida | Wetland Functions and Values (Greeson *et al.*, 1979) |
| 1979 | Higgins Lake, Michigan | Freshwater Wetland and Sanitary Wastewater Disposal (Sutherland and Kadlec, 1979) |
| 1979 | Davis, California | Aquaculture Systems for Wastewater Treatment (Bastian and Reed, 1979) |
| 1981 | St. Paul, Minnesota | Wetland Values and Management (Richardson, 1981) |
| 1982 | Amherst, Massachusetts | Ecological Considerations in Wetlands Treatment of Municipal Wastewaters (Godfrey *et al.*, 1985) |
| 1986 | Orlando, Florida | Aquatic Plants for Water Treatment and Resource Recovery (Reddy and Smith, 1987) |
| 1988 | Chattanooga, Tennessee | 1st International Conference on Constructed Wetlands for Wastewater Treatment (Hammer, 1989) |
| 1989 | Tampa, Florida | Wetlands: Concerns and Successes (Fisk, 1989) |
| 1990 | Cambridge, United Kingdom | 2nd International Conference on Constructed Wetlands for Water Pollution Control (Cooper and Findlater, 1990) |
| 1991 | Pensacola, Florida | Constructed Wetlands for Water Quality Improvement (Moshiri, 1993) |
| 1992 | Columbus, Ohio | INTECOL Wetlands Conference (Mitsch, 1994) |
| 1992 | Sydney, Australia | 3rd International Conference on Wetland Systems for Water Pollution Control (Pilgrim, 1992) |
| 1994 | Guangzhou, China | 4th International Conference on Wetland Systems for Water Pollution Control (Chuncai, 1994) |
| 1994 | Atlanta, Georgia | On-Site Wastewater Treatment; 7th Symposium on Individual and Small Community Sewage Systems (Collins, 1994) |
| 1995 | Třeboň, Czech Republic | Nutrient Cycling and Retention in Wetlands and Their Use for Wastewater Treatment (Vymazal, 1996) |
| 1996 | Vienna, Austria | 5th International Conference on Wetland Systems for Water Pollution Control (IGWA, 1996) |
| 1996 | Niagara-on-the-Lake, Ontario | Constructed Wetlands in Cold Climates (Friends of Fort George, 1996) |
| 1997 | Romulus, Michigan | Constructed Wetlands for the Treatment of Landfill Leachates (Mulamoottil *et al.*, 1998) |
| 1997 | Třeboň, Czech Republic | Nutrient Cycling and Retention in Natural and Constructed Wetlands (Vymazal, 1999) |
| 1998 | Aguas de São Pedro, Brazil | 6th International Conference on Wetland Systems for Water Pollution Control (Tauk-Tornisielo and Salati, 1998) |
| 1998 | Orlando, Florida | On-Site Wastewater Treatment; 8th Symposium on Individual and Small Community Sewage Systems (Sievers, 1998) |
| 1999 | Salt Lake City, Utah | Wetlands and Remediation (Means and Hinchee, 2000) |
| 1999 | Třeboň, Czech Republic | Transformations of Nutrients in Natural and Constructed Wetlands (Vymazal, 2001) |
| 1999 | Baltimore, Maryland | Wetlands for Wastewater Recycling |
| 1999 | Tartu, Estonia | Constructed Wetlands for Wastewater Treatment in Cold Climates (Mander and Jenssen, 2002) |
| 2000 | Quebec, Canada | INTECOL Wetlands Conference (Pries, 2002) |
| 2000 | Orlando, Florida | 7th International Conference on Wetland Systems for Water Pollution Control (Reddy and Kadlec, 2000) |
| 2001 | Burlington, Vermont | Wetlands and Remediation II (Nehring and Brauning, 2002) |
| 2001 | Třeboň, Czech Republic | Wetlands: Nutrients, Metals, and Mass Cycling (Vymazal, 2003) |
| 2001 | Fort Worth, Texas | On-Site Wastewater Treatment: 9th Symposium on Individual and Small Community Sewage Systems (Mancl, 2001) |
| 2002 | Dar es Salaam, Tanzania | 8th International Conference on Wetland Systems for Water Pollution Control (Mbwette, 2002) |
| 2003 | Borová Lada, Czech Republic | Natural and Constructed Wetlands: Nutrients, Metals, and Management (Vymazal, 2005) |
| 2003 | Tartu, Estonia | Constructed and Riverine Wetlands for Optimal Control of Wastewater at Catchment Scale (Mander, 2003) |
| 2003 | Lisbon, Portugal | The Use of Aquatic Macrophytes for Wastewater Treatment in Constructed Wetlands (Dias and Vymazal, 2003) |
| 2004 | Wexford, Ireland | Nutrient Management in Agricultural Watersheds: A Wetlands Solution (Dunne *et al.*, 2005) |
| 2004 | Avignon, France | 9th International Conference on Wetland Systems for Water Pollution Control (Liénard, 2004) |
| 2005 | Ghent, Belgium | 1st Wetland Pollutant Dynamics and Control (WETPOL) (Tack *et al.*, 2005) |
| 2006 | Třeboň, Czech Republic | Wastewater Treatment, Plant Dynamics, and Management in Constructed and Natural Wetlands (Vymazal, 2008) |
| 2006 | Lisbon, Portugal | 10th International Conference on Wetland Systems for Water Pollution Control (Dias and Vymazal, 2006) |
| 2007 | Padua, Italy | Multi-Functions of Wetland Systems (Borin and Bacelle, 2007) |
| 2007 | Tartu, Estonia | 2nd Wetland Pollutant Dynamics and Control (WETPOL) (Mander *et al.*, 2007) |

initiation dates, and key technical conferences at which the use of wetlands for water quality control was a featured topic.

In the early years of the technology development, it was possible and desirable to identify the numbers of systems, along with their characteristics. The benefit was the ability to document that constructed wetlands were being used in considerable numbers, and therefore were to be accorded some measure of recognition by the regulatory agencies and by the cadre of consulting engineers. For example, the North American treatment wetland database effort (NADB, Knight *et al.*, 1992) catalogued information on 127 treatment wetland systems, which was a springboard to data analyses that advanced the technology. However, growth has been exponential, and by 2005, a volunteer response survey produced a total of 497 small-scale constructed wetlands (flow less than 2,000 $m^3/d$) (Wallace and Knight, 2006). Enumeration is no longer a fruitful exercise, because there is no longer a need to demonstrate weight of numbers. It suffices to recognize that there are now many thousands of

treatment wetlands across the globe. At this stage of technology development, attention is better directed to those systems that have, or are, producing data that enables optimization.

## Development of Treatment Wetlands in North America

Wetland technology progress followed two parallel paths. The first, based on the recognition of the value of natural wetlands in water quality improvement, consisted of studies of projects that intentionally discharged wastewaters to existing wetlands. The second, the implementation of constructed wetlands, both FWS and HSSF, was initiated a few years later.

### Natural Wetlands

Between 1967 and 1972, Howard T. Odum and A.C. Chestnut of the University of North Carolina began a five-year study of using coastal lagoons (with marsh wetland littoral vegetation) for recycling and reuse of municipal wastewaters (Odum, 1985). The studies included an examination of a natural *Spartina* salt marsh ecosystem that was receiving a discharge of secondarily treated wastewater (Camp *et al.*, 1971; Marshall, 1971; McMahan *et al.*, 1972; Stiven and Hunter, 1976).

In 1972, the University of Florida began a research effort directed at assessing the effectiveness of natural cypress wetlands for municipal wastewater recycling. From March 1974 until September 1977, secondarily treated municipal wastewater from a trailer park north of Gainesville was discharged to two isolated cypress wetlands (domes), and two control wetlands were also monitored. Research studies measured nearly all aspects of the physical, chemical, and biological processes occurring in the wastewater and control cypress domes (Ewel and Odum, 1984).

Concurrently, Robert Kadlec and coworkers at the University of Michigan began the first in-depth study of using engineered wetlands for wastewater treatment in a cold climate region. Work at the Porter Ranch peat-based wetland (peatland) located near the community of Houghton Lake, Michigan, began in 1971, with two years of discharges to 32 mesocosm plots in the peatland. A 360 $m^3$/d facility was operated with seasonal discharges for the next three years under the direction of university personnel. A full-scale system was initiated in 1978, and continues with a 2006 flow of 6,032 $m^3$/d during the summer to the Porter Ranch peatland (Figure 1.9). This system continues to operate today, and information from the Houghton Lake Natural Peatland Treatment System represents the longest data set on this aspect of the technology.

The public was not prepared to wait for results from these and other research efforts underway in the 1970s. In 1972, the city of Bellaire, Michigan, began discharging stabilized municipal wastewater to a 16-ha forested wetland (Kadlec, 1983). Although research was conducted on this system, the wetland was the primary means of effluent disposal for the city.



**FIGURE 1.9** The Houghton Lake, Michigan, system utilizes a preexisting peatland for seasonal discharges.

The Reedy Creek Wetland Treatment System was implemented at Walt Disney World, Orlando, Florida, in 1977 (Knight *et al.*, 1987). The Reedy Creek system (Figure 1.10) used two wetlands, one with about 34 ha of natural mixed cypress/hardwood forested swamp, and the second with about 0.2 ha of constructed marsh and 5.6 ha of natural swamp forest to provide advanced wastewater treatment between 1977 and 1991 for monthly average flows as high as 22,700 $m^3$/d (7.2 cm/d). Flow to this wetland was discontinued in 1991 when a zero surface discharge option was implemented through landscape irrigation and groundwater recharge.



**FIGURE 1.10** The Reedy Creek, Florida, system utilizes a preexisting forested wetland for wastewater polishing.

With wetlands being protected by federal law, the use of natural wetlands for wastewater treatment became limited at the end of the 20th century. Hammer and Bastian (1989) during the conference on constructed wetlands in Chattanooga, Tennessee, stated:

> Although some natural wetlands have been effectively used for water quality improvement, we do not wish to encourage additional use. We have recently become aware that natural wetlands are valuable resources that must not be wasted. Much remains to be learned about their many values and functions and the long-term consequences of wetland destruction. However, enough is known to conclude that it is not worth risking the unnecessary loss of any remaining natural wetlands without a better understanding of their important role in biological productivity, fish and wildlife habitat, flood protection, groundwater discharge, base flow stabilization of rivers, and water quality improvement. On the other hand, constructed wetlands may provide a relatively simple and inexpensive solution for controlling many water pollution problems without detrimentally affecting our natural wetlands resources. Although all of the processes are not well understood, constructed wetlands are capable of moderating, removing, or transforming a variety of water pollutants while also providing wildlife and recreational benefits commonly associated with natural wetland systems.

## FWS Constructed Wetlands

In 1973, the Mt. View Sanitary District in Martinez, California, constructed about 8.5 ha of FWS wetland marshes for wildlife habitat and wastewater discharge (James and Bogart, 1989). Also in 1973, the first intentionally engineered, constructed wetland treatment pilot systems in North America were constructed at Brookhaven National Laboratory near Brookhaven, New York. These pilot treatment systems combined a marsh wetland with a pond and a meadow in series and were designated as the meadow/marsh/pond treatment system (Small, 1978). Industrial stormwaters and process waters were also applied to constructed pond/wetland systems as early as 1975 at Amoco Oil Company's Mandan Refinery in North Dakota (Litchfield and Schatz, 1989) (Figure 1.11). In 1976, the communities of Pinetop and Lakeside, Arizona, and in 1977 Show Low, Arizona, created a series of constructed lake/wetland areas for effluent evaporative disposal and wildlife production (Wilhelm *et al.*, 1989).



**FIGURE 1.11** The Mandan, North Dakota, FWS system serves to polish refinery wastewaters.

North America has a rich history of constructing large-scale FWS treatment wetlands over the last 20 years. Florida has a number of the largest constructed wetland treatment areas in the world, including the Lakeland and Orlando constructed wetlands, both of which were started in 1987. Each wetland has about 500 ha for advanced treatment of municipal wastewater. Six treatment wetlands, called stormwater treatment areas, totalling over 16,000 ha, have been built in south Florida. These were designed to achieve a goal of 50 µg/L effluent phosphorus, and the last was placed in operation in 2004. Over 30 km of levees, and a comparable length of canals, are used to enclose each of these constructed treatment wetlands, and convey water. The approximate cost, including supporting research, was $1 billion, of which about $800 million was for construction. These are to treat an estimated thirty-year annual average flow of 4,400,000 $m^3$/d, and remove approximately 80 metric tons of phosphorus per year.

However, FWS treatment wetlands are by no means restricted to warm climates. Pries (1994) documented 67 constructed wetlands in Canada, including some in the Northwest Territories. The North American Database v. 2.0 (1998) has information for a total of 257 sites, 352 systems, and 622 cells from treatment wetlands in North America, and it is known that it was incomplete at the time of issuance. Of these 257 sites, 161 treat municipal wastewater, 10 receive industrial effluents, 68 receive livestock wastewaters, and 16 receive other wastewater types including stormwaters. Of the systems described in NADB v. 2.0, 270 are surface flow, 53 are HSSF, and 8 are hybrids of these two designs.

## HSSF Constructed Wetlands

HSSF wetlands have gained widespread acceptance in North America as well. The first systems were built in 1972 near Seymour, Wisconsin, and researched through 1975 (Spangler *et al.*, 1976b). The researchers concluded that:

> Emergent vegetation has been used to treat wastewater biologically to a degree of purity which suggests that continued research could lead to widespread applicability of the process.

It took about ten years for the concept to develop, and by 1990, 98 HSSF systems were identified in the United States (Reed, 1990; 1991). Most of these (80) were in the southern states. The mean flow to these was 1,250 $m^3$/d, and the mean area was 3,400 $m^2$. Many of these were not properly designed to produce SSF, and operated in the flooded mode (Figure 1.12). In the 1990s, research was conducted at two primary HSSF sites in the United States: Baxter, Tennessee (George *et al.*, 1998), and Minoa, New York (Theis and Young, 2000). Results from those studies complemented the input–output data from other operating wetlands.



**FIGURE 1.12** The Denham Springs, Louisiana, "HSSF" system was a lagoon add-on that could not carry the flow below the media surface.



**FIGURE 1.14** The Salem, Oregon, VF system just after planting. (Photo courtesy of the city of Salem.)

By 2005, an effort to collect volunteer data identified several hundred HSSF systems in the United States, but this survey missed the majority of the single-home systems, because they have no reporting requirements (Wallace and Knight, 2006). For instance, in the United States, over 4,000 single-home HSSF systems are estimated to be in the state of Kentucky alone (Thom *et al.*, 1998). These single-home systems are subject to prescriptive design specifications, and very few produce data that may be used to further the technology development (Figure 1.13).

### VF Constructed Wetlands

Very few VF wetlands have been implemented in North America, and these remain an area of technology development. Pulse-loaded VF systems based on European criteria are gaining increasing acceptance in North America (Kadlec, 2003c) (Figure 1.14). However, sprinkled beds with rotation are also in use (Burgoon *et al.*, 1999). Recirculating designs (based on gravel filters), a form of vertical flow wetlands, have been implemented in Canada (Lemon *et al.*, 1996).

Biosolids wetlands have been implemented at a number of mechanical wastewater treatment plants for stabilization of waste activated sludge, although the technology is not widespread. The current number of biosolids wetlands in North American is not known. There is an increasing interest in the technology from wastewater treatment plant operators, because biosolid wetland systems remain the simplest method to meet federally-mandated pathogen reduction requirements. To date, the technology has been used mainly in cold climates (Figure 1.15), because it is believed freezing during the winter aids in dewatering.

### Treatment Wetlands in Europe

Development of constructed wetlands in Europe started with the work of Käthe Seidel, who began experimenting with aquatic macrophytes for water quality improvement (Seidel, 1953). This work was expanded in the 1950s and 1960s for various waste streams, including phenol wastewaters (Seidel, 1965; 1966), dairy wastewaters (Seidel, 1976), and livestock wastewaters (Seidel, 1961). The system evolved into a series of vertical and HSSF filter beds, and became known most commonly as the Max Planck Institute Process (MPIP) (Brix, 1994a; 1994d). This system was the basis for the "hybrid" wetland systems that were revived at the end of the 20th century.

In the mid-1960s, Seidel began collaboration with Reinhold Kickuth from Göttingen University. This collaboration ended after a few years due to personal reasons. Kickuth went on



**FIGURE 1.13 (A color version of this figure follows page 550)** Single-home HSSF wetland in Comfort Lake, Minnesota.



**FIGURE 1.15** Biosolids stabilization wetland cells at Pine River, Minnesota.

to develop a HSSF wetland process commonly known as the root zone method (RZM). RZM wetlands were constructed with a soil media (typically clay loam to sandy clay) and planted with *Phragmites* in the belief that the root systems of this plant would improve the hydraulic conductivity of the media (Kickuth and Könemann, 1987).

The two scientists and their respective schools became rivals, producing conflicting information that created confusion among wastewater engineers and regulatory authorities (Brix, 1994a). By the 1980s, most constructed wetlands in Germany were RZM systems, although examples of the MPIP system were constructed in St. Bohaire, France (Liénard *et al.*, 1990), and Oaklands Park, United Kingdom (Burka and Lawrence, 1990).

### HSSF Constructed Wetlands

The first full-scale RZM wetland into operation in 1974 in Liebenburg-Othfresen, Germany, for treatment of municipal wastewater (Kickuth, 1977). The area of about 22 ha was originally used to dump waste material (silt, clay, and dross) derived from mining of iron ore. Kickuth's concept of using heavy cohesive soils with low hydraulic conductivity was related to the traditional understanding of soil treatment of sewage, based on the "sewage farming" experiences in the United Kingdom (Cooper and Boon, 1987; Hiley, 1994). However, the predicted increases in the hydraulic conductivity of the bed media from root and rhizome growth did not occur, resulting in overland flow across the surface of the bed (Börner *et al.*, 1998).

In 1983, German designs (based on the root zone method) were introduced in Denmark, where the technology was recognized as being favorable for small community wastewater treatment. By 1987 about 80 horizontal flow constructed wetlands had been built (Brix, 1987; 1998). Despite problems with bed clogging and associated overland (surface) flow, these soil-based systems provided effective treatment if a bed area in excess of 3–5 $m^2$/PE was used. In order to overcome the overland flow problems, later Danish systems were designed with very wide beds and a short flow path (Brix, 1998). However, flow distribution was a problem with these very wide beds and the wetland was subdivided into two or more separate cells that could be loaded separately in order to get better control on the distribution of water (Brix, 1998).

In 1985, following visits to existing German and Danish systems, the first two HSSF constructed wetlands were built in the United Kingdom (where they are commonly called reed bed treatment systems). By the end of 1986, more than 20 systems had been designed (Cooper and Boon, 1987) (Figure 1.16). At the present time, there are over 1,000 systems in the database of the Constructed Wetland Association of the United Kingdom.

One major design change that was implemented in the United Kingdom was to switch to the use of coarser bed media (gravel) in order to maintain SSF within the wetland bed.



**FIGURE 1.16** The HSSF reedbed at Acle, United Kingdom.

In the late 1980s, the first horizontal flow constructed wetlands were built in many European countries. By the 1990s, the technology had become a preferred method for wastewater treatment for small villages and other decentralized wastewater applications (Vymazal *et al.*, 1998).

The Mediterranean countries of Europe have developed large numbers of treatment wetlands, mainly over the past 15 years. Portugal documented 128 constructed wetlands in 2003 (Dias and Martins-Dias, 2003) which had grown to 176 by 2006 (Dias *et al.*, 2006), and they are growing in numbers in Spain, Italy, Greece, and Turkey.

### Vertical Flow Constructed Wetlands

Constructed wetlands with VF date back to the original MPIP process developed by Seidel, where they were utilized as filtration beds in the first stage of the wetland treatment process (see Figures 15.9, 15.10).

The earliest full-scale VF wetlands were termed infiltration (or percolation) fields with VF through a soil or sand medium and with effluent discharged through underdrain pipes. This design was used to treat the wastewater from a recreation site in Lauwersoog, The Netherlands, in 1975 (Greiner and de Jong, 1984; Butijn and Greiner, 1985) (Figure 1.17). The system consisted of a preliminary settling/distribution ditch, four infiltration compartments, and an effluent ditch. Raw wastewater was discharged in the preliminary settling/distribution ditches. After settling, the water was intermittently fed to one of the VF wetland cells, which were alternately loaded and rested. This system has received intensive study (Rijs and Veenstra, 1990; Mueleman, 1999; Mueleman *et al.*, 2002).

VF constructed wetlands in Europe comprise a flat bed of graded gravel topped with sand that is planted with *Phragmites*. The beds are pulse-loaded with a large batch of water to temporarily flood the surface of the bed. Wastewater then percolates down through the bed via unsaturated flow. As the bed drains, air is drawn into the bed, reaerating the microbial biofilms. This pulse loading provides good oxygen transfer. As a result, VF wetland beds are known for their ability to nitrify (Cooper *et al.*, 1996).



**FIGURE 1.17** Vertical flow wetland at Lauwersoog, The Netherlands.

VF constructed wetlands typically provide a good removal of organics and suspended solids, but these systems typically provide little denitrification. Consequently, removal of total nitrogen in these systems is limited.

VF constructed wetlands require less land (1–3 $m^2$/PE) as compared to horizontal flow systems (5–10 $m^2$/PE) but require more operation and maintenance. VF systems are very often used in Austria, Denmark, France, Germany, and the United Kingdom, especially for small sources of pollution. This wetland technology has been adopted in most European countries.

## Hybrid Constructed Wetlands

Different types of constructed wetlands may be combined in order to achieve higher removal efficiency. These systems date back to the original MPIP system of Seidel. Currently, most hybrid systems employ combinations of horizontal and VF wetland cells. The most common configuration to date has been a VF stage followed by horizontal SSF wetland cells.

Over the last ten years, these types of vertical flow–horizontal flow systems were built in many European countries, such as Slovenia (Urbanc-Bercic and Bulc, 1994), Norway (Mæhlum and Stålnacke, 1999), Austria (Mitterer-Reichmann, 2002), and Ireland (O'Hogain, 2003). Hybrid systems are receiving more attention in most European countries because of more stringent requirements for ammonia removal.

An alternate hybrid wetland consisting of a horizontal flow bed followed by VF wetland cells has also been developed (Johansen and Brix, 1996). The large horizontal flow bed is placed first to remove organics and suspended solids and to provide denitrification. A pulse-loaded small VF bed is designed for further removal of organics and suspended solids and to nitrify ammonia to nitrate. A portion of the treated effluent is recirculated back to the influent in order to promote denitrification in the horizontal flow bed and improve total nitrogen removal (Brix *et al.*, 2003). Similar systems have been built in Poland at Sobiechy (Ciupa, 1996) and in Nepal at Dhulikhel in collaboration with Austrian researchers (Laber *et al.*, 1999).

More recent hybrid constructed wetlands use multiple wetland types, including FWS wetlands. An example of this approach can be found at Kõo in Estonia; this system consists of two VF beds, followed by a horizontal flow bed and two FWS wetlands (Mander *et al.*, 2003). In Italy, hybrid constructed wetlands are being successfully used for treatment of concentrated winery wastewaters (Masi *et al.*, 2002). The system at Ornellaia, Italy, consists of two VF beds, followed by a horizontal flow bed and a FWS wetland. The system at Cecchi, Italy, consists of horizontal flow beds followed by a FWS wetland and a pond.

## FWS Constructed Wetlands

The IJsselmeer Polders Development Authority in Flevoland, The Netherlands, constructed the first European FWS wetland in 1967 (Veenstra, 1998). The wetland had a design depth of 0.4 m and the total area was 1 ha. A star-shape layout was chosen in order to obtain optimum utilization of the available area, however, this shape complicated macrophyte harvesting (de Jong, 1976). Therefore, longitudinal channels were added to facilitate mechanical biomass harvesting and system maintenance. The new wetland design included channels of 3 m wide and 200 m long (Figure 1.18), separated by



**FIGURE 1.18** Free water surface (FWS) wetland at Elburg, The Netherlands.

parallel stretches of 3 m, resulting in an increase in land requirement from 5 $m^2$/PE for the star arrangement to 10 $m^2$/PE. The system exhibited a very good treatment effect and by the early 1970s, about 20 FWS wetlands of this type, called *planted sewage farms* (or Lelystad process farms), were in operation in The Netherlands (Greiner and de Jong, 1984; Veenstra, 1998).

In 1968, FWS-constructed wetlands were created in Hungary near Keszthely in order to preserve the water quality of Lake Balaton and to treat municipal wastewater (Lakatos, 1998). The constructed wetland was established in place of an existing natural wetland. The system originally consisted of six ponds 40–60 cm deep with a surface area of 10 ha. The ponds were fed with 8,000 $m^3$/d of mechanically pretreated wastewater. By 1985, the protection of Lake Balaton had grown to include the 1,800-ha Keszthely pond, which turned out to be a submerged aquatic vegetation (SAV) system (Clement *et al.*, 1998).

In contrast to North America, FWS-constructed wetland technology did not spread rapidly throughout Europe, and the main technology focus has been on HSSF and VF systems. However, FWS constructed wetlands are in operation in many European countries (e.g., Norway, Sweden, Denmark, Poland, Estonia, and Belgium). In Sweden, FWS systems have been constructed with nitrogen removal as a primary goal but other aims, such as biodiversity and irrigation, are also taken into consideration (Vymazal, 2006). Sometimes, the aim is to provide phosphorus polishing after chemical treatment and a buffer in case of treatment failure in the conventional treatment plant (Sunblad, 1998) (Figure 1.19). More than 2,350 ha of wetlands have been created in Sweden in the agricultural landscape between 1996 and 2002 in Denmark about 3,200 ha have been created prior to 2004 (Vymazal, 2006).

### Biosolids Wetlands

The concept of vertical flow wetlands to remove organic matter extends back to the original system of Seidel (Seidel, 1965), but is also used in a modern context for VF wetlands in France, which are typically designed to accumulate biosolids



**FIGURE 1.19** The treatment wetland at Oxelösund, Sweden.



**FIGURE 1.20** Biosolids mineralization reedbeds at Skovby, Denmark. The bed in the foreground is newly excavated and replanted, and is receiving initial doses of biosolids. The bed in the left background is a mature stand of *Phragmites* that has been receiving biosolids for several years.

associated with raw sewage (Boutin *et al.*, 2002; Chazarenc and Merlin, 2004; Molle *et al.*, 2004a). Wetland beds designed specifically for biosolids dewatering have been most extensively developed in Denmark, where over 110 systems have been constructed since 1988 (Nielsen, 2006) (Figure 1.20). The largest current system is in Kolding, Denmark (123,000 PE). Use of wetlands for stabilization of organic biosolids is expanding throughout Europe (DeMaeseneer, 1997; Barjenbruch *et al.*, 2002; Obarska-Pempkowiak and Sobocinski, 2002; Lesavre and Iwema, 2002).

## Treatment Wetlands in Australia, New Zealand, Africa, Asia, and South America

### Australia

Aquatic macrophytes in Australia were initially evaluated for water quality improvement in the 1970s (Mitchell, 1976). In the 1980s, pilot-scale HSSF wetlands were evaluated for the treatment of piggery wastes and abattoir wastewater (Finlayson and Chick, 1983; Finlayson *et al.*, 1987). Extensive pilot-scale experiments, for both HSSF and FWS systems, were carried out at University of Western Sydney (Bavor *et al.*, 1987).

In 1992, the Cooperative Research Center for Constructed Wetlands was established, and several research projects were conducted on both FWS (e.g., the Byron Bay, New South Wales, full-scale system) and SSF wetlands (e.g., the Coff's Harbor, New South Wales, full-scale system). In the early 1990s, nine pilot wetlands were established in Queensland, of which eight were FWS (Greenway and Woolley, 1999; 2001).

Treatment wetlands for industrial wastewaters have been implemented; for instance, Noller *et al.* (1994) lists results from seven mine water wetlands in northern Australia, and oil refinery waters have also been treated (Simi, 2000).

Single-home HSSF wetlands have been extensively studied by Davison *et al.* (2001), but such domestic applications are still localized. In contrast, the application of stormwater

treatment wetlands has proceeded with a considerable growth in numbers, in part spurred by the research endeavors of Wong and coworkers (Wong *et al*., 1999; Wong *et al*., 2006).

### New Zealand

Some of the earliest constructed wetlands in New Zealand were for treatment of meat processing waters (Van Oostrom and Cooper, 1990; Van Oostrom and Russell, 1994). Research at the National Institute for Water and Air (NIWA), under the direction of C.C. Tanner, produced many valuable insights into the performance of HSSF wetlands over the period from 1994 to the present. The development and growth of constructed wetland technology has been stimulated by low investment and operating costs, and the technology, to some extent, addresses the Maori cultural and spiritual values.

According to a survey carried out by Tanner *et al.* (2000), constructed wetlands had been adopted enthusiastically by many New Zealand communities as a cost-effective means of secondary and tertiary wastewater treatment. Out of 83 constructed wetlands for wastewater treatment, excluding those treating stormwaters and farm dairy wastes, FWS treatment wetlands were most common (45%), followed by SSF and hybrid systems (35% and 14%, respectively). The remaining systems were called "enhanced natural wetlands." The surface flow systems were much larger (average size 2.2 ha) than those with SSF (average size 0.4 ha).

At present, constructed wetlands in New Zealand are also used to treat agricultural waters. Dairy runoff and pasture runoff are the focal points of new applications of wetlands.

### Africa

Since the mid-1980s, the concept of using constructed wetlands gained support in Southern Africa, and by 1990, there were approximately 30 systems either in operation or under construction (Wood and Hensman, 1989; Batchelor *et al*., 1990). These were intended to serve a number of functions, including treating raw sewage and secondary domestic effluents, septic tank and oxidation pond effluents, stormwaters, agricultural and aquaculture wastes, and a variety of industrial and mining wastewaters.

In the late 1990s, wetlands were piloted in Egypt, at Alexandria and at Abbu Attwa, Ismailia (Butler *et al*., 1990). Several systems were implemented in Morocco (Mandi *et al*., 1998; Radoux *et al*., 2003). A very extensive constructed wetland demonstration project, the Bar el Baqar drain, located on one of the branches of the Nile as it enters Mediterranean estuaries, concluded in late 2006. Constructed wetlands have become more popular in central Africa, and there are now many examples of all types of constructed wetlands treating municipal sewage as well as industrial wastewaters and mine drainage waters (Kivaisi, 2001; Kaseva *et al*., 2002; Mbuligwe, 2005; Abira *et al*., 2005; Bojcevska *et al*., 2006). Biosolids wetlands have also been implemented in Cameroon (Noumsi *et al*., 2006) for stabilization of fecal sludges from primary settling tanks.



**FIGURE 1.21** The treatment wetland system at Bainikeng, China.

### Asia

The traditional expertise of Asian farmers in recycling human and animal wastes through aquaculture provides a good basis for what we choose to call "engineered wetland treatment systems" (Abbasi, 1987). As early as 1969, Sinha and Sinha reported on the use of the water hyacinth to treat digested sugar factory wastes. However, the first information about the use of constructed wetlands with emergent vegetation appeared only in the early 1990s (Juwarkar *et al*., 1992). During the IWA conference in China in 1994, many papers on both horizontal and VF constructed wetlands from Asia, and especially China, were presented (Figure 1.21). Difficulties in language and communication have likely impeded the transfer of Asian information to the western world. At present, many constructed wetlands with emergent vegetation are in operation in India, China, Korea, Taiwan, Japan, Nepal, Malaysia, and Thailand for various types of wastewater.

### South America

Since 1980, research has been conducted in Brazil on the possibility of the use of water hyacinth ponds in combination with vertical upflow constructed wetlands planted with rice (Salati, 1987). However, other types of constructed wetlands with emergent macrophytes have been adopted (Dallas *et al*., 2004). There are limited numbers of constructed wetlands with emergent vegetation in South America, but systems are in operation in Brazil, Peru, Colombia, Honduras, Ecuador, Uruguay, Argentina, and also in Central America (Platzer *et al*., 2002; Masi *et al*., 2006).

## SUMMARY

Constructed wetlands are mechanically simple treatment systems that rely primarily on passive treatment processes. These treatment systems are very favorable for use in rural settings or areas of low population density because they are relatively low-maintenance (compared to other treatment alternatives) and can usually be constructed from local

materials. From global perspective, treatment wetland systems are gaining popularity as the market for cost-effective wastewater management expands in both developed and developing countries. This is primarily because treatment wetlands are perceived as a cost-effective and environmentally conscious treatment technology. There is also a growing realization that urban expansion may be best served by satellite wastewater treatment systems, rather that the continued expansion of centralized plants.

Natural wetlands have been used as receptors for wastewater since ancient times. The 20th century brought about the development of man-made wetland systems that are designed to emphasize specific characteristics of the natural wetland environment, aiming to improve the overall treatment capacity of the system. From this, three types of engineered wetlands have evolved for use in small-scale applications: FWS, HSSF, and VF.

FWS wetlands are similar to natural wetlands in that they contain areas of open water, floating vegetation, and emergent vegetation. They offer habitat benefits similar to natural wetlands, and invariably attract a variety of wildlife. These wetlands typically are used to polish effluent from secondary treatment processes such as lagoons, trickling filters, or activated sludge systems. They are rarely used as a stand-alone secondary treatment process due to their size and buffer requirements. FWS systems are virtually always the choice for stormwater treatment, and for animal wastewater treatment.

HSSF wetlands differ from FWS wetlands in that the wastewater is kept belowground. These wetlands are comprised of a lined gravel or soil-based bed planted with emergent vegetation. The wastewater is treated as it flows through the gravel media and around the roots and rhizomes of the plants. Because the wastewater is not exposed during the treatment process, the risk of pathogen exposure is minimized. HSSF wetlands are typically used to treat primary effluent to secondary treatment standards.

Vertical flow (VF) wetlands have found their widest application in Europe where the design goal is to produce a nitrified effluent. Because these systems accumulate biosolids on the surface of the bed, they may be incompatible with North American regulatory standards, which typically prohibit the surface exposure of fecal material. Nevertheless, VF systems are being used more and more in a global context.

Biosolids dewatering wetlands, a version of VF systems, are gaining an increasing level of support from operators of traditional sewage treatment works due to their simplicity and low operation and maintenance (O&M) requirements. This is especially true in cold climates where freezing promotes dewatering of accumulated biosolids.

The information summarized in this book represents the efforts of hundreds of scientists and engineers over the past three decades. While a synthesis of this massive collection of information is necessary to carry the wetland treatment technology to a wider audience, it is still beneficial for the reader to refer to the original sources for more details and a regional perspective, and to examine the evolution of wetland engineering during this formative period. The reader is encouraged to examine the references cited throughout the text and provided at the end of this book.

# 2 Hydrology and Hydraulics

The success or failure of a treatment wetland is contingent upon creating and maintaining correct water depths and flows. In this chapter, the processes that add and subtract water from the wetland are discussed, together with the relationships between flow and depth. Internal water movement in wetlands is a related subject, which is critical to understanding of pollutant reductions.

The water status of a wetland defines its extent, and is the determinant of plant species composition in natural wetlands (Mitsch and Gosselink, 2000). Hydrologic conditions also influence the soils and nutrients, which in turn influence the character of the biota. Flow and storage volume determine the length of time that water spends in the wetland, and thus the opportunity for interactions between waterborne substances and the wetland ecosystem.

The ability to control water depths is critical to the operation of treatment wetlands. This operational flexibility is needed to maintain the hydraulic regime within the hydrologic needs of desired wetland plant species, and is also needed to avoid unintended operational consequences, such as inlet zone flooding of horizontal subsurface flow (HSSF) treatment wetlands. It is therefore necessary to understand the hydraulic factors that relate depth and flow rate, including vegetation density and aspect ratio. In free water surface (FWS) wetlands, this requires an understanding of stem drag effects on water surface profiles. For HSSF and vertical flow (VF) wetlands, there are additional issues concerning the bed media size, hydraulic conductivity, and clogging.

## 2.1 WETLAND HYDROLOGY

Water enters wetlands via streamflow, runoff, groundwater discharge and precipitation (Figure 2.1). These flows are extremely variable in most instances, and the variations are stochastic in character. Stormwater treatment wetlands generally possess this same suite of inflows. Treatment wetlands dealing with continuous sources of wastewater may have these same inputs, although streamflow and groundwater inputs are typically absent. The steady inflow associated with continuous source treatment wetlands represents an important distinguishing feature. A dominant steady inflow drives the ecosystem toward an ecological condition that is somewhat different from a stochastically driven system.

Wetlands lose water via streamflow, groundwater recharge, and evapotranspiration (Figure 2.1). Stormwater treatment wetlands also possess this suite of outflows. Continuous source treatment wetlands would normally be isolated from groundwater, and the majority of the water would leave via streamflow in most cases. Evapotranspiration (*ET*) occurs with strong diurnal and seasonal cycles, because it is driven by solar radiation, which undergoes such cycles. Thus, *ET* can be an important water loss on a periodic basis.

Wetland water storage is determined by the inflows and outflows together with the characteristics of the wetland basin. Depth and storage in natural wetlands are likely to be modulated by landscape features, such as the depth of an adjoining water body or the conveyance capacity of an outlet stream. Large variations in storage are therefore possible, in response to the high variability in the inflows and outflows. Indeed, some natural wetlands are wet only a small fraction of the year, and others may be dry for interim periods of several years. Such periods of dry-out have strong implications for the vegetative structure of the ecosystem. Constructed treatment wetlands, on the other hand, typically have some form of outlet water level control structure. Therefore, there is little or no variation in water level, except in stormwater treatment wetlands. Dry-out in treatment wetlands does not normally occur, and only the vegetation that can withstand continuous flooding will survive.

The important features of wetland hydrology from the standpoint of treatment efficiency are those that determine the duration of water–biota interactions, and the proximity of waterborne substances to the sites of biological and physical activity. There is a strong tendency in the wetland treatment literature to borrow the detention time concept from other aquatic systems, such as "conventional" wastewater treatment processes. In purely aquatic environments, reactive organisms are distributed throughout the water, and there is often a clear understanding of the flow paths through the vessel or pond. However, wetland ecosystems are more complex, and therefore require more descriptors.

### Hydrologic Nomenclature

Literature terminology is somewhat ambiguous concerning hydrologic variables. The definitions used in this book are specified below. The notation and parent variables are illustrated in Figure 2.1.

#### Hydraulic Loading Rate

The hydraulic loading rate (HLR, or $q$) is defined as the rainfall equivalent of whatever flow is under consideration. It does not imply uniform physical distribution of water over the wetland surface. In FWS wetlands, the wetted area is

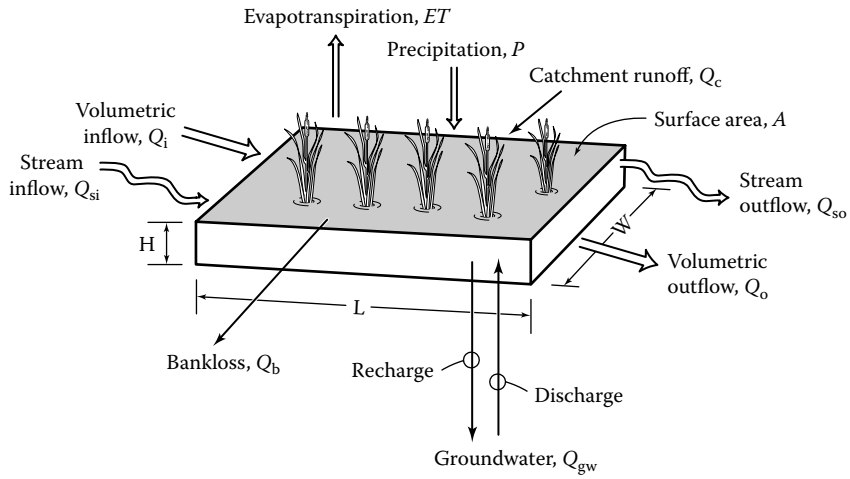



**FIGURE 2.1** Components of the wetland water budget. (From Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

usually known with good accuracy, because of berms or other confining features. The defining equation is:

$$q = \frac{Q}{A} \tag{2.1}$$

where

$q$ = hydraulic loading rate (HLR), m/d
$A$ = wetland area (wetted land area), $m^2$
$Q$ = water flow rate, $m^3/d$

The definition is most often applied to the wastewater addition flow at the wetland inlet: $q_i = Q_i/A$. The subscript $i$, which denotes the inlet flow, is often omitted for simplicity.

Some wetlands are operated with intermittent feed, notably vertical flow wetlands. Under these circumstances, the term hydraulic loading rate refers to the time average flow rate. The loading rate during a feed portion of a cycle is *the instantaneous hydraulic loading rate*, which is also called the *hydraulic application rate*. Some wetlands are operated seasonally, for instance, during warm weather conditions in northern climates. Although these are in some sense intermittently fed, common usage is to refer to the loading rate during operation and not to average over the entire year. This means the instantaneous loading rate is used and not the annual average loading rate.

### Mean Water Depth

Mean water depth is here denoted by the variable $h$. In FWS wetlands, the mean depth calculation requires a detailed survey of the wetland bottom topography, combined with a survey of the water surface elevation. The accuracy and precision must be better than normal, because of the small depths usually found in FWS wetlands. The two surveys combine to give the local depth:

$$h = H - G \tag{2.2}$$

where

$G$ = local ground elevation, m
$h$ = water depth, m
$H$ = local water elevation, m

As-built surveys under dry conditions may not suffice for determination of ground levels, because of possible soil swelling and lift upon wetting. If the substrate is a peat or muck, there is not a well defined soil-water interface. Common practice in that event is to place the surveyor's staff "firmly" into the diffuse interface. Water surface surveys may be necessary in situations where head loss is incurred. This includes many HSSF wetlands, and some larger, densely vegetated FWS wetlands. Local water depth is then determined as the difference between two field measurements, and hence is subject to double inaccuracy.

The difficulties outlined above have prevented accurate mean depth determinations in many treatment wetlands. For example, detailed bathymetric surveys were conducted for a number of 0.2-ha FWS "test cells" in Florida (SFWMD, 2001) (Table 2.1). These were designed to be flat bottom wetlands, but proved to be quite irregular. The average coefficient of spatial variation in bottom elevations for seven of the ten cells was 39%. More importantly, there are errors ranging from –53 to +43% in the nominal volume of water in the wetlands. Errors of this magnitude have important consequences in the determination of nominal detention time.

HSSF wetlands typically have nonuniform hydraulic gradients due to clogging of the inlet region, as discussed further in this chapter. Therefore, the water depth may not be either flat or uniform in HSSF systems.

### Wetland Water Volume and Nominal Detention Time

#### Free Water Surface Wetlands

For a FWS wetland, the nominal wetland water volume is defined as the volume enclosed by the upper water surface

**TABLE 2.1**
**Bathymetry of Ten FWS Wetlands at the Everglades Nutrient Removal Project**

| Wetland Cell | Water Area ($m^2$) | Theoretical Depth (cm) | Measured Depth (cm) | Theoretical Volume ($m^3$) | Measured Volume ($m^3$) | Percent Difference |
|---|---|---|---|---|---|---|
| STC 1 | 2,251 | 60.0 | 54.9 | 1,255 | 1,140 | 10% |
| STC 2 | 2,296 | 15.0 | 12.4 | 341 | 280 | 22% |
| STC 4 | 2,474 | 30.5 | 21.3 | 754 | 528 | 43% |
| STC 9 | 2,534 | 32.6 | 45.4 | 826 | 1,151 | –28% |
| STC 15 | 2,731 | 60.0 | 76.6 | 1,449 | 1,902 | –24% |
| NTC 1 | 2,468 | 63.4 | 74.4 | 1,565 | 1,835 | –15% |
| NTC 5 | 2,747 | 60.0 | 79.0 | 1,449 | 1,968 | –26% |
| NTC 7 | 2,400 | 15.0 | 28.2 | 341 | 651 | –48% |
| NTC 8 | 2,422 | 15.0 | 31.4 | 341 | 728 | –53% |
| NTC 15 | 2,731 | 63.4 | 96.0 | 1,731 | 2,622 | –34% |

*Note:* STC = South Test Cell Site; NTC = North Test Cell Site.

and the bottom and sides of the impoundment. For a VF or HSSF wetland, it is that enclosed volume multiplied by the porosity of the media. *Actual wetland detention time* ($\tau$) is defined as the wetland water volume involved in flow divided by the volumetric water flow:

$$\tau = \frac{V_{active}}{Q} = \frac{\varepsilon\, h\, A_{active}}{Q} \tag{2.3}$$

where

$Q$ = flow rate, $m^3/d$
$A_{active}$ = area of wetland containing water in active flow, $m^2$
$h$ = wetland water depth, m
$V_{active}$ = volume of wetland containing water in active flow, $m^3$
$\varepsilon$ = porosity (fraction of volume occupied by water), dimensionless
$\tau$ = detention time, d

It is sometimes convenient to work with the nominal parameters of a given wetland. To that end, a *nominal detention time* ($\tau_n$) is defined:

$$\tau_n = \frac{V_{nominal}}{Q} = \frac{(LWh)_{nominal}}{Q} \tag{2.4}$$

A very common alternative designation for nominal detention time is HRT. Equation 2.3 is a rather innocuous relation, but has no less than four difficulties, which have led to misunderstandings in the literature. First, there is ambiguity about the choice of the flow rate: Should it be inlet, or outlet, or an average? Differences in inlet and outlet flow rates are further discussed in this chapter.

Second, for FWS systems, some of the wetland volume is occupied by stems and litter, such that $\varepsilon \leq 1$. This quantity is difficult to measure, because of spatial heterogeneity, both vertical and horizontal. It is known to be approximately 0.95 for cattails in a northern environment (Kadlec, 1998), and for submerged aquatic vegetation (SAV) systems in the Everglades (Chimney, 2000), and for an emergent community (Lagrace *et al.*, 2000, as cited by U.S. EPA 1999).

Third, not all the water in a wetland may be involved in active flow. Stagnant pockets sometimes exist, particularly in complex geometries. As a result, $A_{active} \leq A = L \cdot W$. A *gross areal efficiency* may be defined as $\eta = A_{active}/A$. Fourth, the mean water depth ($h$) is difficult to determine with a satisfactory degree of accuracy, especially for large wetlands. That variability translates directly to a comparable uncertainty in the water depths, as noted in Table 2.1. These effects may be empirically lumped, and a volumetric efficiency ($e_V$) defined as:

$$e_V = \frac{V_{active}}{(LWh)_{nominal}} = \frac{\varepsilon\, \eta\, h}{h_{nominal}} \tag{2.5}$$

where

$e_V$ = wetland volumetric efficiency, dimensionless
$V_{active}$ = active wetland volume, $m^3$
$\varepsilon$ = fraction of volume occupied by water, dimensionless
$\eta$ = gross areal efficiency, dimensionless
$h$ = water depth, m
$h_{nominal}$ = nominal, water depth, m
$LWh_{nominal}$ = nominal wetland volume, $m^3$

It is then clear that:

$$\tau = e_V \tau_n \tag{2.6}$$

Volumetric efficiency reflects ineffective volume within a wetland, compared to presumed nominal conditions. Portions of the nominal volume are blocked by submerged biomass ($\varepsilon$), bypassed ($\eta$), or do not exist because of poor bathymetry ($h/h_{nominal}$).

Confusion in nomenclature exists in the literature, where $e_V$ is sometimes identified as *wetland porosity.* For dense emergent vegetation in FWS wetlands, this has presumptively been assigned a value in the range 0.65–0.75 (Reed *et al.*, 1995; Crites and Tchobanoglous, 1998; Water Environment Federation, 2001) (all of which use the symbol $n$ in place of $e_V$). U.S. EPA (1999; 2000a) presumptively assigned the range 0.7–0.9 (both of which use the symbol $\varepsilon$ in place of $e_V$).

It may be assumed that conservative tracer testing will provide a direct measure of the actual detention time in a wetland (Fogler, 1992; Levenspiel, 1995). Then, via Equation 2.6, there is a direct measure of $e_V$, although there is no knowledge gained about the three contributions to $e_V$ by this process. At this point in the development of constructed wetland technology, there have been numerous such tracer tests. Summary results from 120 tests on 65 ponds and FWS wetlands present some insights (Table 2.2). First, the range of values for wetlands is indeed from 0.7 to over 0.9. But the range is even lower for basins devoid of vegetation, 0.55 to 0.9. That observation applies to the Stairs (1993) studies, which show empty basins with the same or lower $e_V$ than identical geometries with plants (Table 2.2). This is a strong indication that the term *porosity* is a misnomer, because $e_V$ is more strongly influenced by $\eta$ and $h/h_{nominal}$.

## Horizontal Subsurface Flow Wetlands

There is a very similar definition of $e_V$ for HSSF systems:

$$e_V = \frac{V_{active}}{V_{nominal}} = \frac{\eta(\varepsilon V_{bed})}{V_{nominal}} \tag{2.7}$$

where

$e_V$ = volumetric efficiency, dimensionless
$\varepsilon$ = wetland bare media porosity, dimensionless
$V_{bed}$ = actual wetland volume (water plus submerged media), $m^3$
$V_{nominal}$ = nominal wetland volume, $m^3$
$\eta$ = gross volumetric efficiency, dimensionless

There is also uncertainty about the volumetric efficiency of subsurface flow wetlands. The mean porosity of a clean sand or gravel media is apt to be in the range 0.30–0.45 (Table 2.3). But, in an operational wetland, roots block some fraction of the pore space, as do accumulations of organic and mineral matter associated with treatment, which is accounted for by the gross areal efficiency, $\eta$. Roots block the upper horizons, and mineral matter preferentially settles to the bottom void spaces. Canister measurements of void fraction are not accurate, because of vessel wall effects and compaction problems. Attempts to measure water-filled void fraction by wetland draining have been thwarted by hold up of residual water. Wetland filling is an unexplored option for porosity determination. HSSF wetlands are often small enough to preclude significant errors in the determination of the bed or water depth, and thus it is expected that the ratio $V_{bed}/V_{nominal}$ is close to unity. It is therefore surprising to find a relatively wide spread in the measured values of $e_V$ (Table 2.3). The range across the individual measurements was $0.15 < e_V < 1.38$. Interestingly, the mean across 22 HSSF wetlands is $e_V = 0.83$, which is virtually identical to that for FWS systems.

## Spatial Flow Variation

There is obviously a possible ambiguity that results from the choice of the flow rate that is used in Equation 2.3 or 2.4.

**TABLE 2.2**
**Hydraulic Characteristics of Ponds and Wetlands**

| Ponds (0.61–2.44 m deep) | Tests | Area ($m^2$) | L:W | Volumetric Efficiency, $e_V$ | Reference |
|---|---|---|---|---|---|
| Three small scale | 24 | 60–65 | 11.3 | 0.91 | Lloyd *et al.* (2003) |
| One lab tank | 3 | 75 | 6.75 | 0.74 | Mangelson (1972) |
| Three pilot scale | 3 | 1,148 | 4 | 0.55 | Peña *et al.* (2000) |
| One pilot scale | 5 | 1,323 | 3 | 0.74 | Stairs (1993) |
| Ten dredge ponds | 10 | 2,860–378,000 | 2.76 | 0.58 | Thackson *et al.* (1987) |
| **Mean** | | | **5.56** | **0.70** | |
| **Wetlands (0.3–0.8 m deep)** | **Tests** | **Area ($m^2$)** | **L:W** | **Volumetric Efficiency, $e_V$** | **Reference** |
| Four pilots | 18 | 1,323 | 3 | 0.78 | Stairs (1993) |
| Six pilots | 6 | 1,000–4,000 | 5.83 | 0.86 | (1) |
| Sixteen pilots | 24 | 1,200–13,400 | 3.95 | 0.69 | (2) |
| Twenty-one pilots | 27 | 2,700 | 3.30 | 0.96 | (3) |
| **Mean** | | | **4.02** | **0.82** | |

*Sources:* Unpublished data: (1) Champion Paper, (2) city of Phoenix, (3) Everglades Test Cells.

**TABLE 2.3**
**Volumetric Efficiency of HSSF Wetlands**

| Study | Number of Tests | Wetlands | Porosity, ε | Volumetric Efficiency, $e_V$ | Combined Effect, $\varepsilon \cdot e_V$ |
|---|---|---|---|---|---|
| García (2003) | 6 | 6 | 0.40 | 1.08 | 0.43 |
| Chazarenc *et al.* (2003) | 8 | 1 | 0.33 | 0.76 | 0.25 |
| Rash and Liehr (1999) | 5 | 2 | 0.41 | 0.28 | 0.12 |
| Grismer *et al.* (2001) | 2 | 2 | 0.36 | 1.02 | 0.37 |
| Bavor *et al.* (1988) | 3 | 3 | 0.33 | 0.93 | 0.31 |
| Marsteiner (1997) | 3 | 3 | 0.37 | 0.77 | 0.29 |
| George *et al.* (1998) | 5 | 5 | 0.36 | 1.08 | 0.40 |
| **Mean or Total** | **32** | **22** | **0.37** | **0.83** | **0.30** |

Wetlands routinely experience water gains (precipitation) and losses (evapotranspiration, seepage), so that outflows differ from inflows. If there is net gain, the water accelerates; if there is net loss, the water slows. A rigorously correct calculation procedure involves integration of transit times from inlet to outlet.

When there are local variations in total flow and water volume, the correct calculation procedure must involve integration of transit times from inlet to outlet. For steady flows, it may be shown that (Chazarenc *et al.*, 2003):

$$\tau_{an} = \tau_i \left( \frac{\ln(R)}{R-1} \right) \tag{2.8}$$

where

$R = Q_o / Q_i$, water recovery fraction, dimensionless
$Q_i$ = inlet flow rate, $m^3/d$
$Q_o$ = outlet flow rate, $m^3/d$
$\tau_{an}$ = actual nominal detention time, d
$\tau_i$ = inlet flow-based nominal detention time, d

In terms of detention time alone, moderate amounts of atmospheric gains or losses ($P - ET$) are not usually of great importance, although there is ambiguity in the choice of flow rate ($Q$). Some authors base the calculation on the average flow rate (inlet plus outlet ÷ 2). This approximation is good to within 4% as long as the water recovery fraction is $0.5 < R < 2.0$.

## Velocities and Hydraulic Loading

The relation between nominal detention time and hydraulic loading rate is:

$$q = \frac{Q_i}{LW} = \frac{\varepsilon h}{\tau_n} \tag{2.9}$$

where

$q$ = hydraulic loading rate, m/d
$Q_i$ = inlet flow rate, $m^3/d$
$L$ = wetland length, m
$W$ = wetland width, m
$\varepsilon$ = porosity of wetland bed media, dimensionless
$h$ = water depth, m
$\tau_n$ = nominal hydraulic retention time, d

Thus, it is seen that hydraulic loading rate is inversely proportional to nominal detention time for a given wetland depth. Hydraulic loading rate therefore embodies the notion of contact duration, just as nominal detention time does.

The *actual water velocity* ($v$) is that which would be measured with a probe in the wetland—a spatial average. In terms of the notation used here:

$$v = \frac{Q}{\varepsilon h W} \tag{2.10}$$

where

$v$ = actual water velocity, m/d
$Q$ = flow rate, $m^3/d$
$W$ = wetland width, m
$\varepsilon$ = wetland bed porosity, dimensionless
$h$ = water depth, m
$\varepsilon hW$ = open area perpendicular to flow, $m^2$

It is noted that there is large spatial and temporal variation in $v$, and hence individual spot measurements may be as much as a factor of ten different from the mean. Field investigations tend to have a bias towards high local measurements because probes do not easily find small pockets of stagnant water.

The *superficial water velocity* ($u$) is the empty wetland velocity—again, a spatial average. In terms of the notation used here:

$$u = \frac{Q}{hW} \tag{2.11}$$

where

$u$ = superficial water velocity, m/d
$Q$ = flow rate, $m^3/d$
$W$ = wetland width, m
$h$ = water depth, m
$hW$ = total wetland area perpendicular to flow, $m^2$

For FWS wetlands, there is not much difference between $u$ and $v$, because FWS porosity is nearly unity (typically around 0.95). However, there is a large difference for HSSF systems because of the porosity of the bed media (typically around 0.35–0.40). Superficial water velocity ($u$) is used in the technical literature on water flow and porous media, and care must be taken to avoid misuse of those literature results.

The relation between superficial and actual velocities is:

$$u = \varepsilon v \tag{2.12}$$

where

$u$ = superficial water velocity, m/d
$\varepsilon$ = wetland bed porosity, dimensionless
$v$ = actual water velocity, m/d

## Overall Water Mass Balances

Transfers of water to and from the wetland follow the same pattern for surface and subsurface flow wetlands (see Figure 2.1). In treatment wetlands, wastewater additions are normally the dominant flow, but under some circumstances, other transfers of water are also important. The dynamic overall water budget for a wetland is:

$$Q_i - Q_o + Q_c - Q_b - Q_{gw} + Q_{sm} + (P \times A) - (ET \times A) = \frac{dV}{dt} \tag{2.13}$$

where

$A$ = wetland top surface area, $m^2$
$ET$ = evapotranspiration rate, m/d
$P$ = precipitation rate, m/d
$Q_b$ = bank loss rate, $m^3/d$
$Q_c$ = catchment runoff rate, $m^3/d$
$Q_{gw}$ = infiltration to groundwater, $m^3/d$
$Q_i$ = input wastewater flowrate, $m^3/d$
$Q_o$ = output wastewater flowrate, $m^3/d$
$Q_{sm}$ = snowmelt rate, $m^3/d$
$t$ = time, d
$V$ = water storage (volume) in wetland, $m^3$

## Inflows and Outflows

Most moderate to large scale facilities will have input flow measurement; a smaller number of facilities will have the capability of independently measuring outflows as well as inflows. Due a lack of outlet flow measurements, the overall water budget Equation 2.13 is often used to calculate the estimated outflow rate. Usually, only rainfall is a significant addition, and only $ET$ is a significant subtraction, to the inflow, simplifying the analysis. This calculation is most easily performed when there is no net change in storage.

The change in storage ($\Delta V$) over an averaging period ($\Delta t$) can be a significant quantity compared to other terms in the water budget. For example, if the nominal detention time in the wetland is 10 days, then a 10% change in stored water represents one day's addition of wastewater. Because water depths in treatment wetlands are typically not large, changes of a few centimeters may be important over short averaging periods. If there is significant infiltration, there are two unknown outflows ($Q_o$ and $Q_{gw} + Q_b$), and the water budget alone is not sufficient to determine either outflow by difference.

## Rainfall

Rainfall amounts may be measured at or near the site for purposes of wetland design or monitoring. However, the gauging location must not be too far removed from the wetland, because some rain events are extremely localized.

For most design purposes, historical monthly average precipitation amounts suffice. These may be obtained from archival sources, such as Climatological Data, a monthly publication of the National Oceanic and Atmospheric Administration (NOAA), National Climatic Data Center, Asheville, North Carolina. In the United States, a very large array of climatological data products are available online at www.ncdc.noaa.gov/oa/climate/climateproducts.html. As an illustration of that service, the (free) normal precipitation map is shown in Figure 2.2.

The total catchment area for a wetland is likely to be just the area enclosed by the containing berms and roads; and that area is easily computed from site characteristics. Rainfall on the catchment area will, in part, reach the wetland basin by overland flow, in an amount equal to the runoff factor times the rainfall amount and the catchment area (Figure 2.3). A very short travel time results in this flow being additive to the rainfall:

$$Q_c = \Psi P A_c \tag{2.14}$$

where

$Q_c$ = flow rate from contributing catchment area, $m^3/d$
$A_c$ = catchment surface area, $m^2$ (does not include the net wetland area)
$\Psi$ = catchment runoff coefficient, dimensionless (1.0 represents an impervious surface)
$P$ = precipitation, m

For small and medium sized wetlands, the catchment area will typically be about 25% of the wetland area, as it is for the Benton, Kentucky, system, for example. About 20% of a site will be taken up by berms and access roads which may drain to the wetland. Runoff coefficients are high, because the berms are impermeable; a range of 0.8–1.0 might be typical. The combined result of impermeable berms, their necessary area, coupled with quick runoff, is an addition, of about 20–25% to direct rainfall on the bed.

## Dynamic Rainfall Response

Many treatment wetland systems are fed a constant flow of wastewater. There is therefore a strong temptation amongst

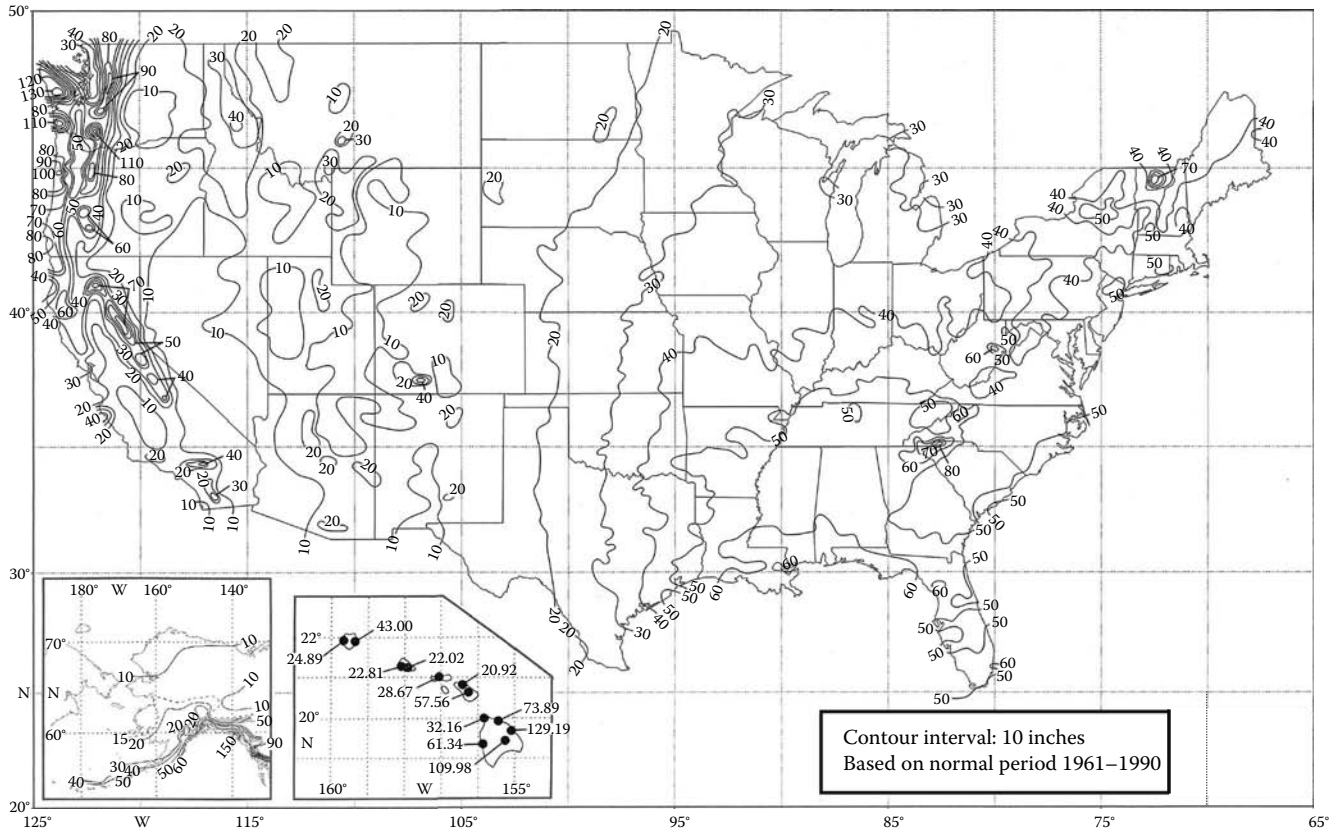



**FIGURE 2.2** Normal precipitation map for the United States.

wetland designers to visualize a relatively constant set of system operating parameters—depths and outflows in particular. This is not the case in practice. There may be significant outflow response to rain events. A sudden rain event, such as a summer thunderstorm, will raise water levels in the wetland. The amount of the level change is magnified by catchment effects, and bed porosity in the case of HSSF systems. A relatively small 3-cm rain event can raise HSSF bed water levels by more than 10 cm. This often exceeds the available head space in the wetland bed. As a result, HSSF wetlands typically experience short-term flooding in response to large storm events and berm heights are usually designed to temporarily store a specified amount of rainfall (such as a 25-year, 24-hour storm event) above the HSSF bed. In any case, outflows from the system increase greatly as the rainwater flushes from the system.

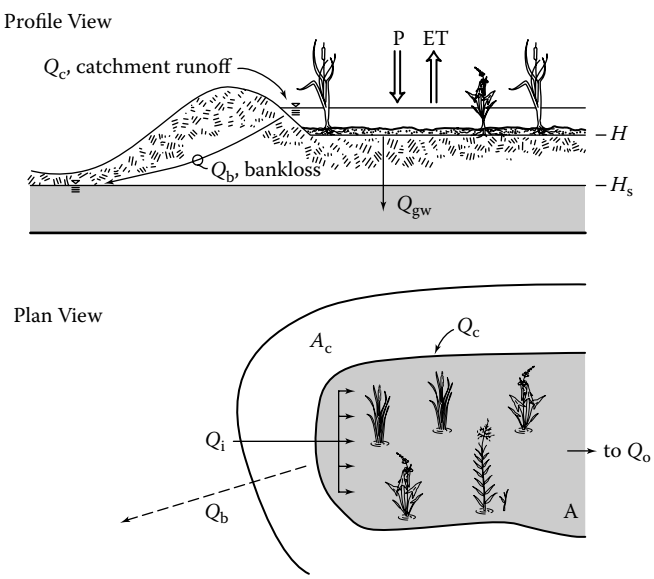



**FIGURE 2.3** Water budget quantities. (Adapted from Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

As an illustration, consider Cell #3 at Benton, Kentucky, in September, 1990. Figure 2.4 shows a rain event of about 2 cm occurring at noon on September 10, 1990. The HSSF bed was subjected to a surplus loading of over 100% of the daily feed in a brief time period. The result was a sudden increase in outflow of about 300%, which subsequently tapered off to the original flow condition.

The implications for water quality are not inconsequential. In this example, samples taken during the ensuing day represent flows much greater than average. Water has been pushed through the bed, and exits on the order of one day early; and has been somewhat diluted. Velocity increases are great enough to move particulates that would otherwise remain anchored. Internal mixing patterns will blur the effects of the rain on water quality.

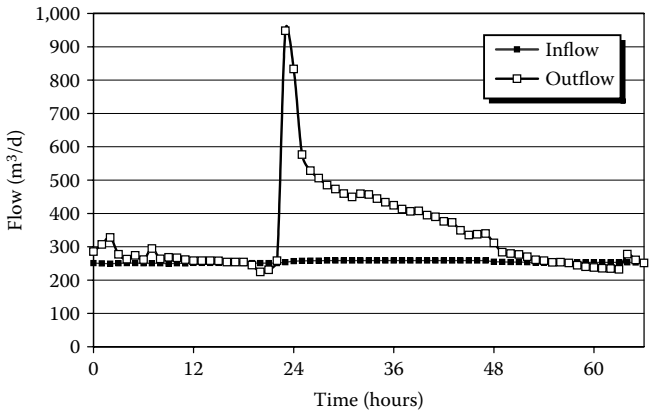



**FIGURE 2.4** Flows into and out of Benton Cell #3 versus time during a rain event period during September 9–11, 1990. Flows were measured automatically via data loggers; the values were stored as hourly averages. The rain event totaled approximately 1.90 cm, or 278 $m^3$. (Data from TVA unpublished data; graph from Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

Sampling intervals are not normally small enough to define these rapid fluctuations. For instance, weekly sampling of Benton Cell #3 would have missed all of the details of the rain event in the illustration above. It is therefore important to realize that compliance samples may give the appearance of having been drawn from a population of large variance, despite the fact that the variability is in large part due to deterministic responses to atmospheric phenomena.

## Evapotranspiration

Water loss to the atmosphere occurs from open or subsurface water surfaces (evaporation), and through emergent plants (transpiration). This water loss is closely tied to wetland water temperature, and is discussed in detail in Chapter 4. Here the impacts of evapotranspiration (*ET*) on the wetland water budget are explored. At this juncture the two simplest estimators will be noted: Large FWS wetland *ET* is roughly equal to lake evaporation, which in turn is roughly equal to 80% of pan evaporation. Table 2.4 shows the distribution of monthly and annual lake evaporation in different regions of the United States.

Wetland treatment systems frequently operate with small hydraulic loading rates. For 100 surface flow wetlands in North America, 1.00 cm/d was the 40th percentile

**TABLE 2.4**
**Lake Evaporation (in mm) at Various Geographic Locations in the United States**

| Location | Jan | Feb | Mar | Apr | May | Jun | Jul | Aug | Sep | Oct | Nov | Dec | Annual |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Yuma, Arizona | 99 | 117 | 165 | 203 | 249 | 292 | 340 | 328 | 272 | 203 | 155 | 114 | 2,540 |
| Sacramento, California | 20 | 36 | 64 | 91 | 127 | 180 | 226 | 218 | 180 | 122 | 66 | 30 | 1,372 |
| Denver, Colorado | 41 | 46 | 64 | 94 | 127 | 188 | 224 | 213 | 170 | 117 | 76 | 48 | 1,397 |
| Miami, Florida | 76 | 86 | 104 | 124 | 127 | 122 | 135 | 130 | 109 | 104 | 109 | 69 | 1,295 |
| Macon, Georgia | 43 | 56 | 79 | 109 | 130 | 157 | 160 | 147 | 132 | 107 | 71 | 46 | 1,245 |
| Eastport, Maine | 20 | 18 | 23 | 28 | 36 | 43 | 51 | 53 | 51 | 41 | 28 | 18 | 406 |
| Minneapolis, Minnesota | 8 | 10 | 23 | 43 | 81 | 112 | 152 | 147 | 117 | 76 | 33 | 10 | 813 |
| Vicksburg, Mississippi | 33 | 48 | 74 | 107 | 127 | 145 | 147 | 140 | 132 | 112 | 74 | 41 | 1,168 |
| Kansas City, Missouri | 23 | 28 | 43 | 79 | 112 | 155 | 203 | 198 | 152 | 114 | 64 | 25 | 1,194 |
| Havre, Montana | 13 | 13 | 28 | 64 | 114 | 155 | 208 | 211 | 142 | 84 | 38 | 18 | 1,092 |
| North Platte, North Dakota | 20 | 28 | 56 | 94 | 127 | 165 | 218 | 213 | 175 | 117 | 66 | 28 | 1,295 |
| Roswell, New Mexico | 53 | 81 | 124 | 173 | 211 | 249 | 239 | 211 | 175 | 140 | 89 | 64 | 1,803 |
| Albany, New York | 15 | 18 | 28 | 51 | 81 | 109 | 132 | 119 | 86 | 61 | 36 | 20 | 762 |
| Bismarck, North Dakota | 10 | 13 | 25 | 58 | 102 | 135 | 185 | 196 | 147 | 84 | 33 | 13 | 991 |
| Columbus, Ohio | 15 | 20 | 28 | 58 | 89 | 117 | 142 | 130 | 104 | 76 | 41 | 15 | 838 |
| Oklahoma City, Oklahoma | 38 | 48 | 79 | 119 | 140 | 198 | 259 | 272 | 224 | 160 | 89 | 51 | 1,676 |
| Baker, Oregon | 13 | 18 | 36 | 64 | 86 | 112 | 175 | 185 | 124 | 74 | 38 | 15 | 940 |
| Columbia, South Carolina | 41 | 61 | 81 | 114 | 137 | 160 | 168 | 152 | 140 | 112 | 76 | 48 | 1,295 |
| Nashville, Tennessee | 23 | 33 | 48 | 84 | 104 | 130 | 147 | 137 | 124 | 94 | 53 | 28 | 991 |
| Galveston, Texas | 23 | 33 | 41 | 66 | 104 | 142 | 157 | 155 | 145 | 117 | 69 | 33 | 1,092 |
| San Antonio, Texas | 56 | 79 | 114 | 142 | 165 | 213 | 239 | 239 | 193 | 147 | 94 | 61 | 1,753 |
| Salt Lake City, Utah | 20 | 25 | 51 | 89 | 130 | 201 | 269 | 264 | 185 | 99 | 51 | 25 | 1,397 |
| Richmond, Virginia | 33 | 43 | 56 | 89 | 104 | 127 | 142 | 124 | 104 | 81 | 61 | 38 | 991 |
| Seattle, Washington | 20 | 20 | 36 | 53 | 69 | 86 | 99 | 86 | 66 | 41 | 28 | 18 | 610 |
| Milwaukee, Wisconsin | 15 | 18 | 23 | 33 | 53 | 81 | 127 | 137 | 119 | 81 | 41 | 15 | 737 |

*Source:* From van der Leeden *et al.* (1990) *The Water Encyclopedia.* Second Edition, Lewis Publishers, Boca Raton, Florida.

in the early days of constructed wetland technology (NADB database, 1993). *ET* losses approach a daily average of 0.50 cm/d in summer in the southern United States; consequently, more than half the daily added water may be lost to *ET* under those circumstances. But *ET* follows a diurnal cycle, with a maximum during early afternoon, and a minimum in the late nighttime hours. Therefore, outflow can cease during the day during periods of high *ET*.

As a second example, Platzer and Netter (1992) report that the nominal detention time, based on inflow, for the subsurface flow wetland at See, Germany, was 20 days. There was a measured net loss of 70% of the water to evapotranspiration in summer. The actual nominal detention time, computed from Equation 2.8, is 34.4 days; the use of an average flow rate gives 30.8 days.

In addition to the consumptive use of water, which may be critical in water-poor regions, *ET* acts to concentrate contaminants remaining in the water. For instance, Platzer and Netter (1992) report that the wetland accomplished 88% ammonia removal on a mass basis. When coupled with the 70% water loss, the ammonia concentration reduction is only 60%.

In mild temperate climates, annual rainfall typically slightly exceeds annual *ET*, and there is little effect of atmospheric gains and losses over the course of a year. But most climatic regions have a dry season and a wet season, which vary depending upon geographical setting. As a consequence evapotranspiration losses may have a seasonally variable impact. For example, *ET* losses are important in northern systems that are operated seasonally. In northern North America, about 80% of the annual *ET* loss occurs in the six months of summer. Therefore, lightly loaded seasonal wetlands in cold, arid climates are strongly influenced by net atmospheric water loss. Examples include the Williams Pipeline HSSF system in Watertown, South Dakota (Wallace, 2001), which operates at zero discharge during the summer, the Roblin, Manitoba, FWS system, which operates at zero discharge two summers out of every three; and the Saginaw, Michigan, FWS system, which operates with 50% water loss (Kadlec, 2003c).

### *Dynamic ET Response*

The diurnal cycle in *ET* can be reflected in water levels and flow rates under light loading conditions. HSSF Cell #3 at Benton, Kentucky, was operated in September 1990 at a hydraulic loading rate (HLR) of 1.7 cm/d, corresponding to a nominal detention time (HRT) of approximately 13 days. Evapotranspiration at this location and at this time of year was estimated to be about 0.5 cm/d. Consequently, *ET* forms a significant fraction of the hydraulic loading. Because *ET* is driven by solar radiation, it occurs on a diurnal cycle. The anticipated effect is a diurnal variation in the outflow from the bed, with amplitude mimicking the amplitude of the combined (feed plus *ET*) loading cycle. This was measured at Benton (Figure 2.5).

In such an instance, because the night outflow peak is nearly double the daytime minimum outflow, it would be desirable to use diurnal timed samples of the outflow, and to appropriately flow-weight them, for determination of water quality.

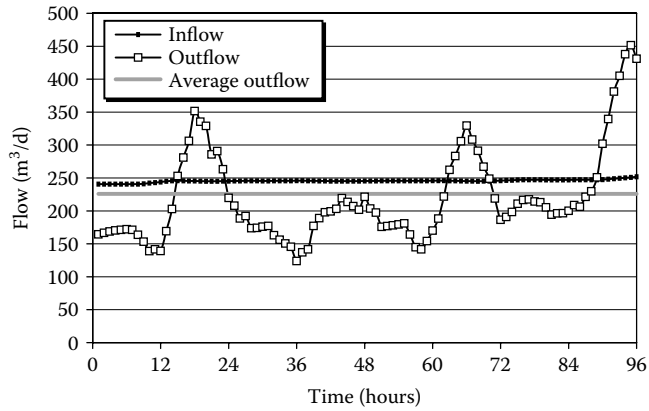



**FIGURE 2.5** Flows into and out of Benton Cell #3 versus time during September 5–8, 1990. Flows were measured automatically via data loggers; the values were stored as hourly averages. The data points on this graph are six-hour running averages, which smooth out short-term "noise" and emphasize the diel trends. (Data from TVA unpublished data; graph from Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

## Seepage Losses and Gains

### *Bank Losses*

Shallow seepage, or bank loss, occurs if there is hydrologic communication between the wetland and adjacent aquifers. This is a nearly horizontal flow (see Figure 2.3). If impermeable embankments or liners have been used, bank losses will be negligibly low. However, there are situations where this is not the case, notably for large wetlands treating nontoxic contaminants. An empirical procedure may then be used in which the bank loss is calibrated to the head difference between the water inside and outside of the berm (Guardo, 1999). A linear version of such a model is:

$$Q_b = \lambda L_b (H - H_s) \tag{2.15}$$

where

$Q_b$ = bank seepage flow rate, $m^3/d$
$H$ = wetland water elevation, m
$H_s$ = external water elevation, m
$L_b$ = length of the berm, m
$\lambda$ = empirical coefficient, m/d

For instance, wetland levees in southern Florida are typically built from the peat and limestone soils native to the area. Leakage is therefore significant, and has been studied extensively in connection with many canal, storage, and treatment projects. The value used is $\lambda = 15$ m/d (Burns and McDonnell, 1992), which represents a very leaky berm.

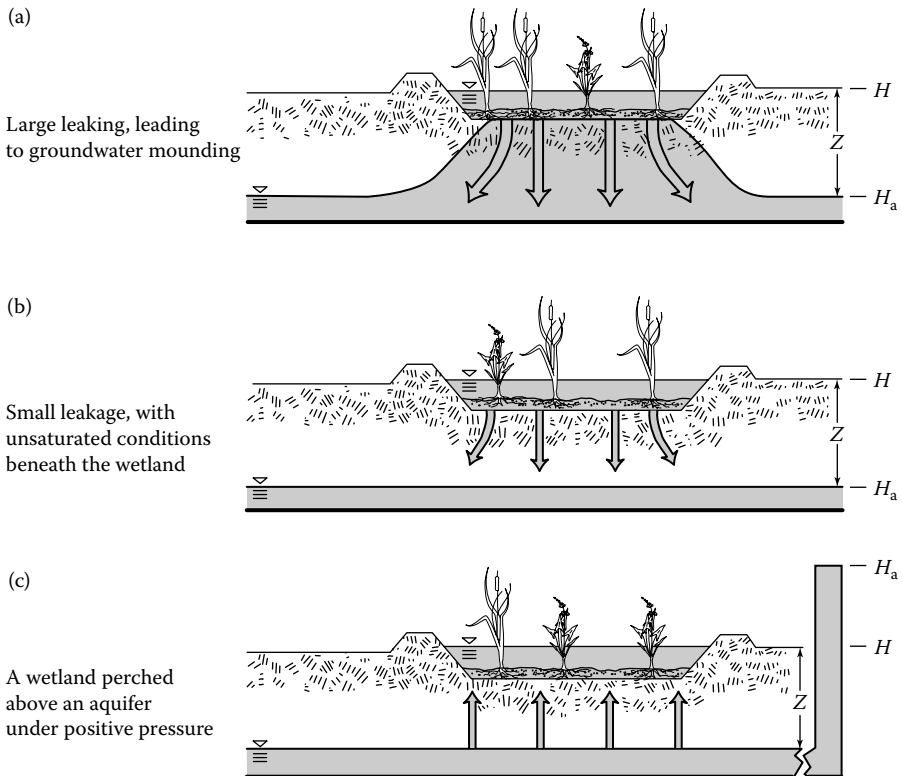



**FIGURE 2.6** Three potential groundwater–wetland interactions. (a) Large leakage, leading to groundwater mounding; (b) small leakage, with unsaturated conditions beneath the wetland; (c) a wetland perched above an aquifer under positive pressure. $H$ = stage in the wetland, $H_a$ = piezometric surface in aquifer, and $Z$ = distance from wetland surface to piezometric surface. (Adapted from Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

### *Infiltration*

Deep seepage, or infiltration occurs by vertical flow. Unless there is an impermeable barrier, wetland waters may pass downward to the regional piezometric surface (Figure 2.6). The soils under a treatment wetland may range in water condition from fully saturated, forming a water mound on the shallow regional aquifer, to unsaturated flow (trickling).

If the wetland is lined with a relatively impervious layer, it is likely that the underlying strata will be partially dry, with the regional shallow aquifer located some distance below (Figure 2.6b). In this case, it is common practice to estimate leakage from the wetland from:

$$Q_{gw} = k\,A\left[\frac{H_w - H_{lb}}{H_{lt} - H_{lb}}\right] \tag{2.16}$$

where

$A$ = wetland area, $m^2$
$H_{lb}$ = elevation of the liner bottom, m
$H_{lt}$ = elevation of the liner top, m
$H_w$ = wetland water surface elevation, m
$k$ = hydraulic conductivity of the liner, m/d
$Q_{gw}$ = infiltration rate, $m^3/d$

The city of Columbia, Missouri, FWS wetlands provide an example of this situation. It was planned to discharge secondary wastewater to 37 ha of constructed wetlands rather than directly to the Missouri River (Brunner and Kadlec, 1993). Those wetlands were sealed with 30 cm of clay, but were situated on rather permeable soils. The hydraulic conductivity of the clay sealant was $1 \times 10^{-7}$ cm/s. Water was to be 30 cm deep, and there was 30 cm of topsoil above the clay as a rooting media for wetland plants. Equation 2.17 may be used to estimate a leakage of approximately 0.79 cm/month. Because of the proximity of Columbia's drinking water supply wells, this leakage rate was experimentally confirmed prior to startup. Over a 27-day period, wetland unit one lost 0.21 cm more than the control, indicating a tighter seal than designed.

If there is enough leakage to create a saturated zone under the wetland (Figure 2.6a), then complex three-dimensional flow calculations must be made to ascertain the flow through the wetland bottom to groundwater. These require a substantial quantity of data on the regional water table, regional groundwater flows, and soil hydraulic conductivities by layer. Such calculations are expensive, and usually warranted only when the amount of seepage is vital to the design.

A third possibility is that the wetland is perched on top of, and is isolated from, the shallow regional aquifer. In some instances, such as the Houghton Lake site, the wetland may be located in a clay "dish," which forms an aquiclude for a regional shallow aquifer under pressure (Figure 2.6c). A well drilled through the wetland to the aquifer displays artesian



**FIGURE 2.7** Water barrel apparatus to test liner leakage in a VF wetland, Diamond Lake Woods, Minnesota.

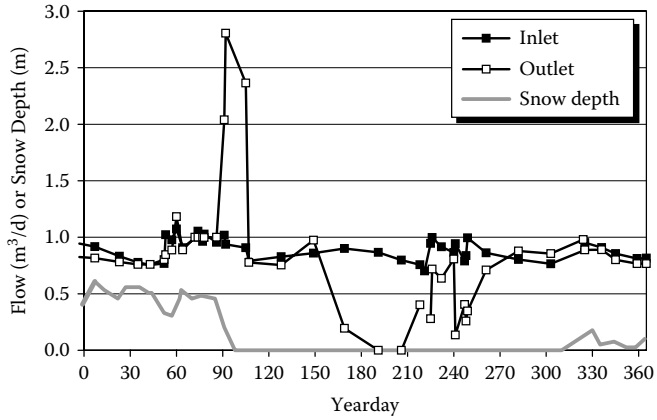



**FIGURE 2.9** Flows into and out of NERCC wetland #2 in 1997. The large spike in outflow corresponds to a sudden snowmelt at the end of March. Evapotranspiration losses are apparent in summer. (From Kadlec (2001b) *Water Science and Technology* 44(11/12): 251–258. Reprinted with permission.)

character. The "in-leak" for this system is very small, because the clay layer is many feet thick (Haag, 1979).

In practice, a leak test is often required to demonstrate that a liner in fact performs as designed. One such procedure is known as the Minnesota barrel test (Minnesota Pollution Control Agency, 1989). The water loss from a bottomless barrel placed in the wetland is compared to the water loss from a barrel with a bottom. The barrels collect rain and evaporate water with equal efficiency, so any additional loss from the bottomless barrel must be due to infiltration (Figures 2.7 and 2.8).

Infiltration is allowable in instances where there is not a perceived threat to groundwater quality necessary for the indicated use. That may be drinking water quality, in which case a liner would be used. But the underlying aquifer may have lesser water quality requirements. Such is the case for the Incline Village, Nevada, FWS wetlands, which are underlain by waters with very high concentrations of dissolved evaporites, mostly sulfates. That aquifer is not useable for potable water, and as a consequence, the wetlands were designed to allow infiltration (no liner) (Kadlec *et al.*, 1990).

In other situations, the affected groundwater is known to discharge into other water bodies that either provide dilution or further treatment. The former case is typified by the Sacramento wetlands, which leaked about 40% of the added water (Nolte and Associates, 1997). The leakage was known to join a large river, which minimized risks to acceptable levels.

## Snowmelt

In northern climates, snowmelt is a springtime component of the liquid water mass balance. The end-of-season snow pack is melted over time, in rough proportion to the temperature excess above freezing. The amount of the snowpack is documented in weather records, such as Climatological Data (NOAA). An example of the effect on flow rate is shown in Figure 2.9, for a HSSF treatment wetland at the NERCC site near Duluth, Minnesota (latitude 46.8°N). The snow depth was about 50 cm in winter, providing insulation enough to prevent freezing of the HSSF wetland bed. A rapid spring

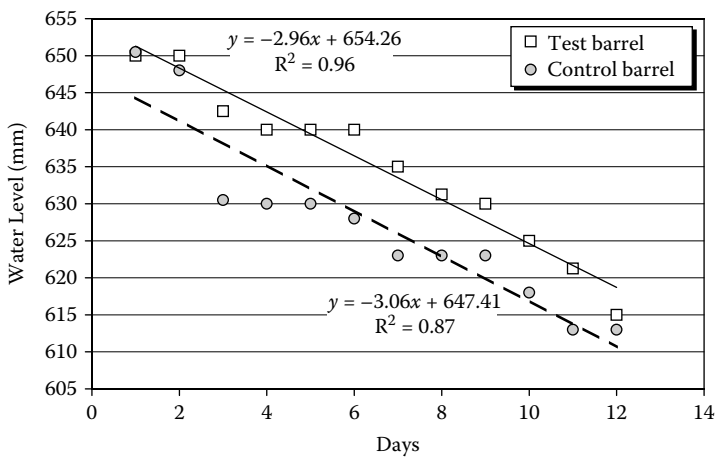



**FIGURE 2.8** Results of VF wetland liner testing using the Minnesota barrel method.

thaw created a large spike of melt water that added to the pumped inflow.

## Water Storage

The computation of the volume of water stored in a FWS wetland involves the stage-storage curve for the wetland. The derivative of this function is the water surface area:

$$A = \frac{dV}{dh} \tag{2.17}$$

where

$A$ = wetland area, $m^2$
$h$ = wetland depth, m
$V$ = wetland water volume, $m^3$

In normal practice, no allowance is made for the volume occupied by vegetation, because of the difficulty of measurement of the vegetation volume. Some wetlands have steeply pitched side slopes, and may be regarded as constant area systems. This implies that the stage-storage curve is a straight line. For instance, Mierau and Trimble (1988) report a nearly linear stage-storage curve for a rectangular diked marsh treating river water. But some wetlands have more complicated topography, such as the treatment wetlands at Des Plaines (Figure 2.10).

This information permits computation of water elevation changes from a knowledge of changes in storage volume. Over any time period, the stage change ($\Delta H$) is given by:

$$\Delta H = \int_{t_1}^{t_2} \frac{dV}{A} = \frac{\Delta V}{A_{avg}} \tag{2.18}$$

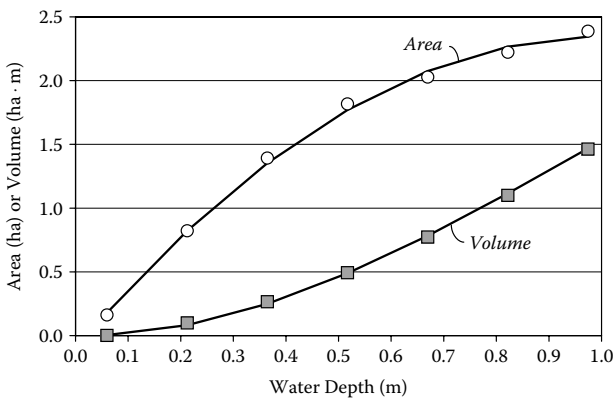



**FIGURE 2.10** Stage-storage and stage-area curves for wetland EW3 at Des Plaines, Illinois. The curves are predicted by the following equations. (From Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

$$V = 0.0022 - 0.104h + 2.44h^2 - 0.809h^3$$

$$A = \frac{dV}{dh} = -0.104 + 4.88h - 2.43h^2$$

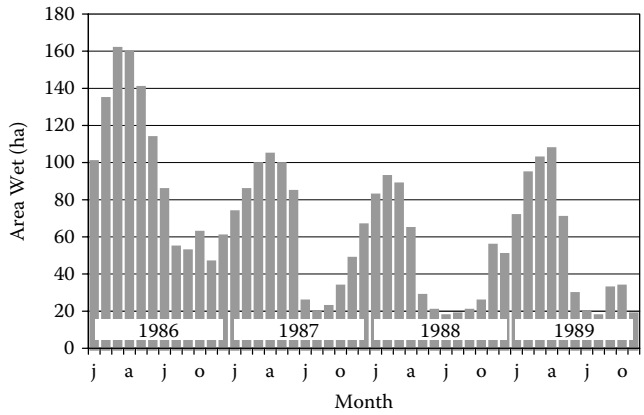



**FIGURE 2.11** The expansion and shrinkage of the Incline Village wastewater wetlands as a function of time. Summer water diversions to agricultural uses accelerate the dryout caused by arid conditions. (Data from Kadlec *et al*. (1990) In *Constructed Wetlands in Water Pollution Control*. Cooper and Findlater (Eds.), Pergamon Press, Oxford, United Kingdom, pp. 127–138; graph from Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press Boca Raton, Florida.)

where

$\Delta H$ = change in wetland water surface elevation, m
$\Delta V$ = change in wetland volume, $m^3$
$A_{avg}$ = mean water surface area over the time period from $t_1$ to $t_2$

In the extreme, a wetland may evaporate much of the added water, such as at Incline Village, Nevada. The area of these wetlands responds by expanding and shrinking in response to added water and evapotranspiration (Figure 2.11).

Stormwater treatment wetlands pose a less extreme but important problem: Given fluctuating water levels and wetted areas, what area or volume should be used in pollutant removal calculations? Although this is a complicated question, a bound may be placed on the effective area. If some of the wetland area is dry some of the time, it cannot participate in removals. For a given time period, the number of wetted hectare·days are cumulated, and divided by the total possible wet hectare·days for the entire system footprint to produce the *treatment opportunity fraction*, $\phi$ (Brown and Caldwell, 1996):

$$\phi = \frac{1}{(t_2 - t_1)A} \int_{t_1}^{t_2} A_{wet}\, dt \tag{2.19}$$

where

$A$ = total wetland area, $m^2$
$A_{wet}$ = wetland wetted area at time $t$, $m^2$
$t_1$ = start of time period, d
$t_2$ = end of time period, d
$\phi$ = treatment opportunity fraction, dimensionless

Event-driven wetlands are discussed in more detail in Chapter 14.

## Combined Effects: The Wetland Water Budget

Equation 2.13, the wetland water balance, states that the change in storage in the wetland results from the difference between inflows and outflows. In theory, any one term may be calculated from Equation 2.13 if all the other terms are known. But in practice, none of the measurements are very precise, and large errors may result for such a calculation (Winter, 1981).

Examples of monthly variability of the wetland water budget are given in Table 2.5, for a periphyton pilot wetland (PSTA Test Cell 8) (CH2M Hill, 2001b) and for a large treatment marsh (Boney Marsh) (Mierau and Trimble, 1988). Importantly, the monthly error in closure of the monthly water budget for Boney Marsh ranged from –18% to +7%, with a root mean square (RMS) error of 9% (one outlier removed). These percentages are based upon the combined water inflow. For PSTA Test Cell 8, errors ranged from –30%

**TABLE 2.5**
**Example Water Budgets for FWS Wetlands**

**Periphyton Test Cell 8**
**Area: 0.25 ha**
**Year: 1999**
**Lined Wetland Cell**

| Month | Inflow ($m^3$) | Outflow ($m^3$) | ET ($m^3$) | Rain ($m^3$) | ΔStorage ($m^3$) | Infiltration ($m^3$) | Residual ($m^3$) | Residual (% of Inflow) |
|---|---|---|---|---|---|---|---|---|
| January | 3,413 | 4,328 | 247 | 75 | –797 | 0 | –291 | –8% |
| February | 3,378 | 4,444 | 272 | 45 | –261 | 0 | –1,031 | –30% |
| March | 3,818 | 4,634 | 339 | 267 | –118 | 0 | –770 | –19% |
| April | 3,803 | 4,336 | 340 | 48 | 121 | 0 | –946 | –25% |
| May | 3,802 | 3,634 | 356 | 14 | 8 | 0 | –59 | –2% |
| June | 3,805 | 4,421 | 276 | 837 | 16 | 0 | –71 | –2% |
| July | 3,807 | 4,414 | 358 | 212 | –24 | 0 | –728 | –18% |
| August | 3,809 | 3,615 | 317 | 628 | 81 | 0 | 425 | 10% |
| September | 3,809 | 5,005 | 281 | 453 | 49 | 0 | –1,074 | –25% |
| October | 3,716 | 4,147 | 257 | 932 | –57 | 0 | 301 | 6% |
| November | 3,889 | 4,418 | 222 | 29 | –63 | 0 | –659 | –17% |
| December | 3,841 | 3,065 | 185 | 100 | 36 | 0 | 635 | 16% |
| **Average** | **3,741** | **4,205** | **287** | **303** | **–84** | **0** | **–356** | **–9%** |
| **RMS Residual** | | | | | | | | **17.3%** |

**Boney Marsh**
**Area: 49 ha**
**Year: 1983**
**Unlined Wetland Cell**

| | Inflow (1,000 $m^3$) | Outflow (1,000 $m^3$) | ET (1,000 $m^3$) | Rain (1,000 $m^3$) | ΔStorage (1,000 $m^3$) | Seepage (1,000 $m^3$) | Residual (1,000 $m^3$) | % Error |
|---|---|---|---|---|---|---|---|---|
| January | 335 | 395 | 30 | 27 | –37 | 3 | –28 | –7.8% |
| February | 313 | 362 | 37 | 92 | 6 | 3 | –2 | –0.5% |
| March | 340 | 418 | 55 | 59 | –4 | 3 | –73 | –18.2% |
| April | 322 | 392 | 62 | 22 | –65 | 3 | –48 | –13.8% |
| May | 66 | 88 | 84 | 10 | –27 | 3 | –72 | –95.1% |
| June | 239 | 199 | 61 | 136 | 110 | 3 | 2 | 0.6% |
| July | 321 | 281 | 67 | 43 | –12 | 3 | 25 | 6.9% |
| August | 354 | 277 | 50 | 45 | 74 | 3 | –6 | –1.4% |
| September | 384 | 259 | 43 | 47 | 108 | 3 | 18 | 4.2% |
| October | 356 | 411 | 43 | 18 | –18 | 3 | –66 | –17.6% |
| November | 303 | 403 | 33 | 16 | –115 | 3 | –3 | –1.0% |
| December | 374 | 399 | 27 | 43 | –1 | 3 | –11 | –2.7% |
| **Average** | **309** | **324** | **49** | **47** | **1** | **3** | **–22** | **–7.1%** |
| **RMS Residual** | | | | | | | | **9.0%** |

to +16%, with a root mean square error of 17%. The RMS error increases with decreasing water budget period. For Boney Marsh, over an eight-year period, the daily, monthly, and annual RMS errors were 67%, 16%, and 7%, respectively (Mierau and Trimble, 1988).

These are not extreme examples. Similar lack of closure has been reported for four wetlands at Sacramento, where all mass balance terms were measured independently, including infiltration measured by drawdown (Nolte and Associates, 1998b). The RMS monthly errors were 60%, 47%, 26%, and 19% for Cells 3, 5, 7, and 9, respectively. The annual percentage residuals were −56%, −44%, −15%, and −15%, respectively. The conclusion was that these apparent water losses were due to faulty inflow or outflow measurements.

These examples serve to alert the wetland designer or operator that care must be taken in water flow measurements and that water balance differencing is apt to provide estimates with large uncertainty. With great care, balance closure may be held to the ±5 to 10% range (Mierau and Trimble, 1988; Guardo, 1999; Martinez and Wise, 2001).

## 2.2 FWS WETLAND HYDRAULICS

Early in the history of research and development related to overland flow in wetlands, mathematical descriptions were often adaptations of turbulent open channel flow formulae. These are discussed in detail in a number of texts—for example, the work of French (1985). The general approach is utilization of mass, energy, and momentum conservation equations, coupled with an equation for frictional resistance. Perhaps the most common friction equation is Manning's equation, which will be further discussed later in this section.

There is a fundamental problem with the utilization of Manning's equation to wetland surface water flows: Manning's equation is a correlation for turbulent flows, whereas FWS wetlands are nearly always in a laminar or transitional flow regime (based on open channel flow criteria). Under these conditions, Manning's $n$ is not constant, but is strongly velocity dependent (Hosokawa and Horie, 1992). There is also a difficulty with the extension of open channel flow concepts to densely vegetated channels. The frictional effects that retard flow in open channels are associated primarily with drag exerted by the channel bottom and sides. Wetland friction in dense macrophyte stands is dominated by drag exerted by the stems and litter, with bottom drag playing a very minor role.

As a consequence, overland flow parameters determined from open channel theory are not applicable to wetlands. In particular, Manning's coefficient is no longer a constant; it depends upon velocity and depth as well as stem density. Predictions from previous information on nonwetland vegetated channels are seriously in error (Hall and Freeman, 1994). Unfortunately, much of the existing information on wetland surface flow has been interpreted and reported via Manning's equation, and so it cannot be avoided.

Major advances in formulating correct and improved approaches to overland flow in wetlands have been made in the past ten years (e.g., Nepf, 1999; Oldham and Sturman, 2001; Choi *et al.*, 2003). This section utilizes the emerging knowledge and calibration database to provide methods to predict depths and velocities in FWS wetlands.

### The Calculation Structure

Wetland water depths and flow rates are controlled by two major wetland features; the outlet structure and resistance to flow within the wetland. In general, it is very desirable to have control at the outlet structure, because then the operator has control over water depth. Under complete outlet control, a level pool of water exists upstream of the outlet structure, regardless of what is growing there. However, that is not always possible, particularly for large or densely vegetated wetlands. Water may be held up by the vegetation at a depth that is independent of the outlet structure setting.

Four different situations may occur, and are easily visualized (Figure 2.12):

1. Very low flow; complete weir control. There is a level pool upstream of the outlet structure, and wetland water stage is spatially invariant.
2. Partial weir control (*M1 profile*). There is a level pool in the region near the weir, but a gradient in stage near the wetland inlet. This is a distance thickening sheet flow.
3. Normal depth flow. Vegetation drag controls the depth to exactly the stage created by the weir.
4. Large flow; partial weir control (*M2 profile*). There is a constant depth flow, at the normal depth, near the inflow, followed by decreasing depth near the outflow. This is a distance thinning sheet flow.

These various possibilities are covered by a backwater calculation. Because wetlands nearly always meet the criterion for gradually varied flow with a small Froude number (French, 1985), the water flow momentum balance can be

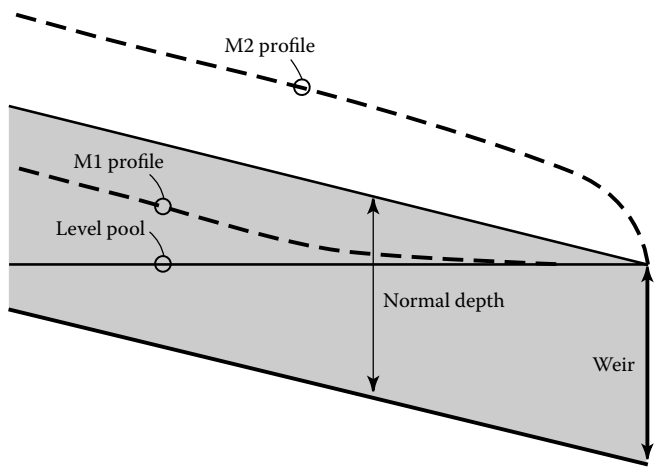



**FIGURE 2.12** FWS water surface profiles for a fixed height over the outlet weir and various inlet flows. The notation follows French (1985).

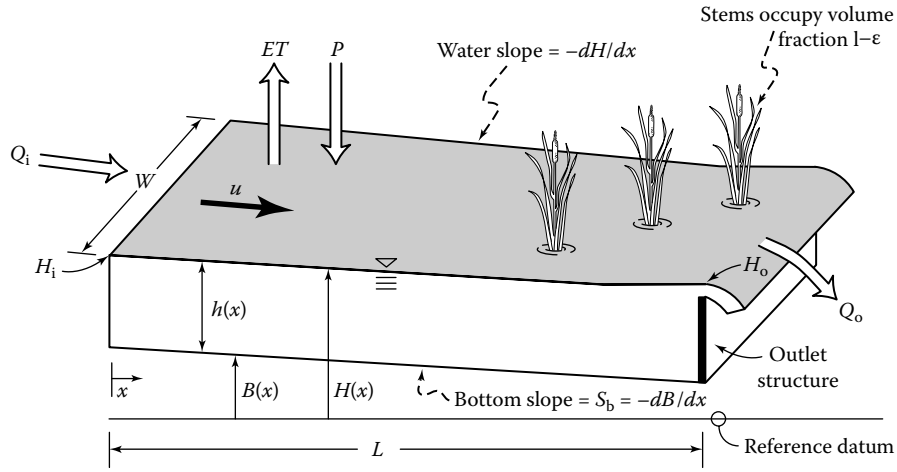



**FIGURE 2.13** Notation for FWS bed hydraulics. (From Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

simplified to contain only gravitational and friction terms. The component pieces are the spatial water mass balance, the friction equation, and specification of inflow, geometry, and outlet depth setting. For one-dimensional (rectangular) systems, *in the absence of rain or ET effects*, the flow situation can be simplified as indicated in Equation 2.20. Notation is given in Figure 2.13.

The spatial water mass balance, water depth ($h$), and superficial flow velocity ($u$) are distance-variable:

$$\frac{dQ}{dx} = \frac{d(hWu)}{dx} = 0 \tag{2.20}$$

where

$h$ = water depth, m
$Q$ = volumetric flow rate, $m^3/d$
$W$ = wetland width, m
$u$ = superficial flow velocity, m/d
$x$ = distance in flow direction, m

Frictional losses can be represented by a general power law relationship. This is discussed in further detail in the next section of this chapter:

$$u = a\, h^{(b-1)} S^c \tag{2.21}$$

where

$a, b, c$ = friction parameters
$u$ = superficial water velocity, m/s
$h$ = water depth, m
$S = -dH/dx$ = negative of the water surface slope, m/m

Water elevation is water depth plus bed bottom elevation profile (Figure 2.13):

$$H = B + h \tag{2.22}$$

where

$B$ = elevation of the bed bottom above datum, m
$H$ = elevation of the water surface, m
$h$ = water depth, m

Equations 2.20, 2.21, and 2.22 combine to give:

$$aWh^b \left( -\frac{d(h+B)}{dx} \right)^c = Q \tag{2.23}$$

The boundary condition necessary to solve Equation 2.23 is typically a specification of the outlet water level, as determined by a weir or receiving pool:

$$H_{at\ x=L} = H_o \tag{2.24}$$

where

$H$ = elevation of the water surface, m
$x$ = distance in flow direction, m
$L$ = length of wetland cell along flow path, m

Equations 2.23 and 2.24 cannot be solved analytically to a closed-form answer, but numerical solution is easy via any one of a number of methods. Required input parameters are the bottom slope profile, the flow rate and the height over the weir, together with the friction parameters $a$, $b$, $c$.

Although there can be any of several types of outflow structure, it is useful to illustrate the determination of the weir overflow stage for that choice of outlet control. The commonly used equation for a rectangular weir is:

$$Q_o = C_E\, W_W\, (H_o - H_W)^{1.5} \tag{2.25}$$

where

$Q_o$ = outlet flow rate, $m^3/d$
$C_E$ = weir discharge coefficient, $(m^3/d)/(m^{2.5})$
$W_W$ = width of weir, m
$H_o$ = water surface elevation at wetland outlet, m
$H_W$ = weir crest elevation, m

## Friction Equations for FWS Wetland Flows

All of the required information for the backwater calculation is readily obtainable, except for the friction parameters $a$, $b$, and $c$. Water flow through the wetland is associated with a local frictional head loss, given by Equation 2.21. This is a power law representation of the fact that the water velocity is related to the water surface slope ($S = -dH/dx$) and to the depth of the water ($h$). This generalized form of Equation 2.21 was first suggested by Horton (1938). He proposed $b$ equal to zero for vegetated flow, 2.0 for laminar flow, and 4/3 for turbulent flow; and $c$ equal to 1.0 for laminar or vegetated flow and 2.0 for turbulent flow; and $a = 1/K$ is a constant (different for the three cases). Transition flows were to be handled by adjusting the value of $b$ between 1.0 and 2.0. We use this form here, although for reasons different from Horton (1938), as will be explained below.

The friction equation is a vertically averaged result, based upon a reluctance to go to the complexity of three-dimensional computational fluid mechanics. This results in two difficulties in the wetland environment:

1. There is a vertical profile of vegetation resistance in many cases, because the submerged plant parts are often stratified.
2. A good deal of the literature presumes flows in evenly flat-bottomed systems, which is not the case for wetlands. It is usual to have a significant amount of microtopographical relief in the wetland, which also factors into vertical averaging.

### Flows Controlled by Bottom Friction

The framework that is very often borrowed from the literature is adaptation of constant depth, open channel flow equations. It is to be noted that *this situation should not apply to vegetated wetlands*, but that has not prevented widespread use of the equations.

When $a = 1/K$, $b = 3$, and $c = 1$, Equation 2.21 becomes the equation for laminar flow in an open channel as shown in Equation 2.26 (Straub *et al.*, 1958):

$$u = \frac{1}{K} h^2 S \tag{2.26}$$

where

$K$ = laminar flow friction coefficient, s·m
$u$ = superficial flow velocity, m/s
$h$ = water depth, m
$S = -dH/dx$ = negative of the water surface slope, m/m

Note that a unit conversion is necessary to convert to the mass balance unit of days. The limit of this formulation for a channel devoid of vegetation is the *depth Reynold's number* (Re) less than 2,500:

$$\mathrm{Re} = \frac{h \rho u}{\mu} < 2{,}500 \tag{2.27}$$

where

Re = depth Reynold's number, dimensionless
$h$ = water depth, m
$u$ = superficial water velocity, m/s
$\mu$ = viscosity of water, kg/m·s
$\rho$ = density of water, kg/m$^3$

For average warm water properties and a typical water depth of 30 cm, a Reynold's number of less than 2,500 translates to flow velocities less than about 700 m/d, a range that includes most FWS wetlands, except for the very largest.

When $a = 1/n$, $b = 5/3$, and $c = 1/2$, Equation 2.21 becomes Manning's equation (French, 1985):

$$u = \frac{1}{n} h^{2/3} S^{1/2} \tag{2.28}$$

where

$n$ = Manning's coefficient, s/m$^{1/3}$
$u$ = superficial water velocity, m/s
$h$ = water depth, m
$S = -dH/dx$ = negative of the water surface slope, m/m

Note that a unit conversion is again necessary to convert to the mass balance unit of days.

Suppose that open channel information were to be used to estimate Manning's $n$ for a wetland. Guidance may be found in estimation procedures in the hydraulics literature, for instance French (1985). The value of $n$ may be estimated from information on the channel character, type of vegetation, changes in cross section, surface irregularity, obstructions, and channel alignment. Using the highest value of every contributing factor, the maximum open channel $n$ value is 0.29 s/m$^{1/3}$ (French, 1985). This is approximately one order of magnitude less than values determined from actual wetland data. *Clearly, open channel, turbulent flow information is inadequate to describe the densely vegetated, low-flow wetland environment.*

Nepf (1999) used both laboratory flumes and field measurements in a Spartina marsh to conclude that bed drag is negligible compared to stem and leaf drag at densities of submerged vegetation of one percent by volume and higher. *Therefore, Equations 2.26 and 2.28 are both inappropriate for vegetated wetlands.*

### Flows Controlled by Stem Drag

The presence of submerged stems, leaves, and litter creates an underwater environment dominated by drag on those surfaces, rather than the channel bottom. The common measures of vegetation density are the number per square meter times their diameter:

$$a = n_s d \tag{2.29}$$

$$ad = n_s d^2 \tag{2.30}$$

where
$a$ = projected plant area normal to flow per unit volume, $m^2/m^3$
$d$ = cylinder diameter of vegetation, m
$ad$ = fraction of volume occupied by plants, $m^3/m^3$
$n_s$ = number of stems per unit area, $\#/m^2$

The traditional measure of vegetative surface area is the leaf area index (LAI). In the context of immersed surfaces and drag, it is the fraction of the total LAI below water and its vertical distribution that are of interest. Although LAI and area normal to flow are not identical, a direct relation between them would be expected.

The resistance to flow through this submersed matrix is described by a drag equation (Nepf and Koch, 1999):

$$S = C_D a\left(\frac{u^2}{2g}\right) \tag{2.31}$$

where
$C_D$ = drag coefficient, dimensionless
$S = -dH/dx$ = negative of the water surface slope, m/m
$a$ = projected plant area normal to flow per unit volume, $m^2/m^3$
$u$ = superficial water velocity, m/s
$g$ = acceleration of gravity, $m/s^2$

If the *stem Reynolds number* ($Re_s$) within the array is less than 200, the flow will be laminar:

$$Re_s = \frac{d\rho u}{\mu} < 200 \tag{2.32}$$

where
$Re_s$ = stem Reynold's number, dimensionless
$d$ = cylinder diameter of vegetation, m
$u$ = superficial flow velocity, m/s
$\mu$ = water viscosity, kg/m·s
$\rho$ = water density, $kg/m^3$

As a point of reference, stems of one cm diameter in a flow of 1,000 m/d would produce an $Re_s = 116$, which is still within the laminar flow range. For flow velocities typically encountered in FWS wetlands, this implies that flows proceed with interfering laminar wakes (Nepf, 1999). Stem densities are such that drag is determined by obstruction of flow (form drag). For this circumstance,

$$C_D = \frac{K_1}{Re_s} \tag{2.33}$$

where
$C_D$ = drag coefficient, dimensionless
$K_1$ = constant, unitless
$Re_s$ = stem Reynold's number, dimensionless

Under these circumstances, it may be shown that yet another set of parameters might be applicable in Equation 2.21, i.e., $b = 1$ and $c = 1$:

$$u = \frac{K_{stem}}{n_s} S \tag{2.34}$$

where
$u$ = superficial flow velocity, m/s
$n_s$ = number of stems per unit area, $\#/m^2$
$K_{stem}$ = conveyance coefficient, $m^{-1}\cdot s^{-1}$
$S = -dH/dx$ = negative of the water surface slope, m/m

Note that a unit conversion is again necessary to convert to the mass balance unit of days. There is no depth effect in this formulation, which is, in effect, Darcy's law for uniform porous media, where the porous media in this case is a bed of submerged vegetation. Data from channels with vertical rods indeed support this analysis (Nepf, 1999; Schmid *et al.*, 2004b). Hall and Freeman (1994) confirmed the direct proportionality of resistance to stem density for bulrushes, which have a plant geometry very similar to vertical rods.

There are, however, several other important features of wetland flows that must be taken into account. There are vertical and spatial profiles of stem-leaf density, wind forces can move water (Jenter and Duff, 1999), and the wetland bottom is not flat (Kadlec, 1990).

## Vertical Profiles of Stem Density

The vertical location of plant stems and leaves varies with the type of vegetation under consideration. One limiting case is floating plants, such as water hyacinths (*Eichhornia crassipes*), which populate only the topmost stratum of the water column. Rooted plants with floating leaves, such as water lilies (*Nymphaea* spp.), also place most drag in the vicinity of the water surface, with a lesser amount in the water column due to stems. In contrast, most of the commonly used emergent macrophytes in treatment wetlands have stems and/or leaves distributed throughout the water column, but the distributions are not necessarily uniform. A bottom layer normally contains dead and prostrate plant parts, which is the litter layer. Stems or culms are dominant portion of these lower horizons. Bulrushes continue with stem morphology exclusively, but leaves are dominant at mid-depths for cattails, sedges, and reeds. In combination, the distributions of drag surfaces, for many emergent marsh systems, are fairly uniform over typical operating depth ranges (Figure 2.14), as indicated by the linearity of the cumulative LAI with depth. Thus, in the absence of any other factors, flow would be expected to follow a stem/leaf drag relationship such as Equation 2.34.

## The Influence of Bathymetric Variability

The bottom elevation of many FWS wetlands is irregular, with local depressions and hummocks. On a large scale, these are

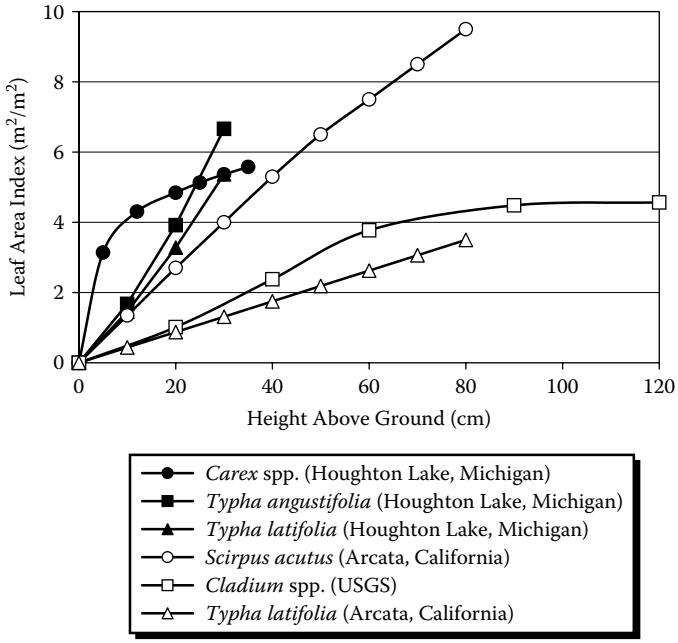



**FIGURE 2.14** Leaf area indices for various emergent macrophytes. These are cumulative numbers, representing the total leaf area below a given elevation above ground. (Data for USGS: Rybicki *et al.* (2000) *Sawgrass density, biomass, and leaf area index a flume study in support of research on wind sheltering effects in the Florida Everglades.* Open File Report 00-172, U.S. Geological Survey: Reston, Virginia; for Arcata: U.S. EPA (1999) *Free water surface wetlands for wastewater treatment: A technology assessment*. EPA 832/R-99/002, U.S. EPA Office of Water: Washington, D.C. 165 pp.; for Houghton Lake: unpublished data; and Kadlec (1990) *Journal of the Hydraulics Division (ASCE)* 116(5): 691–706.) Corresponding porosities were:

| | | |
|---|---|---|
| *Carex* spp. | Houghton Lake | 99% |
| *Typha angustifolia* | Houghton Lake | 96% |
| *Typha latifolia* | Houghton Lake | 96% |
| *Scirpus acutus* | Arcata | 97% |
| *Cladium* spp. | USGS | 98% |
| *Typha latifolia* | Arcata | 93% |

quantified by depth–area–volume relations (see Figure 2.10). On a small scale, these features define the micro-topography of the wetland bottom, and are represented by a soil surface elevation distribution. Small constructed wetlands are typically designed to be graded at a specified tolerance, such as ±5 cm. In practice, these tolerances often either are not achieved during construction, or change as the bottom of the wetland accumulates sediments and plant detritus over time (Figure 2.15). Interestingly, some natural wetlands have about the same fine-scale distributions of soil elevations as do constructed wetlands.

The effect of such uneven bottoms upon the friction model depends upon the orientation and shape of the high spots and depressions (Stothoff and Mitchell-Bruker, 2003). Ridge features may either be parallel to flow, and act as flow-straighteners, or be perpendicular to flow and act as "speed bumps." To illustrate the potential effects, assume the bottom elevation distribution represents the flow cross section (Kadlec, 1990; Choi *et al.*, 2003). In order that water depth remain positive, depth is measured with respect to the lowest soil elevation. A purely geometric effect prevails: there is not much cross section available for flows at very low

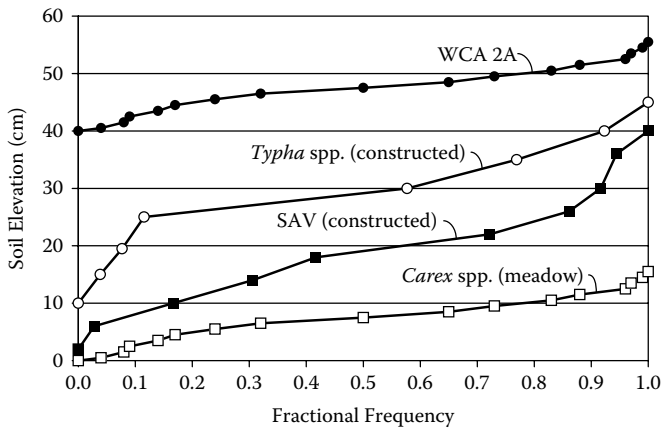



**FIGURE 2.15** Bottom elevation variability in natural and constructed wetlands, measured on a 10–20 m spacing. The datum in each case is arbitrary, and has been adjusted to provide vertical separation of the curves.

**TABLE 2.6**
**Speed and Direction of Water Movement at Various Depths in a Sparsely Vegetated Marsh, WCA2A**

| Depth (cm) | Speed (cm/s) | Direction (degrees) |
|---|---|---|
| 5 | 0.42 | 103 |
| 19 | 0.25 | 44 |
| 25 | 0.30 | 11 |
| 34 | 0.40 | 170 |
| 39 | 0.31 | 246 |
| 44 | 0.25 | 108 |

*Source:* Data from Romanowicz and Richardson (1997) *Hydrologic investigation of water conservation area 2-A*, Chapter 12 in the 1996-1997 Biennial Report to the Everglades Agricultural Area Environmental Protection District, Publication 97-05, Duke Wetland Center: Durham, North Carolina.

water stage. This effect is not lost until water depths are well above the point of complete inundation. Application of Equation 2.34 to a linear distribution produces a depth effect on conveyance capacity. For example, a straight line approximation to the cattail data in Figure 2.14, applied to flows at depths up to 60 cm, introduces depth dependence as represented by $b = 1.94$ in the general Equation 2.21, with $c = 1.0$.

### Wind Effects

Densely vegetated emergent FWS wetlands provide shelter from wind and minimize wind-driven water flow. However, the same is not true for open water areas, with or without submergent vegetation. It is possible to assess the potential for wind driven flow by comparing the drag force created by wind to that created by drag on submerged plant parts. For instance, at a wind speed of 5 m/s, for 200 one-cm stems per square meter and a water depth of 30 cm, wind drag is three times as strong as stem drag (based on Teeter *et al.*, 2001). As a consequence, surface water moves in the direction of the wind, with compensatory flows in lower water regions (Table 2.6). As yet, there is no practical predictive method of dealing with wind friction, and it therefore contributes to the variability of marsh friction calibrations.

## Wetland Data

### Generalized Friction Parameters

It would be desirable to have predictive methods for the parameters $a$, $b$, and $c$ in friction equations such as Equation 2.21. At the present time, data exist for only a few wetlands (Table 2.7). As discussed above, site-specific factors are known to be very important, and it is very dangerous to extrapolate from nonwetland information. Manning's coefficient is clearly not constant for the wetland environment, and it is preferable to utilize a model which describes the depth variability, namely Equation 2.21. The exponent $c$ is 0.5 in the turbulent open channel formulation. However, investigations on wetland systems indicate a higher value of $c$ is appropriate. As a limiting value, laminar flow around a uniform array of submerged objects over a flat bottom is theoretically described by $c = 1.0$. Until more data becomes available, a value of $c = 1.0$ is recommended.

The exponent $b$ is 1.67 in the turbulent open channel formulation. But the depth variability measured for wetlands increases this value, due to bottom irregularity and other factors. Until more data becomes available, a value of $b = 3.0$ is recommended for FWS wetland treatment systems. The coefficient $a$ remains a function of vegetation and litter density. Until more data become available, a value of $a = 1.0 \times 10^7$ $m^{-1}d^{-1}$ is recommended for densely vegetated wetlands, and $a = 5.0 \times 10^7$ $m^{-1}d^{-1}$ is recommended for sparsely vegetated wetlands.

Summary of recommended of recommended parameters for the generalized FWS friction relationship (Equation 2.21):

$$a = 1.0 \times 10^7 \text{ m}^{-1}\text{d}^{-1} \text{ (densely vegetated)}$$

$$a = 5.0 \times 10^7 \text{ m}^{-1}\text{d}^{-1} \text{ (sparsely vegetated)}$$

$$b = 3.0$$

$$c = 1.0$$

**TABLE 2.7**
**Friction Equation Coefficients for FWS Wetlands**

| Vegetation | Depth Range (cm) | Depth Exponent, $b$ | Slope Exponent, $c$ | Conveyance Coefficient, $a$ (m/d)/m$^{(b-1)}$ | Reference |
|---|---|---|---|---|---|
| Sedge | 0.05–0.25 | 3.00 | 0.71 | 2.00E + 08 | Kadlec *et al.* (1981) |
| Sedge | 0.08–0.30 | 2.50 | 1.00 | 5.00E + 07 | Kadlec (1990) |
| Sparse emergents | 0.20–0.80 | 1.44 | 1.00 | 6.20E + 06 | Bolster and Saiers (2002) |
| Sparse cattails | 0.30–0.85 | 1.60 | 1.00 | 1.80E + 07 | Choi *et al.* (2003) |
| Sparse sawgrass | 0.30–0.85 | 1.64 | 1.00 | 4.70E + 07 | Choi *et al.* (2003) |
| Cattail | 0.05–0.21 | 3.00 | 1.00 | 6.00E + 07 | Hammer and Kadlec (1986) |
| Cattail | 0.05–0.21 | 2.00 | 1.00 | 9.00E + 06 | Hammer and Kadlec (1986) |

**TABLE 2.8**
**Values of Manning's *n* Measured for FWS Wetlands**

| Project | Vegetation | Depth (m) | Velocity (m/d) | Reynolds Number | Manning's *n* ($s/m^{(1/3)}$) | Source |
|---|---|---|---|---|---|---|
| ENR Cell 1 | Cattails + SAV | 0.36–0.79 | 30–867 | 125–7,900 | 0.43–2.50 | Unpublished data SFWMD |
| ENR Cell 4 | SAV | 0.36–0.81 | 277–1,562 | 351–4,265 | 0.42–1.33 | Unpublished data SFWMD |
| Sacramento Cell 3 | Dense bulrush | 0.45–0.60 | 50–60 | 257–448 | 5.9–6.7 | Dombeck *et al.* (1998) |
| Sacramento Cell 7 | Dense bulrush | 0.45–0.60 | 40–75 | 367–928 | 2.1–7.6 | Dombeck *et al.* (1998) |
| Benton Cell 1 | Cattail | 0.17–0.35 | 400 | 770–1,070 | 13.8 | Unpublished data TVA |
| Benton Cell 2 | Woolgrass | 0.12–0.42 | 110–358 | 520 | 3.3 | Unpublished data TVA |
| Lewisville, Texas, Flume | Dense bulrush | 0.10–0.43 | 2,075–13,400 | 7,000–47,500 | 0.16–0.93 | Freeman *et al.* (1994) |
| Stennis Space Flume | Sawgrass | 0.15–0.75 | 132–3,950 | 460–23,000 | 0.32–1.80 | Jenter and Schaffranek (2001) |
| Boney Marsh | Mix | 0.30–0.70 | 35–135 | 108–713 | 1–4 | Mierau and Trimble (1988) |
| Chandler Slough | Water hyacinth | 0.40–0.70 | — | — | 0.20–0.55 | Shih and Rahi (1982) |
| Chandler Slough | Pickerel weed, buttonbush | 0.35–0.65 | — | — | 0.18–0.47 | Shih and Rahi (1982) |
| Shark River Slough | Sparse emergents | 0.10–0.60 | — | — | 0.40–2.50 | Rosendahl (1981) |
| WCA1 | Sawgrass | 0.15–1.50 | — | — | 0.33–1.20 | Shih *et al.* (1979) |
| WCA2A | Sawgrass | 0.15–1.50 | — | — | 0.32–1.20 | Shih *et al.* (1979) |
| Chandler Slough | Mix | 0.15–1.50 | — | — | 0.29–0.68 | Shih *et al.* (1979) |

## Manning's Coefficients

Although not appropriate for FWS wetlands, Manning's Equation (2.28) has, nevertheless, been widely used and calibrated in FWS wetlands (Table 2.8). Florida emergent marsh studies comprise a large fraction of the available wetland friction information. These serve to provide general guidelines for site-specific factors.

Generally, Manning's $n$ is strongly depth dependent for FWS systems, decreasing as depth increases. The nature of this dependence is illustrated in Figure 2.16 for two Florida marsh studies. Over a depth range of 30–90 cm, Manning's $n$ decreased by a factor of five for an emergent and submerged aquatic vegetation (SAV) wetland, and by a factor of three for a SAV-only wetland. This is somewhat surprising, because open channel theory predicts an increase in $n$ with increasing depth. Although that theoretical result has not been observed in treatment wetlands, there are examples of lesser depth dependence, such as the Boney Marsh FWS wetlands. Mierau and Trimble (1988) found no depth dependence of $n$ in an eight-year data analysis. Shih *et al.* (1979) found only a factor of two decrease over a depth range of 30–90 cm.

Likewise, $n$ values are dependent on vegetation density, because stems and litter provide the dominant drag surfaces. A linear relationship was found for *Schoenoplectus* (*Scirpus) validus* (Hall and Freeman, 1994). Therefore it is not surprising to find a strong seasonal dependence of $n$, because vegetation

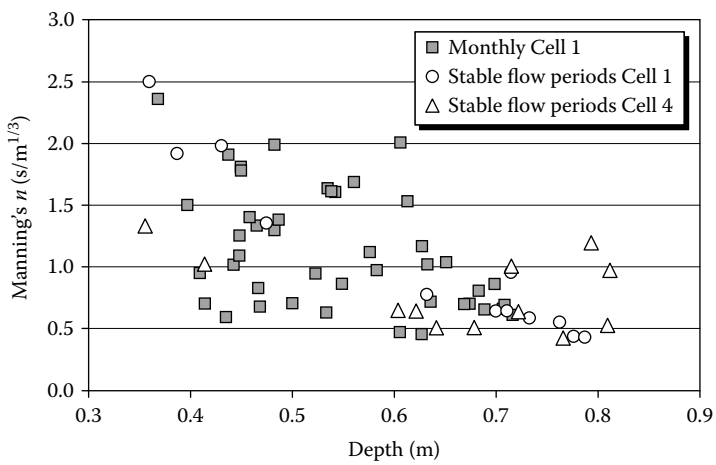



**FIGURE 2.16** Manning's $n$ versus depth for ENR project Cells. Cell 1 was an emergent-SAV mix; Cell 4 was SAV. Data span four years. Stable flow periods are at constant flow, monthly values include changes in storage and flow.

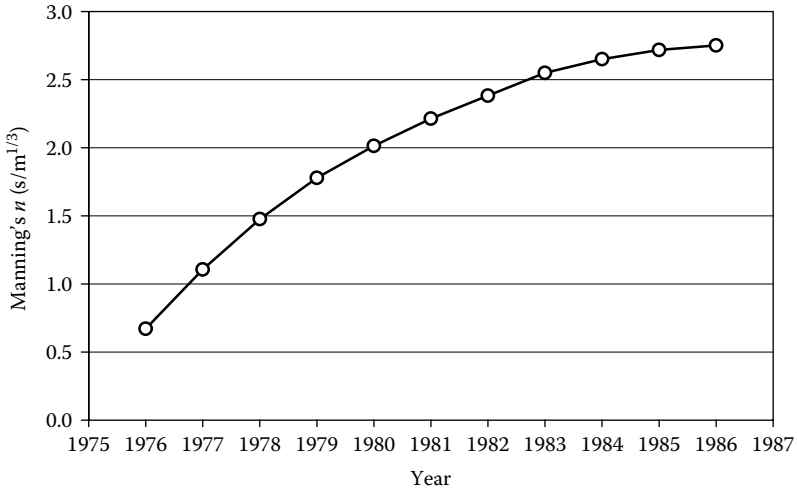



**FIGURE 2.17** The increase of friction in a developing wetland, as shown by the central tendency of each year's data. Vegetation densities in Boney Marsh may have been only part of the reason for increases: increases in litter and sediment accumulation could have altered these measurements. (Data from Mierau and Trimble (1988) *Hydrologic characteristics of the Kissimmee River Floodplain Boney Marsh Experimental Area*. Technical memorandum, South Florida Water Management District (September 1988); graph from Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

changes seasonally (Shih and Rahi, 1982). Because both litter and live stems are involved, the relation is not easily predictable; it depends on litterfall events.

The progress of a constructed system from an initial sparse vegetation to a more densely vegetated condition is accompanied by increases in the friction coefficient. Boney Marsh, Florida, received pumped river water over several years beginning in 1976. Hydrologic studies produced weekly values of Manning's $n$ (Mierau and Trimble, 1988). The biological dynamics of the Boney marsh operation produced considerable scatter in Manning's $n$, but the year-to-year trend line was upward from 0.6 to 2.7 s/m$^{1/3}$ (Figure 2.17).

## Head Loss Calculations

The implementation of Equation 2.23 requires numerical integration, which is inconvenient in conceptual design calculations. But because of the extreme nonlinearity of the equations, it is very inaccurate to use average values. Accordingly, it is better to use precalculated values of the head loss for the intended design conditions. To accomplish this, the case of a rectangular constructed wetland is considered, with a negligible loss or gain of water due to $P$ and $ET$. Equation 2.23 is de-dimensionalized using the wetland length and the outlet water depth:

$$y = \frac{h}{h_o} \qquad y_b = \frac{B}{h_o} \qquad z = \frac{x}{L} \qquad S_1 = \frac{dy_b}{dz} \tag{2.35}$$

$$y^3 = \left(\frac{d(y+y_b)}{dz}\right) = y^3\left(-\frac{dy}{dz} - S_1\right) = \frac{qL^2}{ah_o^4} = M_1 \tag{2.36}$$

where

$h$ = water depth, m
$h_o$ = water depth at outlet, m
$\boldsymbol{B}$ = elevation of the bed bottom above datum, m
$L$ = wetland length, m
$q$ = hydraulic loading rate, m/d

and the rest of the new variables are defined in Equations 2.35 and 2.36. It is presumed that the outlet water depth is fixed. Integration of Equation 2.36 yields the inlet water depth, and hence the head loss for a given wetland. Solutions depend on two parameters: $S_1$, which represents the bed slope, and $M_1$, which contains the friction coefficient, the hydraulic loading rate, outlet depth, and wetland length. Figure 2.18 presents the solution of Equation 2.36 for different parameter values. It may be used to estimate head losses in FWS wetlands.

### *An Example*

A surface flow wetland is to be built to treat 200 m$^3$/d of secondary municipal wastewater. The appropriate hydraulic loading rate has been determined to be 2 cm/d. Site considerations indicate that a length of 400 meters is desirable. A bed slope of 20 cm over the 400-m length is to be used to provide drainage. The outlet weir is to be set to maintain 20 cm depth at the outlet. What is the estimated head loss?

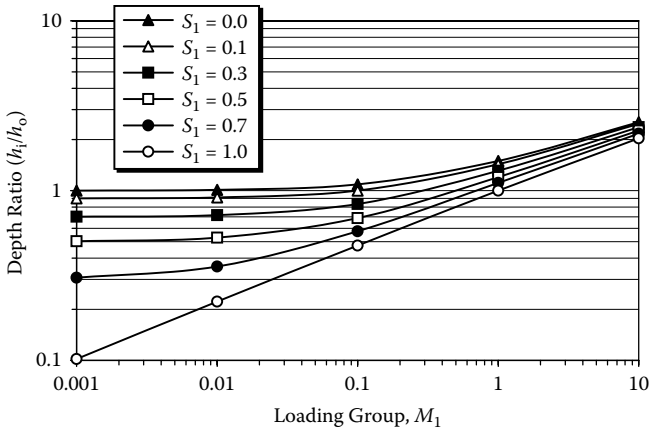



**FIGURE 2.18** Inlet/outlet depth ratio for FWS wetlands of different slopes and different loading rates. The friction power law is used, with $b = 3$ and $c = 1$. (From Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

$$S_1 = \frac{L}{h_0}\left(-\frac{dB}{dx}\right) \quad M_1 = \frac{qL^2}{h_0^4 a}$$

The constants needed to use Figure 2.18 are:

$$a = 1 \times 10^7 \text{ m}^{-1}\text{d}^{-1}$$
$$dB/dx = 0.20 / 400 = 0.0005$$
$$h_o = 0.20 \text{ m}$$
$$S_1 = 1.0$$
$$q = 0.02 \text{ m/d}$$
$$L = 200 \text{ m}$$
$$M_1 = \frac{(0.02)(400)^2}{(0.2)^4(1 \times 10^7)} = 0.2$$

Referring to Figure 2.18, the ratio of inlet depth to outlet depth is 0.6. Therefore:

$$h_i = (0.6)(0.20) = 0.12 \text{ m}$$
$$H_i = B_i + h_i = 0.20 + 0.12 = 0.32 \text{ m}$$
$$\Delta H = 0.32 - 0.20 = 0.12 \text{ m} = 12 \text{ cm}$$

## 2.3 HSSF WETLAND HYDRAULICS

The idea of flowing water through a planted bed of porous media seems simple enough; yet numerous difficulties have arisen in practice. Sometimes these problems have been traced to incorrect design calculations; at other times problems have resulted from changes in the conditions in the bed. A great deal of confusion has been evidenced regarding the movement of water through HSSF wetlands. Rules of thumb abound in the literature, many of which do not acknowledge the simple physics of water movement. The literature is replete with misapplications of the fundamental relations between head loss and flow rate. In this section, relevant calculations are examined and bounds placed on the variables governing the ability of wetlands to operate in subsurface flow with rooted macrophytes.

Prior to 1995, gravel bed HSSF wetlands in the United States were frequently observed to be flooded (Kadlec and Knight, 1996). The two leading causes were clogging of the media and improper hydraulic design. The same appeared to be true for other countries as well (Brix, 1994a), especially HSSF wetlands that used soil for the bed medium. Flooded HSSF systems have been tolerated in many instances because the hydraulically failed mode of flooded operation is the FWS wetland, which may provide treatment performance nearly as efficient as the HSSF wetland.

### Flow in Porous Media

There is a very long history of research and development related to flow in porous media. Descriptions of flow phenomena started with the propositions of Darcy in 1856 (Brown, 2002), and have grown to include several texts on the subject. Several types of flow can occur in general; here the concern is solely for the case of fully saturated flow with an unconfined top interface with air, either in or above the bed. Full saturation refers to the absence of a capillary fringe, in which both air and water occupy the voids between particles.

HSSF wetlands operate in thin sheet flow, with a free upper surface. Flows may be averaged over the vertical (thin) dimension, for the case of the upper surface exposed to the atmosphere, to yield the one-dimensional Dupuit–Forcheimer equation:

$$\frac{\partial(\varepsilon H)}{\partial t} = \frac{\partial}{\partial x}\left(kH\frac{\partial H}{\partial x}\right) + P - ET \tag{2.37}$$

where

$ET$ = evapotranspiration loss, m/d
$P$ = precipitation, m/d
$k$ = hydraulic conductivity, m/d
$H$ = elevation of the free water surface, m
$x$ = longitudinal distance, m
$\varepsilon$ = porosity, dimensionless

It is important to note that this equation embodies the assumption that the driving force for flow is a tilt to the water surface ($\partial H/\partial x$). A simpler version of this theory will suffice for HSSF wetland design purposes.

### Adaptations for HSSF Wetlands

The following developments presume that the wetland is in a steady state condition, but later it will be shown that this is rarely the case. The representation will therefore be for long-term, average performance. It is presumed that the porous medium is isotropic. This is probably not true, due to the presence of plant roots and other introduced particulates. The variability in the vertical and transverse directions is accounted for by averaging. Longitudinal variations in

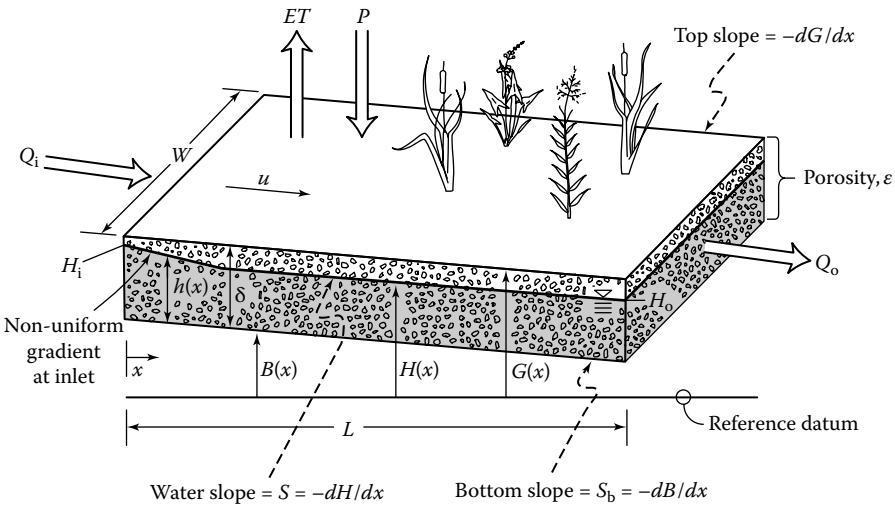



**FIGURE 2.19** Notation for HSSF bed hydraulic calculation for the simplest case. The actual velocity of water is $v = u/\varepsilon$. The subscripts *i* and *o* stand for *inlet* and *outlet*, respectively. (From Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.) Note:

$B(x)$ = elevation of bed bottom, m
$G(x)$ = elevation of bed surface, m
$H(x)$ = elevation of water surface, m
P = precipitation, m/d
$x$ = distance from inlet, m
ET = evapotranspiration, m/d
$h(x)$ = water depth, m
$L$ = bed length, m
$Q$ = volumetric flow rate, $m^3/d$
$\delta$ = bed depth, m

hydraulic conductivity are also present after the wetland has been in operation for a time. Most HSSF wetlands are rectangular, and so that feature is added to the list of restrictions. Notation is outlined on Figure 2.19.

The mass balances and geometrical definitions have been presented in Equations 2.20 through 2.22, which also hold for HSSF wetlands. The porosity is lower, usually in the range 0.35–0.45 $m^3/m^3$ for sands and gravels; and there is the added geometry of a bed surface to consider. The elevation of the top surface of the media is:

$$G = B + \delta \tag{2.38}$$

where
$G$ = elevation of the bed top above datum, m
$\delta$ = thickness of the bed media, m

The freeboard, or headspace, is defined to be the distance from the top surface of the media down to water:

$$f = \delta - h \tag{2.39}$$

where
$f$ = freeboard, m

In general, the variables $h$, $H$, $G$, $\delta$, $f$, and $B$ are each dependent on distance from the bed inlet.

## Bed Friction and Hydraulic Conductivity

The simplest friction relationship states that superficial velocity is proportional to the slope of the water surface:

$$u = -k\frac{dH}{dx} \tag{2.40}$$

where
$H$ = elevation of the water surface, m
$k$ = hydraulic conductivity, m/d

This is the one dimensional version of Darcy's law. It is restricted to the laminar flow regime.

A more general correlation spans both laminar and turbulent flow. The laminar term in Equation 2.40 is preserved, and a turbulent term is added:

$$-\frac{dH}{dx} = \frac{1}{k}u + \omega u^2 \tag{2.41}$$

where
$\omega$ = turbulence factor, $d^2/m^2$

The turbulent contribution $\omega u^2$ is negligible when the particle Reynolds number is less than 1.0, and may be ignored

with small error at Reynolds numbers up to 10. The particle Reynolds number is defined as:

$$\mathrm{Re} = \frac{D\rho u}{(1-\varepsilon)\mu} \tag{2.42}$$

where
$D$ = particle diameter, m
$\rho$ = density of water, kg/m$^3$
$\mu$ = viscosity of water, kg/m·d

Sand media will typically be in the laminar range; but rock media will often be in the transition region between laminar and turbulent, with significant contributions from the turbulent term. Simple rearrangement of Equation 2.42 gives:

$$u = -k_e \frac{dH}{dx} \tag{2.43}$$

where
$k_e$ = effective hydraulic conductivity, m/d

Comparison of Equations 2.41 and 2.43 indicates that:

$$\frac{1}{k_e} = \frac{1}{k} + \omega u \tag{2.44}$$

When velocity is beyond the laminar range, the effective hydraulic conductivity will depend on velocity.

## Correlations for Hydraulic Conductivity of Clean Bed Porous Media

The original "clean bed" hydraulic conductivity and turbulence factor for a particulate media depend on the characteristics of the media:

1. Mean particle diameter
2. Variance of the particle size distribution
3. Particle shape
4. Porosity of the bed
5. Arrangement of the particles

Of these, the effects of particle size and porosity have been quantified in the form of equations in the nonwetland literature. For instance, the Ergun equation (Ergun, 1952) is widely accepted for random packing of uniform spheres:

$$-\frac{dH}{dx} = \left(\frac{150\left(1-\varepsilon\right)^2 \mu}{\rho g \varepsilon^3 D^2}\right) u + \left(\frac{1.75\left(1-\varepsilon\right)}{g \varepsilon^3 D}\right) u^2 \tag{2.45}$$

where
$H$ = elevation of water surface, m
$\varepsilon$ = porosity, dimensionless
$D$ = particle diameter, m
$\rho$ = density of water, kg/m$^3$
$\mu$ = viscosity of water, kg/m/d
$u$ = superficial flow velocity, m/d
$g$ = acceleration of gravity, m/d$^2$

Comparison with Equation 2.41 indicates that:

$$k = \frac{\rho g \varepsilon^3 D^2}{150(1-\varepsilon)^2 \mu} \tag{2.46}$$

$$\omega = \frac{1.75(1-\varepsilon)}{g \varepsilon^3 D} \tag{2.47}$$

Equation 2.45 works for spheres of a single size; but gravel bed wetlands do not utilize such media. Hu (1992) applied Equation 2.45 to a HSSF system at Bainikeng, China, and found that Ergun-predicted depths were about 10 cm too large. The effects of a nonspherical shape are also significant (Brown and Associates, 1956). Idelchik (1986) gives a correlation for crushed, angular materials, which predicts conductivities about three times lower than those for spheres of the same size.

Most media possess a distribution of sizes. The presence of a particle size distribution lowers the hydraulic conductivity. This occurs because small particles have a disproportionately large amount of surface area, which causes drag on the water, and because the small particles can fit in the spaces between the larger particles. For instance, Freeze and Cherry (1979) present a technique based on work of Masch and Denny (1966) that utilizes the variance of the particle size distribution to estimate a correction factor for the hydraulic conductivity of large sand particles. For a variance of 50% of the mean particle size, the reduction is a factor of two.

Given all the uncertainties above, each of which can greatly influence the hydraulic conductivity of the clean media, it is prudent to measure the conductivity of the candidate media for a proposed project. Correlations may be used to guide the initial selection, but should not be trusted for final design purposes, because the gradient, porosity, and velocities have seldom been reported. Data for media from eighteen treatment wetland sites are displayed along with a prediction based upon a modification of Equation 2.45 in Figure 2.20. It is very important to recognize that Figure 2.20 is valid only for bare media with a porosity near 0.35 and a size variance near 50%.

## Clogging of HSSF Bed Media

The HSSF bed will not maintain the clean-bed hydraulic conductivity once the system is placed into operation. For example, if one third of the pore space is blocked, the hydraulic conductivity will decrease by factor of ten, according to Equation 2.46, because hydraulic conductivity is extremely sensitive to porosity. This phenomenon must be acknowledged in design if the potential for bed flooding is to be minimized. Clogging of HSSF wetland beds occurs via the following mechanisms:

1. Deposition of inert (mineral) suspended solids in the inlet region of the wetland bed
2. Accumulation of refractory organic material (resistant to microbial degradation) in the inlet zone of the wetland bed

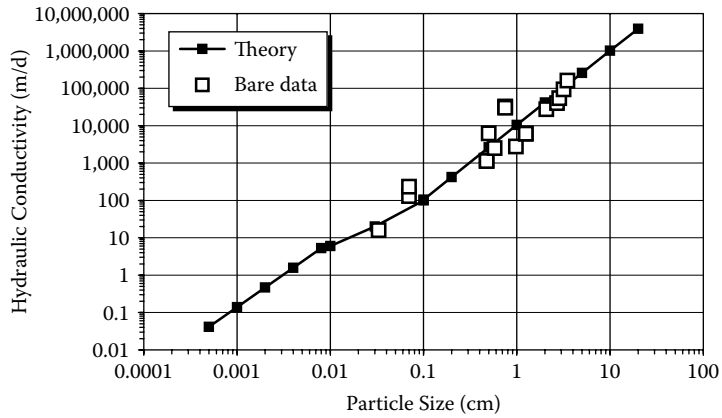



**FIGURE 2.20** The dependence of clean bed hydraulic conductivity on media grain size. This plot is approximate; it is based on a porosity of 35%, and a 50% variance in the particle size distribution. Other size distributions, and deviation in particle shape and packing, will influence values for specific media. Data from 18 SSF wetlands is superimposed. (Data from Wolstenholme and Bayes (1990) In *Constructed Wetlands in Water Pollution Control*. Cooper and Findlater (Eds.), Pergamon Press, Oxford, United Kingdom, pp. 139–148; McIntyre and Riha (1991) *Journal of Environmental Quality* 20: 259–263; Sanford *et al*. (1995a) *Ecological Engineering* 4(4): 321–336; Kadlec and Watson (1993) In *Constructed Wetlands for Water Quality Improvement*. Moshiri (Ed.), Lewis Publishers, Boca Raton, Florida, pp. 227–235; Fisher (1990) In *Constructed Wetlands in Water Pollution Control*. Cooper and Findlater (Eds.), Pergamon Press, Oxford, United Kingdom, pp. 21–32; George *et al*. (1998) *Development of guidelines and design equations for subsurface flow constructed wetlands treating municipal wastewater.* Draft report to U.S. EPA, Cooperative Agreement CR818724–01–3, Cincinnati, Ohio; Watson and Choate (2001) *Hydraulic conductivity of onsite constructed wetlands*. Mancl (Ed.), Proceedings of the Ninth National Symposium on Individual and Small Community Sewage Systems; American Society of Agricultural Engineers: St. Joseph, Michigan, pp. 632–649; Drury and Mainzhausen (2000) *Hydraulic characteristics of subsurface flow wetlands.* Proceedings of the Billings Land Reclamation Symposium, Billings, Montana; graph from Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

3. Deposition of chemical precipitates in the wetland bed
4. Loading of organic matter (both suspended and dissolved) that stimulates the growth of microbial biofilms on the bed media
5. Development of plant root networks that occupy pore volume within the wetland bed

### Sediment Deposition

Solids deposition can occur for a variety of reasons, beginning with the placement of the media. Unwashed media will carry a load of fine dust or soil. Mud on the wheels of vehicles can add to the dirt supply during placement. And, those beds which are constructed with a layer of fine media on top of coarse media can be subject to the penetration of the lower layer by the upper-lying layer of finer material. Planting activities can introduce soils associated with the roots of the plants.

Due to the low flow velocities that occur within HSSF wetland beds, influent total suspended solids (TSS) will settle and deposit within the inlet region of the wetland bed. This deposition typically occurs within the first 5% of the wetland bed. As pore volume is occupied by suspended solids, the hydraulic conductivity is reduced accordingly, as described by Equation 2.46. This mechanism applies both to mineral (or inert) sediments as well as organic sediments that are refractory and resistant to microbial degradation (Mechanisms #1 and #2).

### Chemical Precipitates

Chemical reactions within HSSF wetlands can result in the formation of insoluble chemical precipitates (Mechanism #3) (Liebowitz *et al.*, 2000; Younger *et al.*, 2002). These precipitates can also block pore spaces within the wetland bed and have the same effect in reducing hydraulic conductivity as described by Equation 2.46. Since the formation of precipitates is primarily governed by the redox potential within the wetland bed, reductions in hydraulic conductivity are not restricted to the inlet end of the wetland bed.

### Biomat Formation

Microbial biofilms form in response to both particulate and soluble organic loading rates (Mechanism #4). These biofilms entrap both organic and inorganic solids (Winter and Goetz, 2003), forming a *biomat*. This biomat varies depending on the nature of the waste being treated. Biomat formation is greatest at the inlet end of the wetland where the organic loading is highest (Ragusa *et al.*, 2004). The loss of pore volume due to biomat formation reduces the hydraulic conductivity in this inlet zone (Zhao *et al.*, 2004). Organic matter is removed as wastewater flows through the wetland, resulting in declining biomat growth. At the outlet, where only small quantities of organic matter are available to microbes, biomat formation is negligible.

## Plant Root Morphology

Wetland plants in HSSF systems develop a preferential rooting preference in the upper region of the granular bed (Mechanism #5). This rhizome morphology is strongly dependent on redox conditions within the HSSF bed (Lockhart, 1999) and is described further in Chapter 3. This limited root penetration can create preferential flow paths through the lower section of the gravel bed (Breen and Chick, 1995; U.S. EPA, 2000a; Whitney *et al.*, 2003; Nivala, 2005).

## HSSF Bed Clogging in the Inlet Region

As a consequence of these factors, the HSSF bed will become clogged over time. Since the primary mechanisms for bed clogging predominate in the inlet region, the greatest reductions in hydraulic conductivity occur at the wetland inlet. Bed clogging has created the majority of operational problems for HSSF wetlands around the world. Bed clogging is not a new phenomenon, although the mechanisms by which it occurs are only now being elucidated. As noted by Zachritz and Fuller (1993), "Clogging ... has been an operational problem since plant start-up" at the Carville, Louisiana facility.

These operational problems are being evaluated in current HSSF wetlands. For instance, Cooper *et al.* (2006) report that 111 of 255 reedbeds inspected were flooded at the inlet end. These had a median age of about ten years.

## Development of HSSF Bed Clogging

Although the processes of bed clogging are still being quantified, there appear to be two distinct sets of mechanisms that contribute to the problem:

- Short-term effects that reduce hydraulic conductivity over the first year of operation. These appear to be related primarily to the development of plant root networks (primarily in the upper regions of the wetland bed) and microbial biomat formation occurs primarily in the inlet region of the wetland (Mechanisms #4 and #5).
- Long-term effects that gradually reduce hydraulic conductivity. These appear to be primarily related to deposition of inert (mineral) suspended solids, accumulation of refractory organic material, and formation of insoluble chemical precipitates (Mechanisms #1, #2, and #3).

### *Short-Term Bed Clogging Mechanisms*

The majority of porosity decrease appears to occur during the first year after bed commissioning (Figure 2.21). That is the period of principal root/rhizome grow-in, and the development of biofilms. In total, the losses in Figure 2.21 are 16% and 23%. The loss of porosity is reflected in a reduction of residence time. For example, Marsteiner *et al.* (1996) reported about a 10% loss of detention time for planted beds versus unplanted beds, and Tanner *et al.* (1998a) estimated root/rhizome blockage to be 4%.

The nonuniform distribution of roots and biomat along the length of the bed results in a nonuniform distribution of hydraulic conductivity throughout the bed, as shown schematically in Figure 2.22.

The result of porosity decrease is a severe decline in bed hydraulic conductivity, primarily in the front end of the bed. McIntyre and Riha (1991) showed that hydraulic conductivity drops during the early months of plant establishment (Figure 2.23). These mesocosms were fed a nutrient solution, and therefore incoming sediments were negligible. Interestingly, both planted and unplanted mesocosms showed reduced conductivity, with plants only slightly increasing the loss. Wolstenholme and Bayes (1990) documented a similar pattern of drastic conductivity reduction for four reedbeds at Valleyfield, Scotland, over the first year of operation.

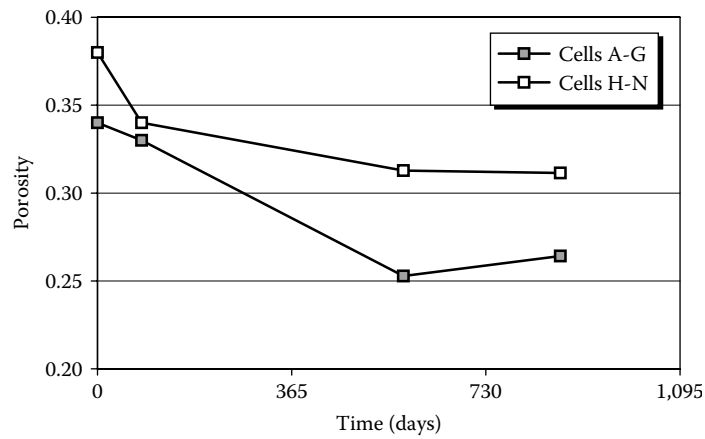



**FIGURE 2.21** The decrease in HSSF bed porosity over time. (Data from George *et al.* (1998) *Development of guidelines and design equations for subsurface flow constructed wetlands treating municipal wastewater.* Draft report to U.S. EPA, Cooperative Agreement CR818724–01–3, Cincinnati, Ohio.)

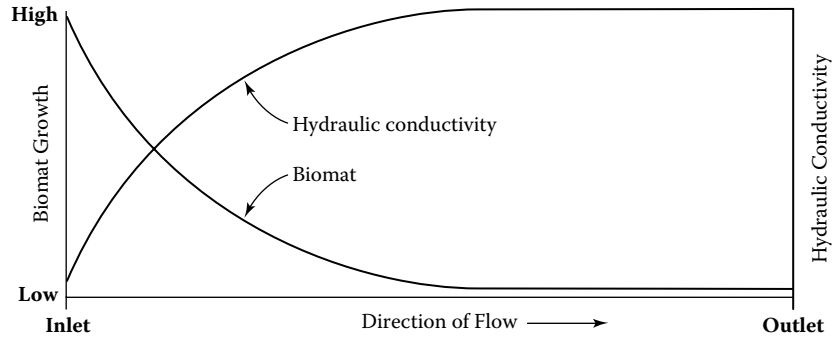



**FIGURE 2.22** Relationship between hydraulic conductivity and biomat formation. (From Wallace and Knight (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements*. Final Report, Project 01-CTS-5, Water Environment Research Foundation (WERF): Alexandria, Virginia. Reprinted with permission.)

### *Long-Term Bed Clogging Mechanisms*

Regardless of short-term effects, HSSF wetlands that receive a sustained load of particulate matter will experience a continuing loss of porosity over time, and corresponding reductions in hydraulic conductivity. Accumulation of inert sediments and refractory organic material will occur primarily in the inlet region of the wetland bed, and exacerbate conductivity losses in this region. Formation of chemical precipitates is dependent on the redox conditions within the HSSF wetland bed, and may not be confined to just the inlet zone.

## HSSF Bed Clogging

The combined effect of short-term and long-term bed clogging mechanism is to produce a drastic reduction in the hydraulic conductivity of the inlet zone of the wetland bed.

The magnitude of conductivity decline may well over a factor of ten, as it was at the Richmond, New South Wales, Australia, site (Figure 2.24). Most of the decline in this particular system was apparently associated with biomat formation in the inlet region, as unplanted gravel beds showed declines similar to those found in the planted systems (Fisher, 1990; Sanford *et al.*, 1994). The microbial populations associated with nutrient cycling and BOD reduction are highest in the inlet section of the bed, in response to the elevated contaminant concentrations in that region. These organisms, together with their associated biomats, reduce the pore volume in the entrance region of the bed to a greater extent than the downstream sections. In turn, this implies a greater reduction of hydraulic conductivity in the inlet region and a resulting nonuniform hydraulic gradient. This effect has been measured by several investigators (Fisher, 1990; Kadlec and Watson, 1993; Watson and Choate, 2001), as illustrated in Figure 2.25.

Kadlec and Watson (1993) found approximately 10% of the voids blocked by volatile and inorganic solids. Tanner *et al.* (1998a) investigated the loss of detention time (porosity) over a five year period for beds receiving organic solids loadings of 1.73, 2.09, and 5.80 g/m$^2$·d. Detention times were 101%, 61%, and 50% of theoretical, respectively. In consideration of these factors, there have been proposed relations

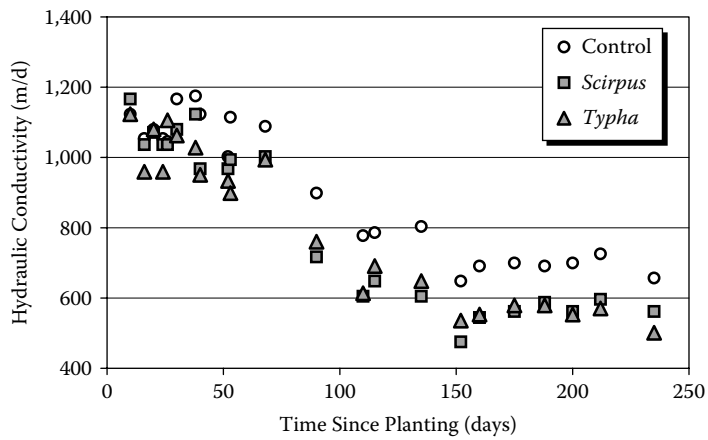



**FIGURE 2.23** Decrease of hydraulic conductivity of planted laboratory mesocosms. There is evidence for reduction by both biofilms and root growth. (Data from McIntyre and Riha (1991) *Journal of Environmental Quality* **20**: 259–263.)

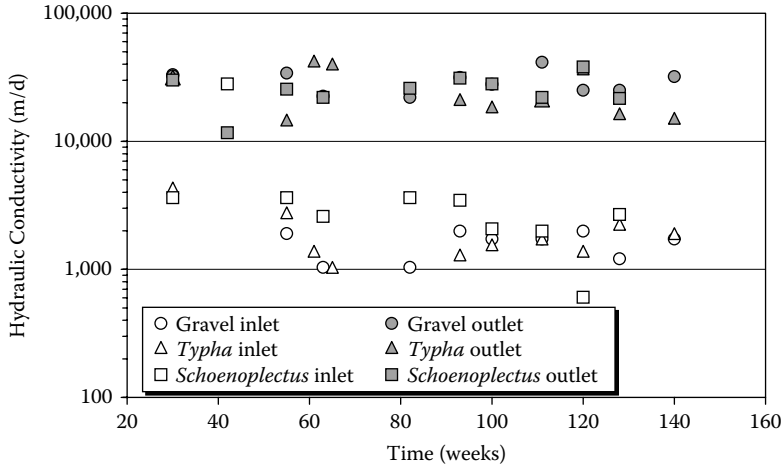



**FIGURE 2.24** The dependence of hydraulic conductivity on time, adapted from Fisher (1990). Three beds are represented: an unplanted gravel bed, gravel planted with *Scirpus*, and gravel planted with *Typha*. Data fall into two groups: the inlet sections, 0–20 m from the inlet; and the rest of the bed, 20–100 m from the inlet. (From Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

between time to complete clogging and the solids loading to the wetland (Bavor and Schulz, 1993; Blazejewski and Murat-Blazejewska, 1997; Langergraber *et al.*, 2003; Wallace and Knight, 2006). Additional factors that may contribute HSSF bed clogging are addressed in Chapter 7.

Soil based systems, as formerly used in Europe, may not display clogging problems, simply because the hydraulic conductivity assumed for design purposes (and associated inlet zone organic loadings) are already extremely low (ÖNORM B 2505, 1997; ATV, 1998) and, as a consequence, may match the conductivity of deposited influent solids. Haberl and Perfler (1990) found no evidence of reduced conductivities over five years for any of three HSSF soil-based wetlands at Mannersdorf, Germany. They found variable changes in hydraulic

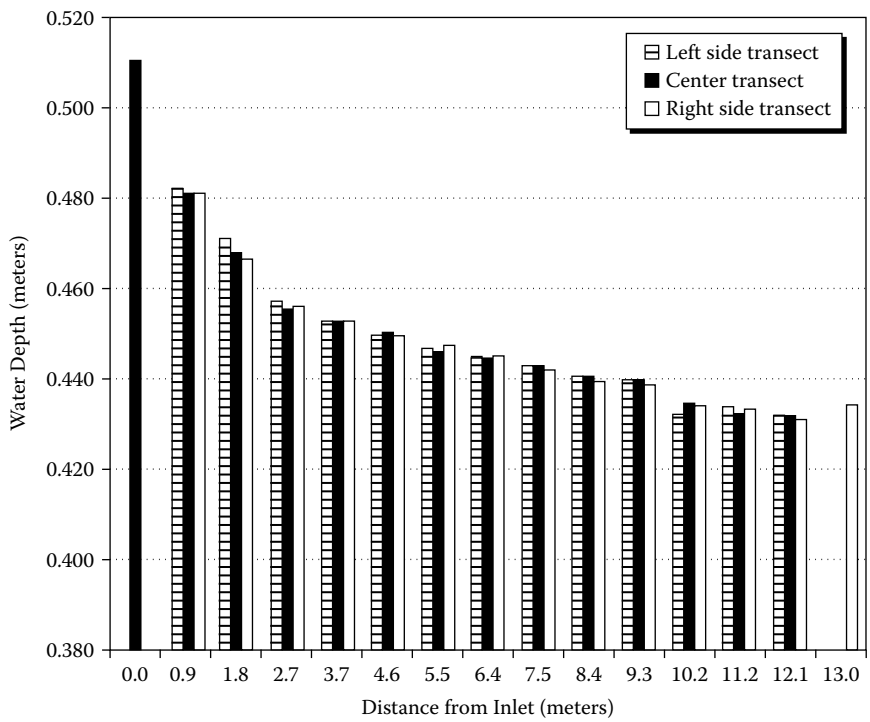



**FIGURE 2.25** Water surface profile of a single-home HSSF wetland in Alabama. (From Watson and Choate (2001) *Hydraulic conductivity of onsite constructed wetlands*. Mancl (Ed.). Proceedings of the Ninth National Symposium on Individual and Small Community Sewage Systems; American Society of Agricultural Engineers: St. Joseph, Michigan, pp. 632–649. Reprinted with permission.)

**TABLE 2.9**
**Typical Values of Hydraulic Conductivity for Soil Materials**

| Soil Texture | Hydraulic Conductivity (m/d) |
|---|---|
| Gravel, coarse sand | 36.0 + |
| Coarse, medium sand | 6.0–36.0 |
| Fine sand, loamy sand | 2.4–6.0 |
| Sandy loam, loam | 1.2–2.4 |
| Loam, porous silt loam | 0.6–1.2 |
| Silty clay loam, clay loam | 0.3–0.6 |

*Source:* Data from U.S. EPA (2002c) *Onsite wastewater treatment systems manual*, EPA 625/R-00/008, U.S. EPA Office of Research and Development: Washington, D.C.

conductivity, with minimum and maximum values of 0.37 ± 0.34, and 3.8 ± 2.2 m/d, respectively. Coombes (1990) reports conductivities for several United Kingdom HSSF reed beds, in the range of 0.2–9.8 m/d, which is the same as the range shown in Table 2.9.

The soil-based system at Acle, Norfolk, United Kingdom, had essentially the same conductivity at year four as at startup. However, because of low conductivity, soil-based systems are prone to flooding and commonly operate in a flooded mode (similar to a FWS wetland).

At the present time, there is not a clear understanding of the rate and development of HSSF bed clogging. Short-term mechanisms appear to be clearly related to plant root development, organic loading, and the size of the bed media. However, even if short-term clogging is avoided through the proper application of design principles, long-term clogging mechanisms are still operative. There is a growing body of knowledge in North America and Europe that indicates HSSF bed clogging may be inevitable (Cooper *et al.*, 2004; Wallace and Knight, 2006; Cooper *et al.*, 2006; Puigagut *et al.*, 2006). Whereas the rate of bed clogging is related to mechanisms discussed in this chapter, regularly scheduled replacement or cleaning of the bed media in the inlet zone of HSSF wetlands may be a routine (and unavoidable) part of the operation of HSSF wetlands.

The implications of these clogging phenomena in design are very important. The HSSF wetland must be able to operate properly and be capable of establishing and sustaining plant growth, in the face of large changes in hydraulic characteristics that will occur over the life of the system.

## HSSF Water Elevation Profiles

More than just Darcy's law is required to calculate flow rates and depths in a HSSF wetland. Previous equations provide the ability to calculate $h$, $H$, $u$, and $Q$ as functions of distance down the length of the bed. Because two of these are differential equations, an integration procedure must be implemented and boundary conditions must be specified. The required equations for the case of steady flow without atmospheric augmentation are:

$$\frac{d\left[u\,(H-B)\right]}{dx} = 0 \tag{2.48}$$

$$-\frac{dH}{dx} = \frac{1}{k_e}u \tag{2.49}$$

Given a bottom elevation profile $B(x)$, Equations 2.48 and 2.49 provide the volume flow and depth profiles from:

$$Q = u\,W\,h \tag{2.50}$$

$$h = H - B \tag{2.51}$$

The boundary conditions would most often be a specification of the exit water elevation (set by a structure), and a specification of the inlet flow rate (set by the delivery system):

$$H_{\,at\ x=L} = H_o \quad \text{and} \quad Q_{\,at\ x=0} = Q_i \tag{2.52}$$

The required input information must also include: bed width $W$, the bottom elevation profile $B(x)$, the hydraulic conductivity profile $k(x)$, and the turbulence factor profile $\omega(x)$.

There is a very important constraint to be met in the course of solution of these model equations: flow must be underneath the media surface, or the hydraulic conductivity Equation 2.49 does not apply. Mathematically, this means satisfying the inequality:

$$0 < h < \delta \tag{2.53}$$

These model equations may be solved on a spreadsheet without great difficulty.

## Water Surface Calculations

The common assumption is that most HSSF beds have uniformly flat but inclined bottoms. Many operate in the laminar flow region. Since HSSF wetlands tend to be loaded at higher hydraulic loading rates than FWS wetlands, effects of atmospheric gains or losses are minimized. Most HSSF systems are intended to operate at constant water depth. In their initial startup condition, the hydraulic conductivity will not be a function of distance from the inlet. Under these ideal conditions, the model reduces to:

$$\frac{H_i - H_o}{L} = \frac{u_{avg}}{k} = \frac{1}{k}\ \frac{Q}{Wh_{avg}} \tag{2.54}$$

It is important to note that *the gradient on the left side of this equation is the slope of the water surface, not the bottom of the wetland bed.*

The bottom bed slope within a HSSF wetland will not drive the flow. Similar to lakes, if there is not a slope to the water surface, there is no flow of water.

The use of Equation 2.54 by itself for design can lead to serious errors, and has done so. The average water depth $h_{avg}$

is not adequate for design; the longitudinal depth profile is required. The problems are related to the large changes in $k$, to the violation of the constraint of Equation 2.53 (flooding), and to the sensitivity of vegetation to the headspace, $f = \delta - h$. Basically, the design goal is to keep the water below the surface of the bed, but high enough for plant roots to reach it.

Some sense of the validity and sensitivity of the model can be gained from its calibration to field data at Benton, Kentucky. The inlet zone of this crushed limestone HSSF cell was flooded at the time of the study, which was about three years after startup. Measurements included detailed surveys of the water surface elevation and of flow rates. The media were tested in the field to determine the *in situ* conductivity (TVA, 1989, unpublished data).

The washed media was also tested in the laboratory (Kadlec and Watson, 1993). Although the lab conductivity was roughly comparable to the field values, the lab value was clearly on the high side (67% above field mean). That kind of difference can easily arise from differences in void fraction resulting from packing factors. Further, the media in the bed contained some considerable amount of dirt from work vehicles. A twelve percent void fraction difference would account for the differences in $k$. The predicted profile of water depth is a straight line in Figure 2.26. However, the conductivity in the inlet zone was low, and increased markedly along the flow direction (Figure 2.27). If clogging is accounted for in the hydraulic conductivity profile, the model fits the water surface profile correctly (Figure 2.26).

These conditions created large variations from the original design intent. Effects on vegetation were huge due to hydroperiod shifts: the inlet section (20%) became an overland flow *Schoenoplectus* (*Scirpus*) wetland due to flooding. The remaining 80% of the bed contained only sparse terrestrial vegetation, because the plants responded to excessive headspace ($f$), which created a large unsaturated rooting depth and effectively precluded the growth of emergent wetland plants.

## Flooded Operation

A combination of clogging and inappropriate design has produced overland flow in many existing HSSF systems (Figure 2.26). Flooding is usually confined to the inlet region of the bed. Overland flow carries the excess water until the hydraulic conductivity and flow gradient over the remaining travel distance are sufficient to permit the flow to be carried below ground. The amount of water carried in overland flow may be estimated from methods presented earlier in this chapter for FWS wetlands. If over-flooding depths are as much as a few centimeters, most of the water will be carried to the HSSF wetland outlet by overland flow. As reported by Spangler *et al.* (1976b), "*The main flow of water, however, was horizontally across the surface to the area immediately above the outlet, and then vertically down to the outlet....*" As a simple approximation, the HSSF system transforms into a FWS wetland.

The Denham Springs, Louisiana, HSSF wetlands are a case in point (Figure 2.28). Even though these beds are filled with 6 cm rock, a combination of the aspect ratio (L:W = 5:1) and the high flow (hydraulic loading rate of 25 cm/d) led to extensive inlet zone flooding. Predictions from Equation 2.54, even based on clean-bed hydraulic conductivity, indicate such flooding should be expected, but such calculations were apparently not performed.

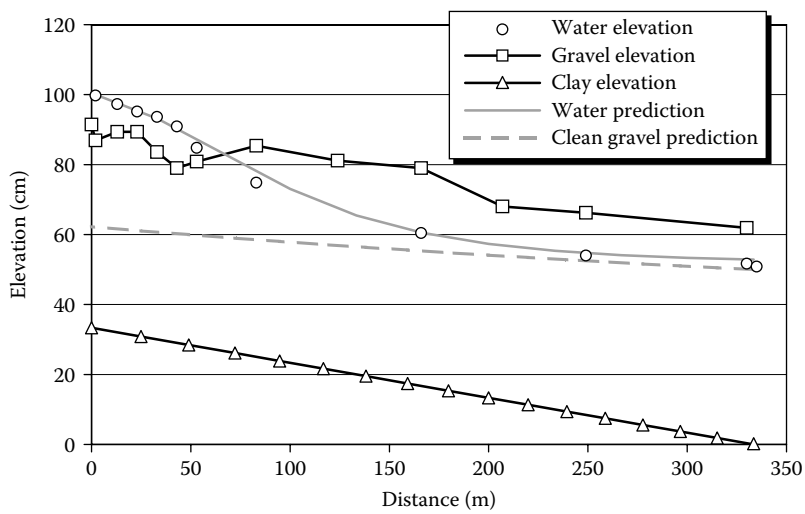



**FIGURE 2.26** Water mounding due to clogging in a HSSF wetland. At the time of the study, the inlet section of the bed was clogged significantly. The hydraulic conductivity increased from 3,070 m/d in the inlet region, to 26,400 m/d in the outlet region. The clean gravel prediction is based on a laboratory measurement of $k$ = 44,000 m/d. (Data from Kadlec and Watson (1993) In *Constructed Wetlands for Water Quality Improvement*. Moshiri (Ed.), Lewis Publishers, Boca Raton, Florida, pp. 227–235; graph from Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

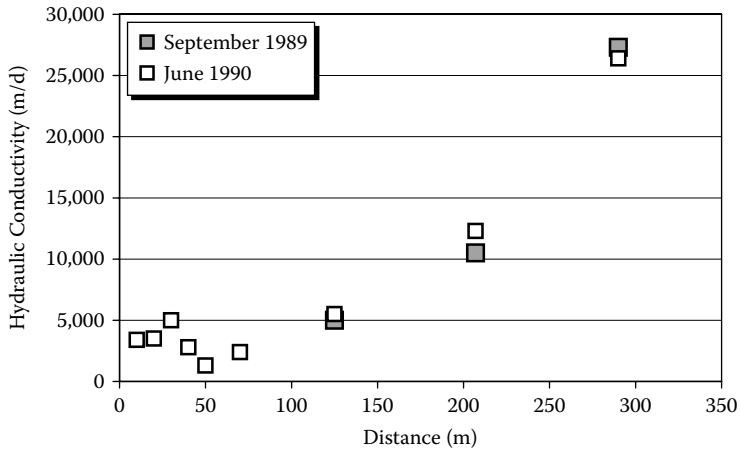



**FIGURE 2.27** Hydraulic conductivity of the crushed stone media in operation two and three years at Benton, Kentucky. (From Kadlec and Watson (1993). In *Constructed Wetlands for Water Quality Improvement*. Moshiri (Ed.), Lewis Publishers, Boca Raton, Florida, pp. 227–235. Reprinted with permission.)

### Flow Stratification

Several factors create vertical stratification of hydraulic conductivity in HSSF beds, and therefore lead to vertical profiles of water flow rates. There have been anecdotal reports of plant root penetration of up to 60 cm (Gersberg *et al.*, 1986), but several subsequent studies have found lesser rooting depths (Daniels and Parr, 1990; Parr, 1990; Pilgrim *et al.*, 1992; Adcock and Ganf, 1994; Kuusemets *et al.*, 2002). For *Phragmites*, in excess of 90% of the biomass of roots and rhizomes are in the upper 30 cm of the media (Figure 2.29). Other species follow similar patterns, but may have somewhat different mat thicknesses. For instance, *Schoenoplectus* (*Scirpus*) *validus* rooted to a thickness of about 20 cm at Benton, Kentucky (Figure 2.30); *Schoenoplecctus* (*Scirpus*) *sylvaticus* to 10 cm at Kodijärve, Estonia; *Typha* to 20 cm at Byron bay, Australia; and *Baumea articulata* to 30 cm at Adelaide, Australia. Laboratory evidence indicates that the depth of root penetration is strongly influenced by the redox potential within the wetland bed. Strongly reducing and nutrient-rich conditions are associated with shallower root penetration and less root biomass (Lockhart, 1999; Wallace and Knight, 2006). Since many HSSF wetlands are used in conjunction with primary-treated effluents, the resulting conditions in the wetland bed limit root penetration.

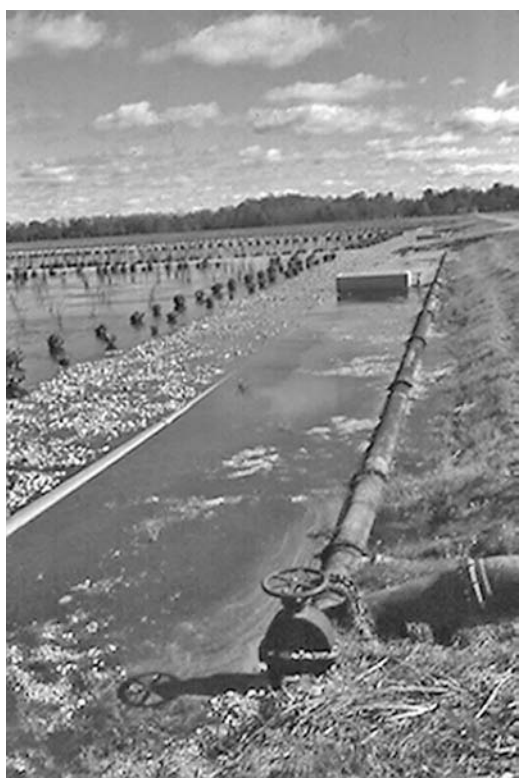


**FIGURE 2.28** The flooded inlet zone of the Denham Springs, Louisiana, HSSF wetland.

Cumulative experience with HSSF wetlands indicates that deeper gravel beds (>40 cm) will contain an upper zone that contains essentially all of the plant roots and a lower zone without roots. The presence of root blockage is an important factor: the root zone impedes flow more than the bare media below it. Several tracer studies have documented this phenomenon (Fisher, 1990; Pilgrim *et al.*, 1992; Marsteiner *et al.*, 1996; García, 2003).

There is considerable evidence that accumulated solids selectively occupy different layers in the media (Kadlec and Watson, 1993; Sanford *et al.*, 1995; Tanner *et al.*, 1998a). No common pattern has been found for the vertical profiles, with some studies showing more solids in the bottom (Kadlec and Watson, 1993; Tanner *et al.*, 1998a), while others show more solids near the surface (Tanner and Sukias, 1995). Deposition of solids within the HSSF bed is likely dependent on the suspended solids loading and the physical characteristics of the sediments (specific gravity, particle size, and biodegradability).

Finally, it has been found that density-induced stratification can occur in FWS wetlands (Stephan *et al.*, 2004) and also in HSSF beds (Rash and Liehr, 1999; Kadlec *et al.*, 2003).

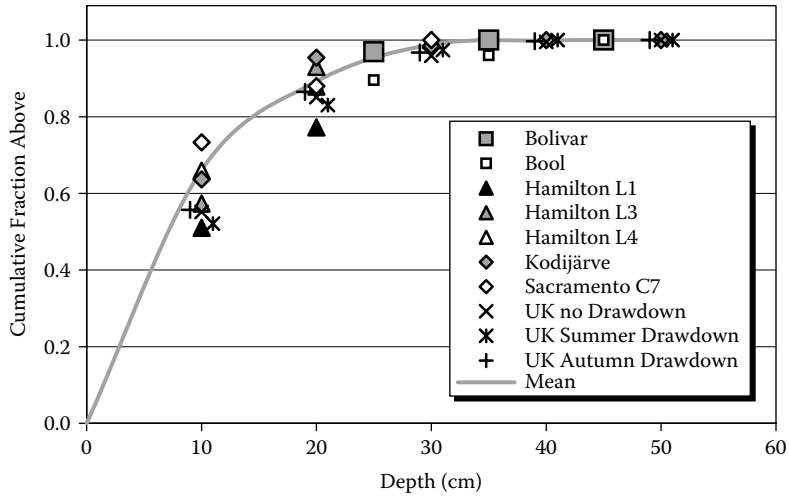



**FIGURE 2.29** Depth profiles of roots and rhizomes for SSF Phragmites beds. (Data from Daniels and Parr (1990) *1989 Survey of reed growth in UK reed bed treatment systems*. WRc/NERC Contract Report T02058F1, The Institute of Terrestrial Ecology: Dorset, United Kingdom; Adcock and Ganf (1994) *Water Science and Technology* **29**(4): 95–102; Kuusemets ***et al.*** (2002) *Nitrogen and phosphorus assimilation and biomass production by* Scirpus sylvaticus *and* Phragmites australis *in a horizontal subsurface flow constructed wetland*. Mbwette (Ed.). Proceedings of the 8th International Conference on Wetland Systems for Water Pollution Control, 16–19 September 2002; Comprint International Limited: University of Dar Es Salaam, Tanzania, pp. 930–937.)

This phenomenon is caused by water of moderate salt content (higher density) moving along the wetland bottom, and rainwater of lesser salt content (lower density) moving in the upper strata of the bed. Such vertical density gradients are very stable, and persist for months (Kadlec *et al.*, 2003), perhaps indefinitely (Nivala, 2005) unless operational steps are taken to address the hydraulic short-circuiting that results from vertical stratification.



**FIGURE 2.30** Roots of *Scirpus validus* in the Benton, Kentucky, HSSF wetland. Scale is in inches. The major portion of the root mat is contained in the top 20 cm.

## 2.4 VF WETLAND HYDRAULICS

Subsurface flow systems may be operated in vertical flow mode, but that mode has many variants. These include:

1. *Intermittent downflow.* This option involves flood application of water on top of the bed for brief periods of time. This operational mode is selected to enhance oxygen transport into the bed. This type is favored in many European countries. It was advanced as part of the original Max Planck Institute system developed in the 1960s (Seidel, 1966). When no plants are used, these are termed intermittent sand filters (Liénard *et al.*, 2001).
2. *Unsaturated downflow.* This variant involves distributing water across the top of a granular media. Water then trickles through the media in unsaturated flow. Distribution pipes may be located above the system, or, in cold climates, buried within the granular media bed. The system may be configured in a single-pass mode or, more commonly, employ flow recirculation so that wastewater passes through the media bed multiple times. These systems are functionally equivalent to recirculating sand or gravel filters (Crites and Tchobanoglous, 1998; Crites *et al.*, 2006). Systems with very low hydraulic loading rates may be unlined. If the primary intent is to harvest a crop, these systems are called slow-rate infiltration systems (Nichols and Boelter, 1982; Water Environment Federation, 2001).
3. *Saturated up- or downflow.* These systems employ continuous saturated flow of water through the

plant root zone. Downflow configurations are used in mine water treatment, where they are termed an anaerobic wetland or alkalinity producing system (Younger *et al.*, 2002). Aerated downflow systems have been employed as polishing reactors for removal of ammonia (Wallace *et al.*, 2006a). Saturated upflow is desirable when the daylighting water must be of highest quality, to minimize contact with contaminants, or root zone contact is to be maximized (Heritage *et al.*, 1995; Tanner *et al.*, 2002a). These systems have been employed in the laboratory as anaerobic reactors to provide reductive dehalogenation of chlorinated solvents (Pardue *et al.*, 2000; Kassenga *et al.*, 2003).

4. *Tidal flow (fill and drain).* These systems employ the cycling filling and draining of a granular bed. During the fill portion of the cycle, the wastewater is fed into the bottom of the wetland bed. Flow moves upwards, gradually filling the bed. Fill is complete when the surface is flooded. The pump is then stopped; the wastewater is then held in the bed in contact with the bacteria growing on the media. After a holding period, the wastewater is drained, and air enters the voids in the bed. These reactors create cycling redox conditions that contain both oxidizing and reducing phases (Maciolek and Austin, 2006). The fill and drain frequency depends on the application, but is typically about two hours in length (Sun *et al.*, 1999). Tidal flow wetlands may be run in parallel pairs, one filling while the other is draining. This mode has been termed reciprocating operation (Behrends, 2000). Flows are a combination of horizontal and vertical during filling, but mostly vertical downward during draining.

Other types of operation have also been tested; for instance, the subsurface introduction of wastewater into saline groundwaters. The density difference creates an upward buoyancy driving force, causing vertical upward flow (Watson and Rusch, 2001; Richardson *et al.*, 2004).

The basis for analysis of vertical flow systems is also the proposition of Darcy, developed in 1856 (Brown, 2002) for saturated flow. In combination with the water mass balance, it is easily extended to the more general case of unsaturated flow (Richards' equation, Freeze and Cherry, 1979). The one-dimensional dynamic version is:

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial x}\left[ K \frac{\partial h}{\partial x} \right] \tag{2.55}$$

where

$h$ = pressure head, m
$K$ = *unsaturated* hydraulic conductivity, m/d
$t$ = time, d
$x$ = vertical distance, m
$\theta$ = fractional water content, $m^3/m^3$

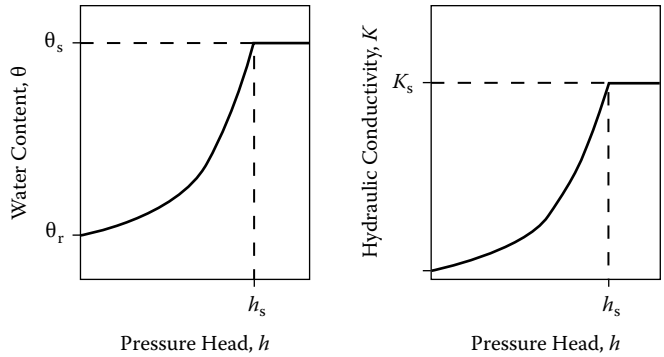



**FIGURE 2.31** Both water content and hydraulic conductivity drop off markedly with decreasing pressure head for a typical porous substrate. At full capacity, the water content is $\theta_s$, with the entire porosity filled. Some portion ($\theta_r$) of the porosity is undrainable. In unsaturated conditions, the hydraulic conductivity decreases with decreasing water content, to only a small fraction of the saturated conductivity.

The complicating feature is that both the water content and the hydraulic conductivity are now functions of the pressure head:

$$\theta = \theta(h) \quad \text{and} \quad K = K(h) \tag{2.56}$$

These functional forms have been well-studied. In clean media, they are typically *s*-shaped curves (Figure 2.31). Dynamic computations of the flows and water contents under intermittent flow in vertical flow wetlands are possible, but require nontrivial numerical procedures (Schwager and Boller, 1997; Langergraber, 2001). Well-established computer programs are available for this purpose, such as HYDRUS-1D and HYDRUS-2D (Simunek *et al.*, 1998; 1999).

Here the results of these calculations are illustrated, to gain an appreciation of the course of a cycle of flooding and draining for a typical intermittent vertical downflow operation.

### Intermittent Downflow Beds

Vertical flow beds often consist of layers of porous media, with the bottom-most layer consisting of coarse media with a network of perforated drainage pipes (Cooper *et al.*, 1996). An example of such layering used in Austria is shown in Figure 2.32 (ÖNORM B 2505, 1997). This bottom layer is freely draining, perhaps with a permanent pool level slightly above the bed bottom. At the continuous, low average hydraulic loading rates usually employed in treatment, a sand or gravel medium will not become saturated; and the flow will be percolation through voids partially filled with air. However, if the water is delivered in a short period of time, the instantaneous loading rate may exceed the drainage rate, and the media will then fill with water.

The sequence of events during a cycle of fast flooding and draining is conceptually straightforward. A dose of water to be treated is introduced as a flood on the bed surface, with up to six doses per day (Figure 2.33). A cycle begins with

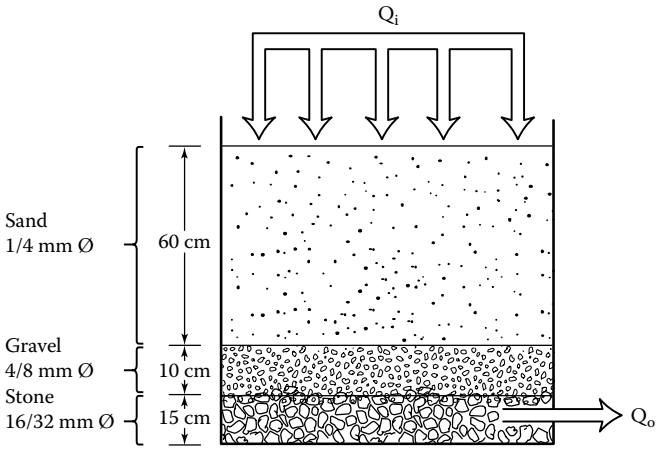



**FIGURE 2.32** Vertical layering of an Austrian design for vertical flow constructed wetlands.

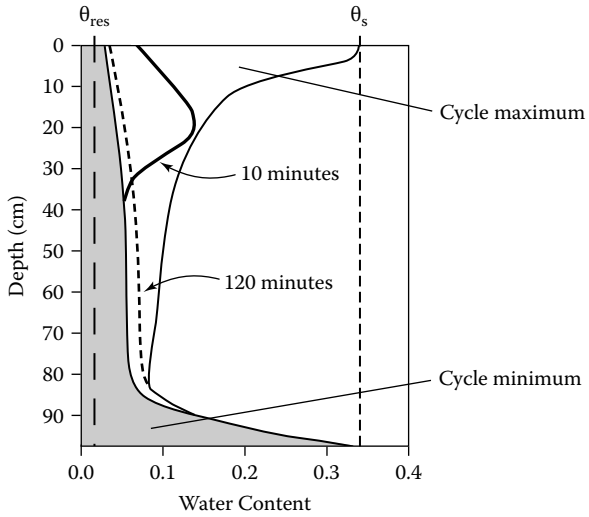



**FIGURE 2.34** Water content profiles of a dosed vertical flow system. The cycle time was four hours (From Schwager and Boller (1997) *Water Science and Technology* 35(6): 13–20. Reprinted with permission.)

a mostly drained bed, containing water at or just above the residual water content of the media (Figure 2.34). The duration of the water introduction is variable, ranging from less than ten minutes (Schwager and Boller, 1997) to an hour or two (Watson and Danzig, 1993). For illustration, suppose the flood is very brief. The effect is then to create saturated conditions in the top layer of the bed, and perhaps surface ponding. The air in the voids is then trapped, and may be compressed by the water above if there is no air relief mechanism in the bed, such as a vent pipe. The period of air entrapment is typically brief, but may last for up to 45 minutes for large dosing pond depths (Schwager and Boller, 1997). Air bubbles can form, venting a portion of the trapped antecedent air (Figure 2.35).

After the air-lock is broken, drainage proceeds as unsaturated flow. Air enters the pores on top of the bed to replace the draining water volume in the bed voids. Air movement in the lower portions of the bed is minimal. Drainage in many VF systems is complete well before the start of the next cycle (Figure 2.34). During the fully drained portion of the period, air moves into the voids as determined by oxygen consumption and diffusion. Thus it is seen that the air deep in the bed has been convected to that location, while air in the upper layers of the bed has been supplied by diffusion (Kayser and Kunst, 2005).

The result of this cyclic operation is a variable outflow from the system. There is a rising outflow for a brief period, followed by a declining outflow (Watson and Danzig, 1993; Langergraber, 2001; Kayser and Kunst, 2005; Dittmer *et al.*, 2005) (Figure 2.36).

A period of resting, after full drainage, is typically included to allow for the oxidation of accumulated organics in and on the top of the bed, to avoid clogging.



**FIGURE 2.33** The flooding of the vertical intermittent sand filters at the Saginaw, Michigan, landfill leachate treatment system. Water is dosed twice per day, with each flooding lasting only a few minutes. Rip-rap splash pads prevent erosion for the three-point distribution.

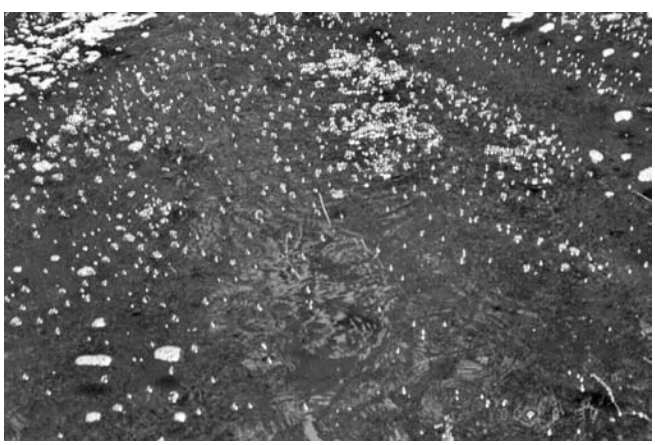


**FIGURE 2.35** Trapped air bubbles up through the ponded water just after dosing at the Saginaw, Michigan, site.

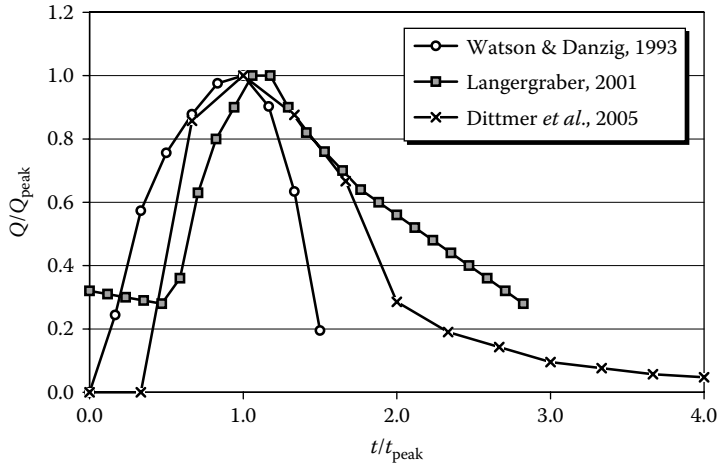



**FIGURE 2.36** Outflow time series for vertical flow, dosed wetlands. (Data from Watson and Danzig (1993). In *Constructed Wetlands for Water Quality Improvement*, Moshiri (Ed.), CRC Press, Boca Raton, Florida, pp. 301–313; Langergraber (2001) Development of a simulation tool for subsurface flow constructed wetlands. Ph.D. Dissertation, IWGA SIG University of Vienna; Dittmer *et al.* (2005) *Water Science and Technology* 51(9): 225–232.)

There are many variants on this simple scheme. For instance, Green *et al.* (1997a; 1998) proposed that water should be accumulated in the bottom of the bed, and discharged rapidly, to induce air entry during the sudden drainage. During the refilling portion of the cycle, "used" air is vented through a perforated pipe located in the media.

Brix and Schierup (1990) provide explanation of a long-term dose and drain hydraulic operation. In their example, the bed is loaded at a constant rate for two days, followed by eight days of draining. The conditions for ponding may be expressed in terms of hydraulic loading:

$$q \geq k \tag{2.57}$$

where

$k$ = saturated hydraulic conductivity of the media, m/d
$q$ = instantaneous hydraulic loading rate, m/d

At lower loading rates, $q \leq K$, the bed can transport all water under the influence of the vertical pressure gradient of a fully saturated bed. During the relatively long period of water addition, the ponding depth increases exponentially, as does the drainage rate. After the cessation of water addition, the ponded water drains, after which the interstitial water drains. Depending on the hydraulic conductivity of the bed, drainage takes 0.5–10 days.

This same long-cycle operation was adopted in the project described by Kadlec *et al.* (1997) and Burgoon *et al.* (1999), only with loadings less than the hydraulic conductivity of the bed. Therefore, a period of unsaturated vertical flow was followed by a period of some days of draining and resting. Morris (1999) dosed intermittently at 3–10 near-instantaneous doses per day for two days, followed by six days' resting.

Empirical equations have been developed to describe the infiltration rates for various media and organic layers for systems designed and operated according to French criteria (Molle *et al.*, 2006). These are heavily loaded, dosed vertical beds. However, site conditions and operating strategies are quite variable, and it is thus not feasible to develop such descriptions for a general case.

### Vertical Flow Tracer Tests

When an inert tracer is added to an intermittently dosed system, the response curves follow the general bell-shaped form that is seen for continuous flow systems, but with slight deflections during the course of each cycle (Schwager and Boller, 1997; Tanner *et al.*, 2002a). For intermittent vertical downward flows, the results of Schwager and Boller (1997) showed a gamma function response, with $N = 5$, and a tracer detention time of about 18 hours. Dosing was every four hours, and the average loading was 12 cm/d on a 90-cm-deep bed. The saturated water volume was about 35%, for a nominal saturated detention time of 2.5 days. However, the bed was far from saturated throughout most of flow (see Figure 2.34), and thus the detention time was much less than for saturated flow. Interestingly, the effect of clogging over time in this mode of operation was to cause an increase in detention time. This was due to the increased water holdup in the "used" bed, although the bed was still not saturated during flow for the clogged condition.

The vertical upflow, saturated system of Tanner *et al.* (2002a) provided an entirely different hydraulic environment. Water was displaced upward in each of five tanks in series, with the overflow from each being piped to the bottom of the next. Dosing was on a six-hour schedule, with about 2 L per dose. Each tank had a clean pore volume of about 8 L, thus the detention time was nominally five days. Four such cascades were operated, two with high-strength meat and dairy wastewaters, and two with pretreated dairy waters. The tracer detention time distributions all showed gamma

distributions, with approximately two "tanks" per each of the five cells in series. However, the two cascades with strong influents displayed (measured) pore blockage of 46–64%, while the pretreated waters caused only half that amount of blockage. This reduction in pore volume caused the detention times to be shortened considerably compared to the nominal clean-volume calculated detention times.

Rogers *et al.* (1990) performed tracer tests on mesocosms in both an upflow and a downflow mode of operation, both batch-dosed twice per day at a hydraulic loading of 2.5 cm/d. The media was fully saturated at all times, with outflow taken as an overflow. They found major differences in root distribution, and concluded that the downflow mode was more akin to plug flow because of a surficial root mat.

## Clogging

Clogging is a well-known phenomenon in sand filtration (Woodward and Ta, 1988). Platzer and Mauch (1997) conducted a literature survey in the context of vertical flow wetlands, and identified three potential mechanisms, to which one may add the presence of roots (Winter and Goetz, 2003):

1. Deposition and filtration of incoming particulates, leading to blockage of pores, especially near the surface
2. Biomass production in the soil pores, otherwise known as biomat formation, due to the favorable conditions created by domestic or municipal wastewater
3. Chemical precipitation in the pores, for example, calcium carbonate
4. The presence of roots

The third mechanism may be of concern in mine water treatment, but would otherwise not be expected to contribute. Inorganic materials accumulated via Mechanisms #1 and #2 are expected to remain in the pores, and eventually create blockage. However, organic materials are subject to oxidation, especially during the resting portion of a cycle, and are therefore removed at some speed determined by decomposition processes. Smaller grain sizes contribute to more rapid clogging. Roots and biofilms are anticipated to block only a small fraction of the pores (Langergraber *et al.*, 2003). We are then left with accumulation of solids as the principal mechanism of clogging.

### The Effect of TSS on Clogging

As a first approximation, solids accumulate in pores as the result of complete filtration of incoming water:

$$M = q\,C_i \tag{2.58}$$

where

$C_i$ = inlet TSS concentration, g/m$^3$
$M$ = mass accumulation rate in pores, g/m$^2$·d
$q$ = hydraulic loading rate, m/d

The result of this blockage is to increase the headloss required to drive the (constant) flow $q$ through the bed. In constant flow potable water treatment, the head loss is inversely proportional to the remaining free area for flow (Woodward and Ta, 1988):

$$L = \frac{L_o}{1 - C_i M t} \tag{2.59}$$

where

$C_i$ = inlet TSS concentration, g/m$^3$
$L$ = headloss at time $t$, m
$L_o$ = headloss at time zero, m
$t$ = time, days

This simple relation does a creditable job of explaining operation of potable water sand filters (Woodward and Ta, 1988), and the concept has been adopted for vertical flow wetlands by Blazejewski and Murat-Blazejewska (1997), Langergraber *et al.* (2003), and Zhao *et al.* (2004). Of interest for the wetland application is the time to clogging, which is determined as a volume of accumulated solids:

$$t_{clog} = a\,\frac{\rho_{solid}}{q C_i} \tag{2.60}$$

where

$a$ = empirical coefficient, m
$t_{clog}$ = clogging time, days
$\rho_{solid}$ = bulk density of accumulating solids, kg/m$^3$

Langergraber *et al.* (2003) suggest $a$ = 0.18 m. Their lab data strongly support Equation 2.60. Blazejewski and Murat-Blazejewska (1997) propose:

$$a = 150\varepsilon \cdot d \tag{2.61}$$

where

$d$ = particle diameter, m
$\varepsilon$ = clean porosity, m$^3$/m$^3$

Equation 2.61 is based upon a clogging depth that is proportional to particle diameter. A good fit to the data of Bavor and Schulz (1993) was found. For very strong wastes, the clogging times, both real and predicted by Equation 2.60, are quite short — a matter of a few days (Zhao *et al.*, 2004).

The value of the coefficient $a$ is very likely to depend upon the size distribution of particles and other bed properties. However, these studies have verified that Equation 2.60 provides a reasonable approximation to clogging due to filtration. The results from a number of VF wetlands suggest a sustained TSS loading of about 5 g/m$^2$·d is all that can be tolerated, but if there is adequate recovery in resting periods, then much higher rates can be sustained.

### The Effect of Organic Content on Clogging

One concept of clogging argues that stronger influents (more BOD or COD) should promote more biofilm growth within the bed, and hence contribute to clogging. Blazejewski and

Murat-Blazejewska (1997) assume that biofilm growth and decomposition are in balance and do not contribute to clogging. Langergraber *et al.* (2003) conclude that biomass growth plays only a minor role compared to suspended solids over the short term. Winter and Goetz (2003) note that TSS and COD are often strongly correlated in the water to be treated, and that it is therefore difficult to sort out the possibilities.

The idea of organic materials contributing a major amount of blockage finds strong support when resting periods are included in the cycle. As noted by Platzer and Mauch (1997), the original conductivity of a bed may often be restored by allowing several days rest. Presumptively, this aerobic resting period causes oxidation of organics, thus freeing pore volume again. Platzer and Mauch (1997) reported a linear decrease in bed conductivity with increasing COD loading, but it is likely that TSS loading also increased. As a result of these uncertainties, a maximum COD loading has been deemed prudent. Platzer and Mauch (1997) suggest 25 $g/m^2 \cdot d$, and Winter and Goetz (2003) suggest 20 $g/m^2 \cdot d$.

Interestingly, there appears to be a second viable operating range, with very high TSS and COD loadings (Molle *et al.*, 2006). The TSS forms a mat on top of the bed, to depths in excess of 20 cm of organic material. This apparently acts as a trap for most incoming TSS, and spares the underlying bed from clogging. Up to 250 $g/m^2 \cdot d$ of COD (30–60 $g/m^2 \cdot d$ of BOD), and 20–50 $g/m^2 \cdot d$ TSS, have been sustainably treated (Chazarenc and Merlin, 2005). Accumulated solids form a compost layer on top of the original bed, with amounts of 20–90 $kg/m^2$ accruing after several years (Chazarenc and Merlin, 2005). This accretion is an effective mulch layer, which also aids in treatment. The surface water ponding that accompanies this mode of operation is problematical for single-home onsite treatment systems in North America, as these septic system codes typically require no daylighting of raw wastewater.

## SUMMARY

This chapter has presented a synthesis of tools necessary to predict water budgets and hydraulics in treatment wetlands. Hydraulic processes in FWS and HSSF wetlands are similar in many respects, but there are significant differences. Adequate prediction methods are critical for treatment wetlands design and successful operation.

Wetland water budgets are dominated by surface inflows and outflows, evapotranspiration, and precipitation. Groundwater interactions are normally slight. Surface flows are generally measurable with sufficient precision. Precipitation may be projected from historical weather data, with the possibility of some error due to changing climatic conditions. Stochastic variability is large, however, on several times scales of interest. Atmospheric additions and losses are predictable by several techniques described in this chapter. In lightly loaded wetlands in warm seasons, this contribution may be very important in design calculations, so methods are presented for modifying pollutant reduction computations.

The internal water budget, or mass balance, for a treatment wetland is required for both conveyance calculations and pollutant reduction models. These equations, which have been detailed for FWS and HSSF wetlands, allow calculations of water depths and elevations and flow rates at interior points in the treatment wetland. Head losses in FWS wetlands have sometimes caused operational problems, and have often caused such difficulties in HSSF systems. Procedures for estimating frictional effects in both types of wetlands have been presented, along with shortcut methods for estimating the necessary design parameters to ensure adequate conveyance.

VF wetlands have been operated in both saturated and unsaturated flow regimes. For vertical saturated flow, many of the concepts outlined for HSSF wetlands directly apply. Pulse loading and associated unsaturated flow has also been discussed in this chapter.

# 3 Treatment Wetland Vegetation

There are many general functions of vegetation in wetlands. Physical functions include transpiration, flow resistance, and particulate trapping, all of which are related to vegetation type and density. Ecological functions include wildlife habitat and human use values. The focus here is water quality and, in particular, the processing of potential pollutants.

There are many effects vegetation can have on chemical processing and removal in treatment wetlands. These may include:

1. The plant growth cycle seasonally stores and releases nutrients, thus providing a "flywheel" effect for a nutrient removal time series.
2. The creation of new, stable residuals accrete in the wetland. These residuals contain chemicals as part of their structure or in absorbed form, and hence accretion represents a burial process for nitrogen.
3. Submersed litter and stems provide surfaces on which microbes reside. These include nitrifiers and denitrifiers, and other microbes that contribute to chemical processing.
4. The presence of vegetation influences the supply of oxygen to the water. Emergent vegetation blocks the wind, and shades out algae, presumably lowering reaeration. Floating vegetation may provide a barrier to atmospheric oxygen transfer. Submerged vegetation may provide photosynthetic oxygen supply directly in the water. To some limited extent, plant oxygen flux supplies protective oxidation in the immediate vicinity of plant roots.
5. The carbon content of plant litter supplies the energy need for heterotrophic denitrifiers.

Plants that occur in natural wetlands are described in many guidebooks and reference collections. They may be categorized by their growth habit with respect to the wetland water surface as:

- Emergent soft tissue plants
- Emergent woody plants
- Submersed aquatic plants
- Floating plants
- Floating mats

Obviously, only the first two categories may be implemented in SSF wetlands, whereas all five are candidates for FWS systems. The emphasis of treatment wetland technology to date has been on soft tissue emergents, including *Phragmites*, *Typha,* and *Schoenoplectus* (*Scirpus*).

Plant selection and establishment for constructed wetlands is covered in Chapters 18 and 21. The topic of biodiversity is covered in Chapter 19. In this chapter, plant species and examples of their usage are described. It is not the intent to provide full botanical specifications, but rather to acquaint the reader with the wide variety of choices of vegetation that have been implemented, and the sources of information that form the botanical foundation of treatment wetlands.

Because of the presence of ample water, wetlands are typically home to a variety of microbial and plant species. The diversity of physical and chemical niches present in wetlands results in a continuum of life forms from the smallest viruses to the largest trees. This biological diversity creates interspecific interactions, resulting in greater diversity, more complete utilization of energy inflows, and ultimately to the treatment properties of the wetlands ecosystem.

The study of organisms and their populations is a convenient way to catalog these life forms into groups with general similarities. However, the genetic and functional responses of wetland organisms are essentially limitless and result in the ability of natural systems to adapt to changing environmental conditions such as the addition of wastewaters. Genetic diversity and functional adaptation allow living organisms to use the constituents in wastewaters for their growth and reproduction. In using these constituents, wetland organisms mediate physical, chemical, and biological transformations of pollutants and modify water quality. In wetlands engineered for water treatment, design is based on the sustainable functions of organisms that provide the desired transformations.

The wetland treatment system designer should not expect to maintain a system with just a few known species. Such attempts frequently fail because of the natural diversity of competitive species and the resulting high management cost associated with eliminating competition, or because of imprecise knowledge of all the physical and chemical requirements of even a few species. Rather, the successful wetland designer creates the gross environmental conditions suitable for groups or guilds of species; seeds the wetland with diversity by planting multiple species, using soil seed banks and inoculating from other similar wetlands; and then uses a minimum of external control to guide wetland development. This form of ecological engineering results in lower initial cost, lower operation and maintenance costs, and most consistent system performance.

This chapter presents an overview of the floristic diversity that naturally develops in treatment wetlands as well as some details of the community types that may be fostered in wetland treatment systems. These microbial and plant

species are typically the dominant structural and functional components in treatment wetlands. An understanding of their basic ecology will provide the wetland design or operator with insight into the mechanics of their "green" wastewater treatment unit.

Information about wetland plant species is voluminous and available from multiple sources. For more detailed information on aquatic and wetland microbial communities the reader is referred to Portier and Palmer (1989), Pennak (1978), or Wetzel (2001). For more detailed information on the ecology of the vascular plant species found in wetlands, the reader is referred to Hutchinson (1975), Sainty and Jacobs (1981), Brock *et al*. (1994), Reddington (1994), Cook (1996; 2004), Mitsch and Gosselink (2000a), or Cronk and Fennessy (2001). There are also multiple regional guides for the nonbotanist, for instance, for the northern United States:

*Through the Looking Glass: A Field Guide to the Aquatic Plants*. S. Borman, R. Korth, and J. Temte, 1997. Wisconsin Department of Natural Resources Publication No. FH-207-97, University of Wisconsin Extension, Stevens Point, Wisconsin.

*National List of Plant Species That Occur in Wetlands for USFWS Region 3 (MI, IN, IL, MO, IA, WI, MN), A Field Guide*. Resource Management Group, Inc., 1992. Prepared by Resource Management Group, Inc., Grand Haven, Michigan.

*A Naturalist's Guide to Wetland Plants: An Ecology for Eastern North America*. D.D. Cox, 2002. Syracuse University Press, Syracuse, New York.

*A Field Guide to Wetland Characterization and Wetland Plant Guide: A Non-Technical Approach*. K. Pritchard, 1991. Washington State University, Cooperative Extension Service, Seattle, Washington.

As another example source, the University of Florida Institute of Food and Agricultural Services maintains the Aquatic, Wetland, and Invasive Plant Information and Retrieval System (APIRS). Available are videos, line drawings, identification decks of color photos, and searches of a 50,000-record database (http://plants.ifas.ufl.edu). Thus, the practitioner can easily find scientific and common names, and gain an appreciation for what the plant looks like and its habitat requirements. We are therefore not reproducing this information here.

## 3.1 ECOLOGY OF WETLAND FLORA

### Wetland Bacteria and Fungi

Wetland and aquatic habitats provide suitable environmental conditions for the growth and reproduction of microscopic organisms. Two important groups of these microbial organisms are bacteria and fungi. These organisms are important in wetland treatment systems primarily because of their role in the assimilation, transformation, and recycling of chemical constituents present in various wastewaters. Bacteria and fungi are typically the first organisms to colonize and begin the sequential decomposition of solids in wastewaters (Gaur *et al*., 1992). Also, microbes typically have first access to dissolved constituents in wastewater and either accomplish sorption and transformation of these constituents directly or live symbiotically with other plants and animals by capturing dissolved elements and making them accessible to their symbionts or hosts.

The taxonomy of microbes is complex and frequently revised, but the general groups of bacteria and fungi are commonly recognized. Bacteria are classified in the Procaryotae (Buchanan and Gibbons, 1974). Procaryotes are distinguished by their lack of a defined nucleus with nucleaic material present in the cytoplasm in a nuclear region. Cyanobacteria or blue-green algae are also classified as procaryotes, but they are discussed with algae below. Fungi are classified as eucaryotes because they have a nucleus separated from the cytoplasm by a nuclear membrane.

#### Bacteria

Bacteria are unicellular, procaryotic organisms classified by their morphology, chemical staining characteristics, nutrition, and metabolism. *Bergey's Manual* (Buchanan and Gibbons, 1974) places bacteria into 19 associated groups with unclear evolutionary relationships. Most bacteria can be classified into four morphological shapes: coccoid or spherical, bacillus or rodlike, spirillum or spiral, and filamentous. These organisms may grow singly or in associated groups of cells including pairs, chains, and colonies. Bacteria typically reproduce by binary fission, in which cells divide into two equal daughter cells. Most bacteria are heterotrophic, which means they obtain their nutrition and energy requirements for growth from organic compounds. In addition, some autotrophic bacteria synthesize organic molecules from inorganic carbon (carbon dioxide, $CO_2$). Some bacteria are sessile while others are motile by use of flagella. In wetlands, most bacteria are associated with solid surfaces of plants, decaying organic matter, and soils.

#### Fungi

Fungi represent a separate kingdom of eucaryotic organisms and include yeasts, molds, and fleshy fungi. All fungi are heterotrophic and obtain their energy and carbon requirements from organic matter. Most fungal nutrition is saprophytic, which means it is based on the degradation of dead organic matter. Fungi are abundant in wetland environments and play an important role in water quality treatment. For general information about fungi, see Ainesworth *et al*. (1973).

Fungi are ecologically important in wetlands because they mediate a significant proportion of the recycling of carbon and other nutrients in wetland and aquatic environments. Aquatic fungi typically colonize niches on decaying vegetation made available following completion of bacterial use. Saprophytic fungal growth conditions dead organic matter for ingestion and further degradation by larger consumers.

Fungi live symbiotically with species of algae (lichens) and higher plants (mycorhizzae), increasing their host's efficiency for sorption of nutrients from air, water, and soil. If fungi are inhibited through the action of toxic metals and other chemicals in the wetland environment, nutrient cycling of scarce nutrients may be reduced, greatly limiting primary productivity of algae and higher plants. In wetlands, fungi are typically found growing in association with dead and decaying plant litter.

## Microbial Metabolism

Microbes are involved in a large proportion of wetland transformations and removals. In many cases, there are several interconnected steps and organisms. The reader is referred to Maier *et al.* (2000) for an introduction to environmental microbial processes. Most of the important chemical transformations conducted by microbes are controlled by enzymes, genetically-specific proteins that catalyze chemical reactions. To a varying extent, bacteria and fungi are classified by their ability to catalyze certain reactions. Microbial metabolism includes the use of enzymes to break apart complex organic compounds into simpler compounds with the release of energy (catabolism) or the synthesis of organic compounds (anabolism) by the use of chemically stored energy. Microbial metabolism not only depends on the presence of appropriate enzymes but also on environmental conditions such as temperature, dissolved oxygen (DO), and hydrogen ion concentration (pH). Also, the concentration of the chemical substrate undergoing the transformation is of primary importance in determining reaction rate.

Microbes can be classified by their metabolic requirements. Photoautotrophic bacteria such as the green and purple sulfur bacteria use light as an energy source to synthesize organic compounds from $CO_2$. Reduced sulfur compounds such as hydrogen sulfide or elemental sulfur serve as electron acceptors in oxidation-reduction reactions. Photoheterotrophs use light as an energy source and organic carbon as a carbon source for cell synthesis. The organic carbon sources most typically used by photoheterotrophs are alcohols, fatty acids, other organic acids, and carbohydrates. Because photosynthetic bacteria do not use water to reduce $CO_2$, they do not produce $O_2$ as a byproduct of metabolism, as do the algae and higher plants.

Chemoautotrophic bacteria derive their energy from the oxidation of reduced inorganic chemicals and use $CO_2$ as a source of carbon for cell synthesis. A number of the bacteria which are important in wetland treatment of wastewater are chemoautotrophs. Bacteria in the genus *Nitrosomonas* oxidize ammonia nitrogen to nitrite, and *Nitrobacter* oxidize nitrite to nitrate, deriving energy, which is used in cell metabolism (see Chapter 9). The genus *Beggiatoa* derives energy from the oxidation of $H_2S$, *Thiobacillus* oxidizes elemental sulfur and ferrous iron, and *Pseudomonas* oxidizes hydrogen gas (see Chapter 11). Chemoheterotrophs derive energy from organic compounds and also use the same or other organic compounds for cell synthesis. Most bacteria, and all fungi, protozoans, and higher animals are chemoheterotrophs.

During microbial metabolism, carbohydrates are broken into pyruvic acid with the net production of two pyruvic acid molecules and two adenosine triphosphate (ATP) molecules for each molecule of glucose and the subsequent decomposition of pyruvic acid through fermentation or respiration. Fermentation by substrate-level phosphorylation does not require oxygen and results in the formation of a variety of organic end products such as lactic acid, ethanol, and other organic acids.

Aerobic respiration is the process of biochemical reactions by which carbohydrates are decomposed to $CO_2$, water, and energy (38 ATP molecules for each glucose molecule fully oxidized). The Krebs Cycle results in the loss of carbon dioxide (decarboxylation) and energy storage (two molecules of ATP per molecule of glucose). For complete oxidation to occur, oxygen and hydrogen ions must be available as the final electron acceptor in a chain of reactions called the electron transport chain. The overall reaction for aerobic respiration can be summarized as follows:

$$C_6H_{12}O_6 + 6H_2O + 6O_2 + 38\ ADP + 38\ P = 6CO_2 + 12H_2O + 38\ ATP \quad (3.1)$$

Also, approximately 60% of the energy of the original glucose molecule is lost as heat during the complete aerobic respiration process.

Anaerobic respiration is an alternative catabolic process that occurs in the absence of free oxygen gas. In anaerobic respiration, some other inorganic compound is used as the final electron acceptor. A variable and lower amount of energy is derived during the process of anaerobic respiration. This form of respiration is important to several groups of bacteria which occur in wetlands and aquatic habitats. Bacteria in the genera *Pseudomonas* and *Bacillus* use nitrate nitrogen as the final electron acceptor, producing nitrite, nitrous oxide ($N_2O$), or nitrogen gas ($N_2$) by the process termed denitrification. *Desulfovibrio* bacteria use sulfate ($SO_4^{2-}$) as the final electron acceptor resulting in the formation of $H_2S$. *Methanobacterium* uses carbonate ($CO_3^{2-}$), forming methane gas ($CH_4$). For more detailed information on microbial metabolism the reader is referred to, for example, Grant and Long (1985), Kuenen and Robertson (1987), Laanbroek (1990), and Paul and Clark (1996) (see also Chapters 8, 9, and 11).

## Wetland Algae

The assemblage of primitive plants that are collectively referred to as algae includes a tremendously diverse array of organisms. Algae may size from single cells as small as one micrometer to large seaweeds which may grow to over 50 meters. Many of the unicellular forms are motile, and may intergrade confusingly with the Protozoa (South and Whittick, 1987). Algae are ubiquitous; they occur in every kind of water habitat (freshwater, brackish, and marine). However,

they can also be found in almost every habitable environment on earth—in soils, permanent ice, snow fields, hot springs, and hot and cold deserts.

Algae may be an important component of a treatment wetland, either as an early colonizing community or as the intended dominant design community. The reader is referred to Vymazal (1995) for a more complete description of algae and element cycling in wetlands.

Algae are unicellular or multicellular, photosynthetic organisms that do not have the variety of tissues and organs of higher plants. Algae are a highly diverse assemblage of species that can live in a wide range of aquatic and wetland habitats. Many species of algae are microscopic and are only discernable as the green or brown color or "slime" occurring on submerged substrates or in the water column of lakes, ponds, and wetlands. Other algal species develop long, intertwined filaments of microscopic cells that look like mats of hair-like seaweed, submerged or floating in ponds and shallow water environments.

For the most part, algae depend on light for their metabolism and growth and serve as the basis for an autochthonous foodchain in aquatic and wetland habitats. Organic compounds created by algal photosynthesis contain stored energy, which is used for respiration or which enters the aquatic foodchain and provides food to a variety of microbes and other heterotrophs. Alternatively, this reduced carbon may be directly deposited as detritus to form organic peat sediments in wetlands and lakes.

Algae also depend on an ample supply of the building blocks of growth including carbon, typically extracted from dissolved carbon dioxide in the water column, and on macro and micronutrients essential to all plant life. When light and nutrients are plentiful, algae can create massive populations and contribute significantly to the overall food web and nutrient cycling of an aquatic or wetland ecosystem. When shaded by the growth of macrophytes, algae frequently play a less important role in wetland energy flows.

Most species of algae need ample water during some or all of their life cycles. Because water quality and climatic variables such as air and water temperature and light intensity are the principal determinants of algal species distribution, the algal flora of wetlands is generally similar to the regional algal flora living in ponds, lakes, springs, streams, rivers, and similar aquatic environments. The algal flora of wetlands differs from the flora of more aquatic environments primarily in response to varying water chemistry, water depth, light inhibition by emergent macrophytes, and seasonal desiccation which is more likely in shallow water environments.

## Classification

Algae comprise a very diverse group of organisms that, since the earliest times, defied precise definition. Bold and Wynne (1985) wrote:

> The term "algae" means different things to different people, and even the professional botanist and biologist find algae embarrassingly elusive to define. The reasons for this are that algae share their more obvious characteristics with other plants, while their really unique features are more subtle.

Algae may be classified by evolutionary or genetic relationships, morphological adaptations, or by ecological functions. Taxonomic identification of algae in wetlands rarely is required to design or operate wetland treatment systems. For detailed taxonomy of this phylum, the reader is referred to Lee (1980), South and Whittick (1987), and Vymazal (1995). Two general schemes for classification of aquatic algae (and microorganisms in general) can be found in the literature (Vymazal, 1995).

One scheme is a two-component system, as follows:

- Plankton: organisms that swim or float in the water
- Benthos: organisms that grow on the bottom of the water body

The second and older system makes a distinction within the attached (epiphytic) component:

- Periphyton: all aquatic organisms that grow on submerged substrates
- Benthos: organisms that grow on the bottom of the water body

Other designations include metaphyton, which is the community of floating algae.

## Plankton

Reynolds (1984) characterize plankton as the "community" of plants and animals adapted to suspension in the sea or in fresh waters and which is liable to passive movement by wind and current. Planktonic organisms are suspended in the water column and lack the means to maintain their position against the current flow, although many of them are capable of limited, local movement with the water mass. Phytoplankton occur in virtually all bodies of water. All algal groups except the Rhodophyceae, Charophyceae, and Phaeophyceae contribute species to the phytoplankton flora. Phytoplankton encompasses a surprising range of cell size and cell volume from the largest forms visible to the naked eye, (e.g., *Volvox* [500–1500 μm]) in the freshwater and *Coscinodiscus* species in the ocean, to the algae as small as 1 μm in diameter (Vymazal, 1995). Phytoplankton algae are mainly unicellular, though many colonial and filamentous forms occur, especially in fresh waters. Example photographs of wetland phytoplankton algae may be found in Vymazal (1995) and in Fox *et al.* (1981) for domestic wastewater. Planktonic or free-floating algae are generally not important in wetland ecosystems unless open or deep water areas are present. Plankton spend most of their life cycle suspended in the water column and are the most important algal component in lakes and

some ponds. Tychoplankton (pseudoplankton) are algae that initially grow as attached species and which subsequently break free from their substrate and live planktonically for part of their life cycle. Tychoplanktonic algal species are most common in streams and in littoral wetlands.

Plankton are probably not important as a component of pollutant processing in most wetlands. However, the use of emergent wetlands to shade out and remove plankton from facultative pond effluents is an important treatment wetland consideration.

### Attached Algae

As far as the attached algal communities are concerned, there are three overlapping terms used to describe algae growing attached to any kind of substrates: benthos, periphyton, and aufwuchs. In the literature, there is a lot of confusion and controversy about these terms (Vymazal, 1995). Benthos is composed of attached and bottom-dwelling organisms (Bold and Wynne, 1985). Epiphytic algae grow attached to various substrates and may be classified as:

- Epilithic (growing on stones)
- Epipelic (attached to mud or sand)
- Epiphytic (attached to plants)
- Epizoic (attached to animals)

Periphyton in its broad definition includes all aquatic organisms (microflora) growing on submergent substrates. Although periphyton usually begin colonization of new plant surfaces by attached algal growth of filamentous and unicellular species, this functional component also includes a variety of free-living algae (not attached to the surface), fungi, bacteria, and protozoans following a period of maturation. Periphyton growing on plants is often called epiphyton. Aufwuchs is a more general term than periphyton and includes all algae and associated microscopic life attached to all surfaces in an aquatic or wetland system. These surfaces frequently include living vascular plants as well as dead plants, leaves, branches, trunks, stones, and exposed substrates. Benthic or attached algae are more specific terms that refer only to the algal component of the periphyton or aufwuchs.

Epiphytic algae generally show little substrate specificity; many epiphytic species are encountered in natural epilithic communities and on artificial substrates. In spite of seeming relative indifference of epiphytic algae to their substrate, the epiphytic habitat has several distinctive attributes. The surface itself has a definite life span. New leaves are colonized as they develop during the growing season resulting in a summer and autumn peak in epiphytic biomass and productivity. The canopy of aquatic macrophytes often creates light-limiting conditions for epiphytic algae (Darley, 1982). On the other hand, decreases in growth and photosynthetic rates, as well as abundance and occurrence of submersed macrophytes, have been attributed to light attenuation by the periphyton complex (Vymazal, 1995).

In their use of nutrients from the sediment (via macrophyte tissue) as well as from the overlying water, epiphytes can play an important role in nutrient cycling. Much of the physiological research on epiphytic algae has focused on the question of nutrient transfer from rooted, aquatic, vascular plants to their epiphytes. A few studies have demonstrated a transfer of organic carbon, nitrogen, and phosphorus from macrophyte to the epiphytic community. Experiments with radio-labeled phosphorus show that this release is small for macrophytes in active growth (3–24%), though larger proportions (60%) can apparently be obtained by firmly attached epiphytic algae when phosphorus availability in the water phase is extremely low (Cattaneo and Kalff, 1979; Moeller *et al.*, 1988) The release is probably larger from senescent leaves, but perhaps of little significance because old leaves are subsequently shed (Sand-Jensen *et al.*, 1982). There is evidence that some rooted aquatic plants act as pumps, transferring phosphorus and other nutrients from the sediments to epiphytes and the water column. The amount of nutrient released, however, is very small (Cattaneo and Kalff, 1979).

Interactions between epiphytic algae and their host macrophytes have been subject to controversy. Competing hypotheses differ as to whether (1) the host macrophyte is a neutral substrate or (2) the host macrophyte influences epiphyton production and community composition by mechanisms independent of morphology. Similarities between natural and artificial macrophyte-substrates in community composition, biomass, and production of colonizing epiphyton support the former hypothesis. On the other hand, it has been found that epiphyton species composition and abundance were related to the macrophyte-mediated changes in the physicochemical environment. The responses of epiphytic and epipelic algae to primary physical, chemical, and biotic parameters have been discussed in detail by Wetzel (2001). Photographic examples of attached algae are given in Vymazal (1995).

### Filamentous Algae

Filamentous algae that occur in wetlands as periphyton or mats may dominate the overall primary productivity of the wetland, controlling dissolved oxygen and carbon dioxide concentrations within the wetland water column. They are opportunistic, because they can grow very rapidly compared to macrophytes. Therefore, the early period of constructed wetland life may create ideal conditions for algal establishment (Figure 3.1). However, macrophytes can later easily shade out the algae. Diurnal DO profiles in wetlands and other aquatic environments with substantial populations of submerged plants undergo major changes in relation to the daily gross and net productivity. Wetland water column DO can fluctuate from near zero during the early morning following a night of high respiration to well over saturation (>20 mg/L) in high algal growth areas during a sunny day. Dissolved carbon dioxide and consequently the pH of the water vary proportionally to DO because of the corresponding use of $CO_2$ by plants during photosynthesis and release at night during respiration. As $CO_2$ is stripped from the water column by algae during the day, pH may rise by 2 to 3 pH



**FIGURE 3.1** Algae were the first colonizers of this 25-ha constructed wetland cell near Carson City, Nevada.

units (a 100- to 1,000-fold decrease in $H^+$ concentration). These daytime pH changes are reversible, and the production of $CO_2$ at night by algal respiration frequently returns the pH to the previous day's value by early morning.

Algae also store and transform essential growth nutrients in wetlands and aquatic habitats. Because of their relatively low contribution to the overall fixed carbon in wetlands, algae do not constitute a major storage reservoir for these elements in wetlands. However, because of their high turnover rates in some aquatic habitats, algae may be important for short-term nutrient fixation and immobilization with subsequent gradual release and recycling. The functional result of this nutrient cycling is that intermittent high inflow concentrations of pollutants used by algae for growth may be immobilized and transformed more effectively than would be possible without these components, thereby reducing the amplitude of wetland constituent outflow concentrations.

For a detailed description of the importance of algae in wetlands, see Vymazal (1995).

### Wetland Macrophytes

Macrophytic plants provide much of the visible structure of wetland treatment systems. There is no doubt that they are essential for the high-quality water treatment performance of most wetland treatment systems. The numerous studies measuring treatment with and without plants have concluded almost invariably that performance is higher when plants are present. This finding led some researchers to conclude that wetland plants were the dominant source of treatment because of their direct uptake and sequestering of pollutants. It is now known that plant uptake is the principal removal mechanism only for some pollutants, and only in lightly loaded systems. During an initial successional period of rapid plant growth, direct pollutant immobilization in wetland plants may be important. For many other pollutants, plant uptake is generally of minor importance compared to microbial and physical transformations that occur within most wetlands. Macrophytic plants are essential in wetland treatment systems because they provide the structure that fosters many removal processes.

The term macrophyte includes vascular plants that have tissues that are easily visible. Vascular plants differ from algae through their internal organization into tissues resulting from specialized cells. A wide variety of macrophytic plants occur naturally in wetland environments. The United States Fish and Wildlife Service has more than 6,700 plant species on their list of obligate and facultative wetland plant species in the United States. Godfrey and Wooten (1979; 1981) list more than 1,900 species (739 monocots and 1,162 dicots) of wetland macrophytes in their taxonomy of the southeastern United States. Obligate wetland plant species are defined as those which are found exclusively in wetland habitats, whereas facultative species are those that may be found in upland or in wetland areas. There are many guidebooks that illustrate wetland plants (for example, Hotchkiss, 1972; Niering, 1985; Cook, 1996). Lists of plant species that occur in wetlands are available (e.g., RMG, 1992).

Wetland macrophytes are the dominant structural component of most wetland treatment systems. A basic understanding of the growth requirements and characteristics of these wetland plants is essential for successful treatment wetland design and operation.

#### Classification

The plant kingdom is divided taxonomically into phyla, classes, and families, with certain families either better represented or occurring only in wetland habitats. The major plant phyla are the mosses and clubmosses (*Bryophyta*) and the vascular plants (*Tracheophyta*). In the vascular plant phylum there are three important classes of plants: ferns (*Filicinae*), conifers (*Gymnospermae*), and flowering plants (Angiospermae). The flowering plants are further divided into the monocots (*Monocotyledonae*) and dicots (*Dicotyledonae*).

Because plant taxonomic families were developed to provide insight into the evolutionary affinity of plant species, it

is not surprising that some families are well represented by multiple obligate wetland species. Vascular plants including wetland plants may also be categorized morphologically by descriptors such as woody, herbaceous, annual, or perennial. Woody species have stems or branches that do not contain chlorophyll. Because these tissues are adapted to survive for more than one year, they are typically more durable or woody in texture. Herbaceous species have aboveground tissues that are leafy and filled with chlorophyll-bearing cells that typically survive for only one growing season. Woody species include shrubs that attain heights up to 2 or 3 m and trees that generally are more than 3 m in height when mature.

Annual plant species survive for only one growing season and must be reestablished annually from seed. Perennial plant species live for more than one year and typically propagate each year from perennial root systems or from perennial aboveground stems and branches. Nearly all woody plant species are perennial, but herbaceous species may be annual or perennial.

Four groups of aquatic macrophytes (Figure 3.2) can be distinguished on a basis of morphology and physiology (Wetzel, 2001):

1. Emergent macrophytes grow on water-saturated or submersed soils from where the water table is about 0.5 m below the soil surface to where the sediment is covered with approximately 1.5 m of water (e.g., *Acorus calamus, Carex rostrata, Phragmites australis, Schoenoplectus (Scirpus) lacustris, Typha latifolia*).
2. Floating-leaved macrophytes are rooted in submersed sediments in water depths of approximately 0.5 to 3 m and possess either floating or slightly aerial leaves (e.g., *Nymphaea odorata, Nuphar luteum*).
3. Submersed macrophytes occur at all depths within the photic zone. Vascular angiosperms (e.g., *Myriophyllum spicatum, Ceratophyllum demersum*) occur only to about 10 m (1 atm hydrostatic pressure) of water depth and nonvascular macroalgae occur to the lower limit of the photic zone (up to 200 m, e.g., *Rhodophyceae*).
4. Freely floating macrophytes are not rooted to the substratum; they float freely on or in the water and are usually restricted to nonturbulent, protected areas (e.g., *Lemna minor, Spirodella polyrhiza, Eichhornia crassipes*).

In addition, a large number of the emergent macrophytes can be established in floating mats, either with or without a supporting structure. Some species have one or more of these growth forms; however, there is usually a dominant form that enables the plant species to be classified. In emergent plant species, most of the aboveground part of the plant emerges above the water line and into the air.

Both floating and submerged vascular plant species may also occur in wetland treatment systems. Floating species have leaves and stems buoyant enough to float on the water surface. Submerged species have buoyant stems and leaves that fill the niche between the sediment surface and the top of the water column. Floating and submerged species prefer deep aquatic habitats, but they may occur in wetlands when water depth exceeds the tolerance range for rooted, emergent species.

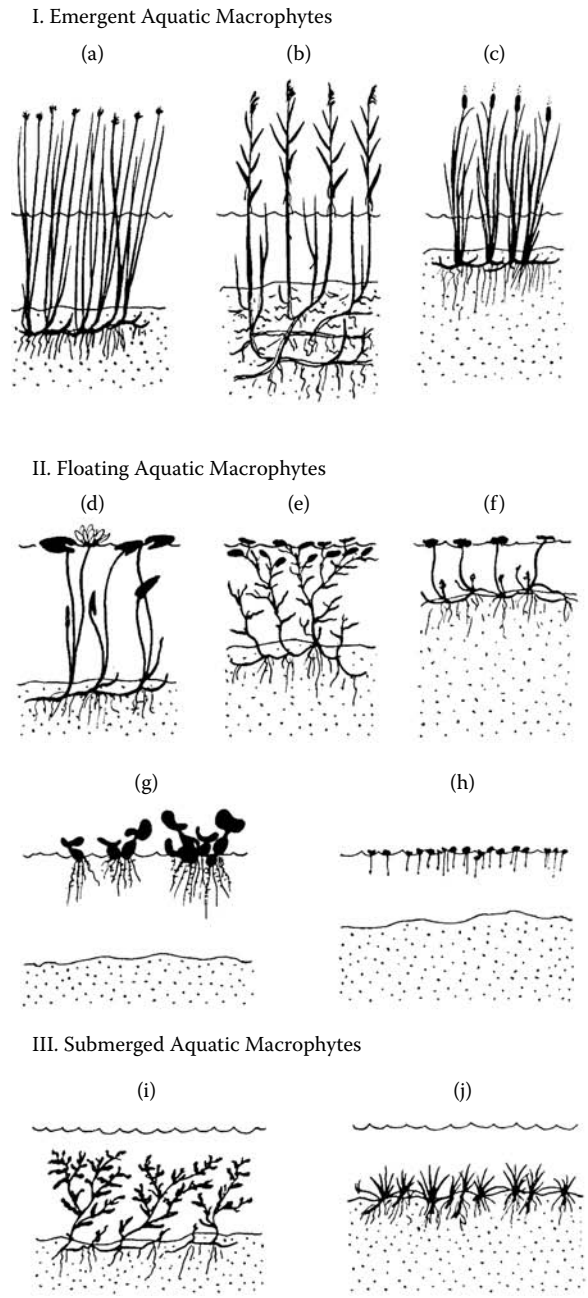



**FIGURE 3.2** Sketch showing the dominant life forms of aquatic macrophytes. The species illustrated are (a) *Scirpus* (*Schoenoplectus*) *lacustris*, (b) *Phragmites australis*, (c) *Typha latifolia*, (d) *Nymphaea alba*, (e) *Potamogeton gramineus*, (f) *Hydrocotyle vulgaris*, (g) *Eichhornia crassipes*, (h) *Lemna minor*, (i) *Potamogeton crispus*, (j) *Littorella uniflora*. (From Brix and Schierup (1989b). *Ambio* 18: 100–107. Reprinted with permission.)

**TABLE 3.1**
**Number of Plant Species by Group Found in Constructed Wetlands in the North American Database, Version 2.0***

| Plant Group | Number of Species Recorded |
|---|---|
| Emergent macrophyte | 501 |
| Floating aquatic plant | 31 |
| Submerged aquatic plant | 10 |
| Shrub | 17 |
| Tree | 25 |
| Unknown | 5 |
| Vine | 5 |
| **Totals** | **594** |

** This database is dominated by FWS wetlands, and covers only a subset of existing systems.*

*Source:* Data from NADB database (1998) *North American Treatment Wetland Database (NADB), Version 2.0*. Compiled by CH2M Hill, Gainesville, Florida.

**TABLE 3.2**
**Dominant Plant Species Found in Constructed Treatment Wetlands**

| Common Name | Scientific Name |
|---|---|
| Bacopa | *Bacopa caroliniana* |
| Bulrush | *Scirpus* spp. |
| Cattail | *Typha* spp. |
| Common reed | *Phragmites australis* |
| Coontail | *Ceratophyllum demersum* |
| Duck potato | *Sagittaria* spp. |
| Duckweed | *Lemna* spp. |
| Frogs-bit | *Limnobium spongea* |
| Pennywort | *Hydrocotyle* spp. |
| Pickerelweed | *Potederia* spp. |
| Pondweed | *Potamogeton* spp. |
| Reed canary grass | *Phalaris arundinacea* |
| Softrush | *Juncus* spp. |
| Spatterdock | *Nuphar luteum* |
| Water hyacinth | *Eichhornia crassipes* |
| Waterweed | *Elodea* spp. |

*Source:* Modified from NADB database (1998) *North American Treatment Wetland Database* (*NADB*), *Version 2.0*. Compiled by CH2M Hill, Gainesville, Florida.

Table 3.1 lists the classes of plants reported in treatment wetlands and their numbers. Table 3.2 lists the dominant plants in treatment wetlands.

## Adaptations to Life in Flooded Conditions

Prolonged flooding or waterlogging restricts oxygen movement from the atmosphere to the soil. Diffusion can occur but it is 10,000 times slower in saturated soils than it is in aerated soils (Greenwood, 1961). Upon flooding, respiration by aerobic bacteria and other organisms consume the oxygen remaining in the soil within hours to days (Pezeshki, 1994). Soil oxygen deficiency (partial hypoxia, complete anoxia) poses the main ecological problem for plant growth as it affects plant functions such as stomatal opening, photosynthesis, water and mineral uptake, and hormonal balance (Kozlowski, 1984b). Life in permanently or periodically anaerobic soils or substrates is more difficult than living in mesic soils due to the nature of a highly reduced environment (low redox potential), possibly together with soluble phytotoxins (Tiner, 1999).

A wide range of adaptations make it possible for plants to grow in water or wetlands. These adaptations include physiological responses, morphological adaptations, behavioral responses, reproductive strategies, and others (Table 3.3). Major plant adaptations in free water surface (FWS) and subsurface constructed wetlands are shown in Figures 3.3 and 3.4. For a detailed description of macrophyte adaptations and responses to flooding see Hook and Crawford (1978), Kozlowski (1984a), Crawford (1987), Hejný and Hroudová (1987), or Jackson *et al*. (1990).

One of the most important adaptations to flooding is the development of aerenchymous plant tissues (Figure 3.5) that transport gases to and from the roots through the vascular tissues of the plant above water and in contact with the atmosphere, providing an aerated root zone and thus lowering the plant's reliance on external oxygen diffusion through water and soil (Armstrong, 1978; Jackson and Drew, 1984; Zimmerman, 1988; Brix, 1993). Lenticels or small openings on the above water portions of these plants provide an entry point for atmospheric oxygen into this aerenchymous tissue network. Lenticel surface area may be increased through plant growth, height increases, or the formation of swollen buttresses in trees and woody herbs and in cypress knees.

Plant survival in flooded environments is a balance between the severity of oxygen limitation and the adaptations available to overcome this oxygen shortage. Thus, hydrophytic plants may be adapted to survive and even grow in specific flooded conditions, such as three months each year, or in "clean" or flowing water, which might have higher in situ dissolved oxygen concentrations (Gosselink and Turner, 1978). However, these same plants may not be able to grow or survive during five months of flooding or in stagnant or "dirty" water conditions. This is shown in Figure 3.3. Likewise, plants may have adaptations that allow prolonged survival in one foot of water but not at two feet. It may be hypothesized that this balance is tilted unfavorably at higher water levels because of reduced aerial plant stem surface area to provide oxygen to the roots

**TABLE 3.3**
**Plant Adaptations or Responses to Flooding and Waterlogging**

| | |
|---|---|
| *Morphological* | Stem hypertrophy (e.g., buttressed tree trunks); large air-filled cavities |
| *Adaptations/responses* | In the center (stele) of roots and stems; aerenchyma tissue in roots and other plant parts; hollow stems; shallow root systems; adventitious roots; pneumatophores (e.g., cypress knees); swollen, loosely packed root nodules; lignification and suberization (thickening) of roots; soil water roots; succulent roots; aerial root-tips; hypertrophied (enlarged) lenticle; relatively pervious cambium (in woody species); heterophylly (e.g., submerged versus emergent leaves on same plants); succulent leaves. |
| *Physiological adaptations* | Transport of oxygen to roots from lenticles and/or leaves (as often evidenced by oxidized rhizospheres); anaerobic respiration; increased ethylene production; reduction of nitrate to nitrous oxide and nitrogen gas; malate production and accumulation; reoxidation of NADH; metabolic adaptations |
| *Other adaptations/responses* | Seed germination under water; viviparous seeds; root regeneration responses (e.g., adventitious roots); growth dormancy (during flooding); elongation of stem or petioles; root elongation; additional cell wall structures in epidermis or cortex; root mycorhizzae near upper soil surface; expansion of coleoptiles (in grasses); change in direction of root or stem growth (horizontal or upward); long lived seeds; breaking of dormancy of stem buds (may produce multiple stems or trunks). |

*Source:* From Tiner (1999) *A Guide to Wetland Identification, Delineation, Classification, and Mapping.* CRC Press, Boca Raton, Florida.

through the lenticels and aerenchymous tissues. This proposed explanation is supported by the finding that hydrophytes generally respond to flooding by growing taller, a growth response that allows a more favorable balance between emergent and submerged plant organs (Grace, 1989).

## Hydropattern

The term hydropattern refers to the time series of water depths in the wetland. The concept of hydropattern, or water regime, includes two interdependent components: (1) the duration of flooded or saturated soil conditions (the hydroperiod

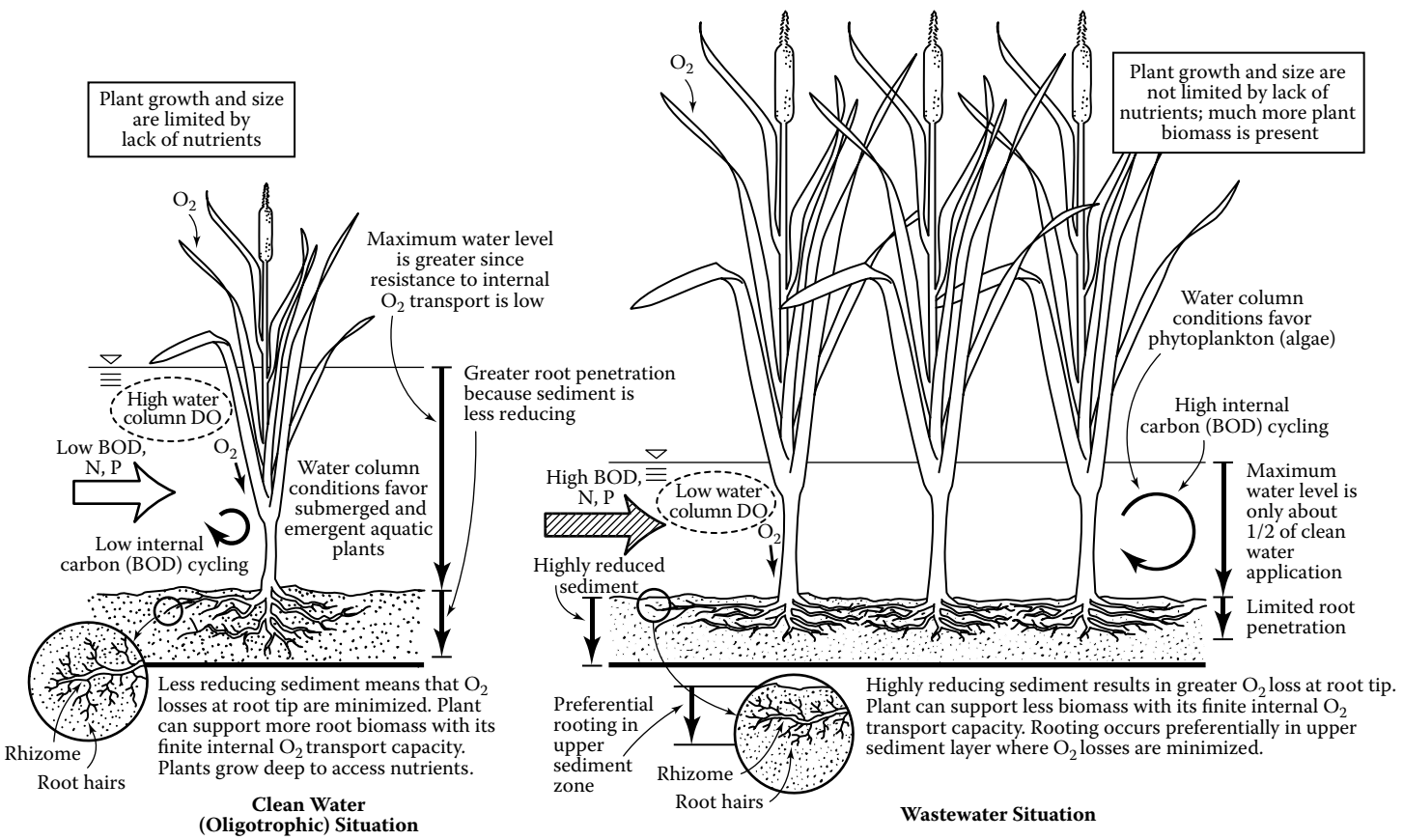



**FIGURE 3.3** Plant adaptations to primary domestic wastewater stresses in FWS wetlands. (Adapted from Wallace and Knight (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements.* Final Report, Project 01-CTS-5, Water Environment Research Foundation (WERF): Alexandria, Virginia. Reprinted with permission.)

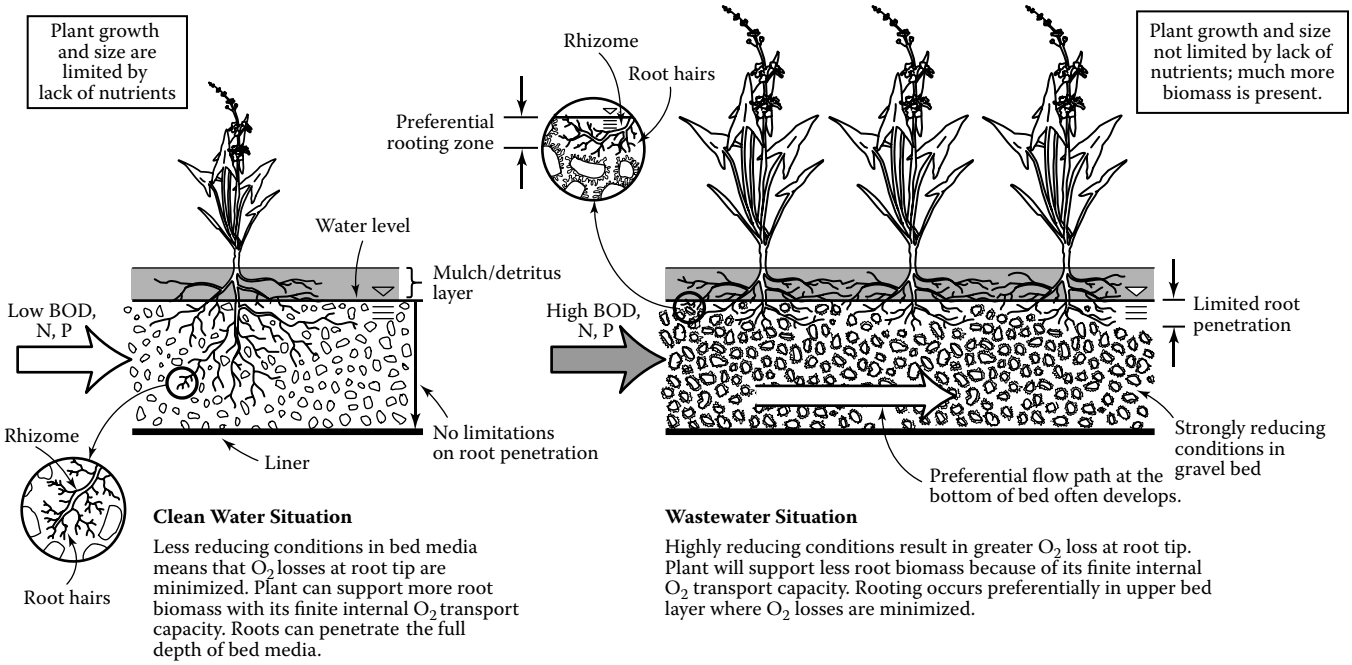



**FIGURE 3.4** Plant adaptations to primary domestic wastewater stresses in HSSF wetlands. (Adapted from Wallace and Knight (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements*. Final Report, Project 01-CTS-5, Water Environment Research Foundation (WERF): Alexandria, Virginia. Reprinted with permission.)

as a percentage of time with flooding), and (2) the depth of flooding (Gunderson, 1989). Although hydroperiod refers to the duration of flooding, the term water regime refers to hydroperiod as well as to the combination of water depth and flooding duration (depth-duration curve). The duration and depth of flooding affect plant physiology because of soil oxygen concentration, soil pH, dissolved and chelated macro and micronutrients, and toxic chemical concentrations. Figure 3.6 uses a graph of water level within a wetland over an annual period to illustrate these two aspects of hydroperiod and water regime. Duration of flooding refers to the percentage of time that a wetland site is flooded or saturated, and depth of flooding refers to the minimum, average, and maximum depths of water at a given or typical spot within

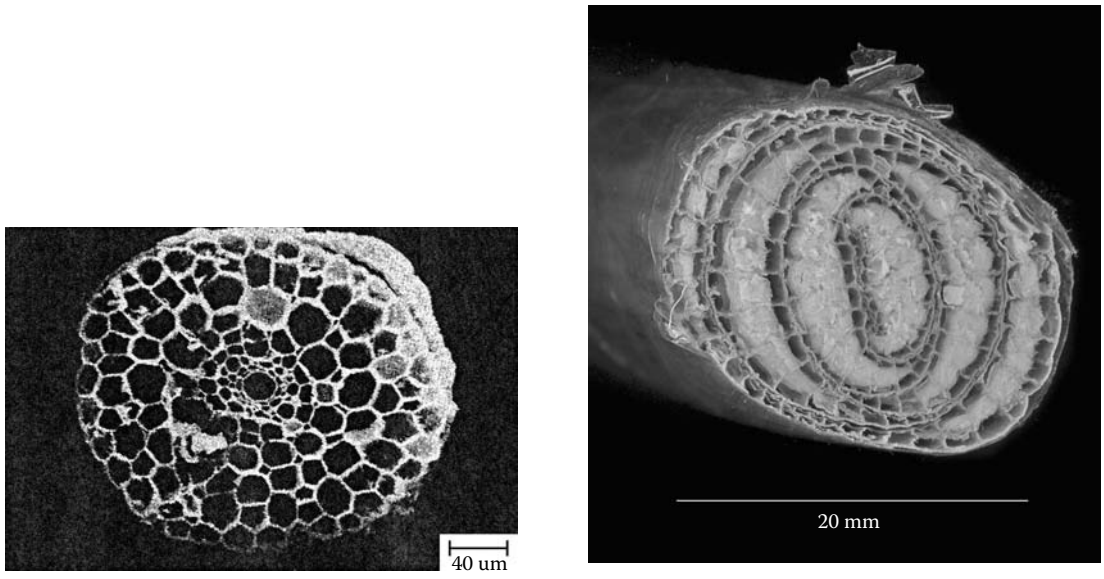



**FIGURE 3.5** (a) Internal gas passages in a *Phragmites* root. (From Armstrong and Armstrong (1990b) In *Constructed Wetlands in Water Pollution Control*. Cooper and Findlater (Eds.), Pergamon Press, Oxford, United Kingdom, pp. 529–534. Reprinted with permission.) (b) Internal gas passages in a *Typha* culm.

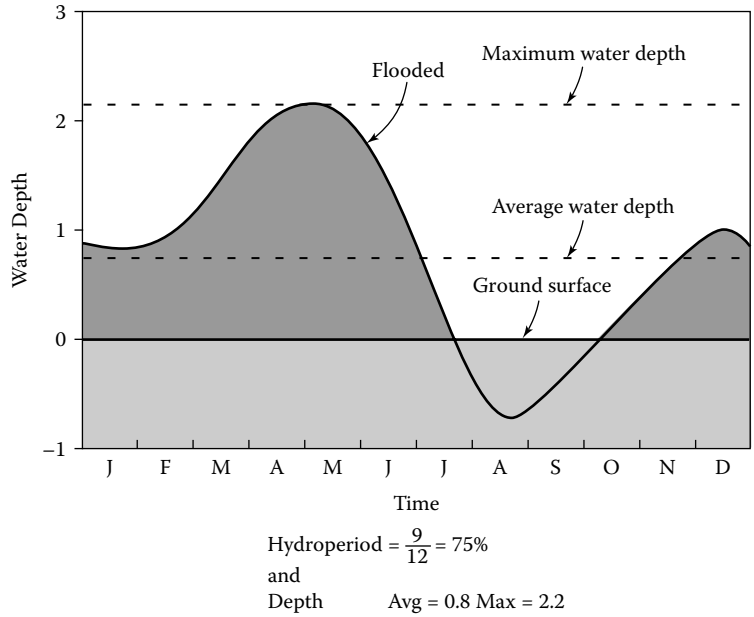



**FIGURE 3.6** Components of hydropattern: hydroperiod and wetland water regime. (From Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

a wetland. Hydroperiod curves provide a convenient method for estimating the percentage of time that a wetland is flooded at any water depth and can summarize water level data over any period of record. Note that water level charts and depth-duration curves also can summarize the time and depth that water is located below the ground surface.

Although the presence of water separates uplands from wetlands and aquatic ecosystems, hydropattern is the most important contributor to wetland type or class (Gosselink and Turner, 1978; Gunderson, 1989). The importance of this factor in wetland treatment system design and operation cannot be overstated because incorrect understanding of the hydroperiod and water regime limitations of wetland plant species is a frequent cause of vegetation problems in natural and constructed wetlands. Measuring the hydroperiod is relatively easy. However, selecting the optimal hydroperiod for wetland treatment design and performance is complex.

## Oxygen Transport as a Treatment Function

In order to survive in the saturated rooting environment, emergent wetland plants transport oxygen from their leaves down through their stalks to the root tissue (Armstrong, 1979). Because the aerenchyma passageways have occasional blockages to prevent flooding if the root tissues are damaged, internal transport of oxygen is a diffusion-limited process. Some plant species can increase oxygen transport by convective flow of gases (Brix, 1990; Armstrong and Armstrong, 1990a; Brix, 1994b). Dead and broken shoots and stubble also form air pipes to the root zone. Of interest here is the fact that significant quantities of oxygen pass down through the airways to the roots (Brix and Schierup, 1990; Brix, 1993); and that significant quantities of other gasses, such as carbon dioxide and methane, pass upward from the root zone. Internal gas spaces in a *Phragmites* root and a *Typha* culm are shown in Figure 3.5.

The oxygen is used for root respiration and to help detoxify the environment encountered by the growing root tip. Consequently, there are limits as to how far plants can propagate their root systems in a highly reducing environment (Armstrong *et al*., 1990). Some—probably most—of the oxygen passing down the plant into the root zone is used in plant respiration (Brix, 1990). The excess supply of $O_2$ over that required for plant respiration is termed the *plant aeration flux* (PAF), has been the subject of many research endeavors (Armstrong *et al*., 1990; Brix, 1990; Gries *et al*., 1990; Sorrell *et al*., 2000; Wu *et al*., 2001; Bezbaruah and Zhang, 2003). The difficulty of measuring processes and concentrations in the root microzones has been a major factor in the widely disparate estimates of PAF (Kadlec and Knight, 1996).

Chemical conditions in the root zone are important determinants of the potential for significant PAF (Sorrell, 1999). Hydroponic studies most often create root environments that do not include a significant sediment oxygen demand. Roots are numerous under such conditions, and exchange oxygen along much of their length (Armstrong *et al*., 1990). The morphology and physiology of roots is very different in the anaerobic environment often associated with treatment wetland soils. Under treatment conditions, the number of roots is significantly less than in clean soil or hydroponic conditions. Roots become armored along much of their length, and $O_2$ losses to the soil and water occur only in a small apical region (Brix, 1994c).

Oxygen transfer by plants was initially thought to be a dominant mechanism in SSF wetland treatment (Kickuth and Könemann, 1987), but recent work has demonstrated that the

vast majority of the oxygen transferred by the plant is used for root metabolism, and the amount released to the rhizosphere is small. Different test methods yield different results, but a value of 0.02 g/m$^2$·d has been established in two independent studies (Brix and Schierup, 1990; Wu *et al*., 2001). As a result, most modern designers have abandoned the concept of plants acting as "solar powered aerators." Since studies have proven plant-induced oxygen transfer rates to be so small, current design guidelines recommend assuming that oxygen delivered to the wastewater by the plant roots is negligible (U.S. EPA, 2000a). For a further discussion of root aeration, see Chapter 5.

## 3.2 BIOMASS AND GROWTH

The term *biomass* is most frequently defined as the mass of all living tissue at a given time in a given unit of Earth's surface (Lieth and Whittaker, 1975). It is commonly divided into belowground (roots, rhizomes, tubers, etc.) and aboveground biomass (all vegetative and reproductive parts above the ground level). The term *standing crop* includes live parts and dead parts of live plants that are still attached. These dead parts of plants together with still standing dead plants are called *standing dead*. The term *litter* refers to those dead parts of the plant that have fallen on the ground or sediment, but in some cases also includes standing dead. These compartments exchange material, but not uniformly, over the course of the year (Figure 3.7).

*Peak standing crop* is defined as the single largest value of plant material present during a year's growth (Richardson and Vymazal, 2001). In tropical communities, with an almost constant biomass, it is not profitable to search for an annual maximum (Westlake, 1969). However, in all other climatic regions the biomass fluctuates widely throughout the year (Dykyjová and Kvet, 1978; Shew *et al*., 1981; Kaswadji *et al*., 1990). The range of standing crop of wetland plants is quite large (Kvet, 1982; Mitsch and Gosselink, 1993; Vymazal, 1995). Another terminology has been advanced by Mueleman *et al*. (2002), which suggests that the total is *phytomass*, which is composed of living material (biomass) and dead (necromass).

*Gross Primary Production* (also called *Gross Primary Productivity*, or GPP) is normally defined as the assimilation of organic matter by a plant community during a specified period, including the amount used by plant respiration. *Net Primary Production*, or NPP, is defined as the biomass that is incorporated into a plant community during a specified time interval, less that respired. This is the quantity that is measured by harvest methods and which has also been called *net assimilation* or *apparent photosynthesis*. The term *Net Aerial (or Aboveground) Primary Production* (NAPP) is defined as the biomass incorporated into the aerial parts (leaf, stem, seed, and associated organs) of the plant community (Milner and Hughes, 1968).

NPP of freshwater marshes is estimated most frequently through harvest of annual peak standing stocks of live and dead plant biomass. When root biomass is measured, it is usually an important part of net annual plant production. Some researchers consider net primary productivity estimates by peak standing stock to be underestimates because they do not account for biomass turnover during the growing season (Pickett *et al*., 1989). Kvet (1982) estimates turnover rates (productivity/biomass) in the range of 1.1–1.5 for submerged species, 1.05–1.5 for short emergent species, 1.05–1.3 for tall emergent species, and 1.15 for tall graminoids. For comparison, phytoplankton has a turnover rate in the range of 450–600. Table 3.4 summarizes some typical estimated net production data from wetland ecosystems, both natural and treatment.

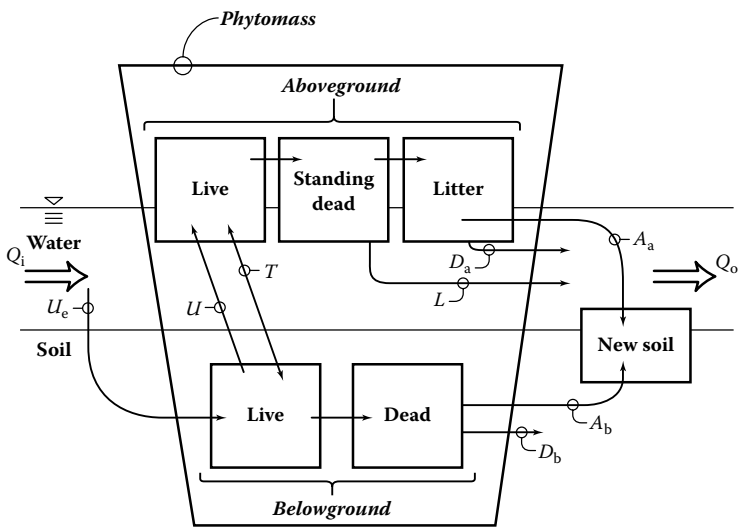



**FIGURE 3.7** Transfers of materials in the biosphere of wetlands. Biomass consists of living, above and below ground components. Necromass consists of dead roots and rhizomes, plus aboveground standing dead and litter. Phytomass is the combination of biomass and necromass. Transfer to the phytomass occurs by external plant uptake ($U_e$). Transfer back to surface water and porewater occurs via leaching ($L$) and decomposition ($D_a$ and $D_b$). Necromass residuals lose their identity, and accrete as new soils and sediments ($A_a$ and $A_b$).

**TABLE 3.4**
**End of Season Plant Biomass in Wetlands**

| Species | Location | Reference | Water | S/P/E | Live Above ($g/m^2$) | Total Above ($g/m^2$) | Roots and Rhizomes ($g/m^2$) |
|---|---|---|---|---|---|---|---|
| **Cattails** | | | | | | | |
| *Typha latifolia* | Wisconsin | Smith *et al.* (1988) | N | 105/245/290 | — | 1,400 | 450 |
| *Typha latifolia* | Texas | Hill (1987) | N | 60/240/345 | — | 2,500 | 2,200 |
| *Typha glauca* | Iowa | van der Valk and Davis (1978) | N | 120/265/290 | 2,000 | — | 1,340 |
| *Typha latifolia* | Michigan | Unpublished data from Houghton Lake | N | 120/245/275 | 490 | 890 | 6,200 |
| *Typha latifolia* | Michigan | Unpublished data from Houghton Lake | S | 120/245/275 | 1,240 | 2,310 | 2,900 |
| *Typha angustifolia* | Michigan | Unpublished data from Houghton Lake | S | 120/245/275 | 1,886 | 3,615 | — |
| *Typha latifolia* | Kentucky | Pullin and Hammer (1989) | P | — | 5,602 | — | 3,817 |
| *Typha angustifolia* | Kentucky | Pullin and Hammer (1989) | P | — | 5,538 | — | 4,860 |
| **Bulrushes** | | | | | | | |
| *Scirpus fluviatilis* | Iowa | van der Valk and Davis (1978) | N | 130/265/285 | 790 | — | 1,370 |
| *Scirpus validus** | Iowa | van der Valk and Davis (1978) | N | 120/210/300 | 2,100 | — | 1,520 |
| *Scirpus validus* | New Zealand | Tanner (2001a) | P | 30/205/350 | 2,100 | 2,650 | 1,200 |
| *Scirpus validus* | Kentucky | Pullin and Hammer (1989) | P | — | — | 2,355 | 7,376 |
| *Scirpus cyperinus* | Kentucky | Pullin and Hammer (1989) | P | — | — | 3,247 | 12,495 |
| **Phragmites** | | | | | | | |
| *Phragmites australis* | U.K. | Mason and Bryant (1975) | N | 75/220/305 | 942 | 1,275 | — |
| *Phragmites australis* | Iowa | van der Valk and Davis (1978) | N | — | — | 1,110 | 1,260 |
| *Phragmites australis* | Netherlands | Mueleman *et al.* (2002) | N | 105/255/350 | 2,900 | 3,200 | 7,150 |
| *Phragmites australis* | Brisbane | Greenway (2002) | S | — | 1,460 | 2,520 | 1,180 |
| *Phragmites australis* | Netherlands | Mueleman *et al.* (2002) | P | 105/255/355 | 5,000 | 5,500 | 3,890 |
| *Phragmites australis* | New York | Peverly *et al.* (1993) | L | 100/270/330 | 10,800 | — | 8,700 |

*Note:* Water type is N = no wastewater, S = nutrients at secondary treatment levels, P = nutrients at primary treatment levels, L = landfill leachate at around 300 $gN/m^3$. S/P/E refers to the start, peak, and end yeardays of the growing season (add 182 days for Southern Hemisphere).

*Currently known as *Schoenoplectus tabernaemontani*.

Primary productivity of wetland plants is increased by the availability of water, light, and nutrients. Adding wastewater to wetlands generally increases the availability of water and nutrients and consequently results in the stimulation of gross and net primary productivity of these ecosystems (Guntenspergen and Stearns, 1981; Nixon and Lee, 1986).

## Fertilizer Response

The growth of wetland plants, like that of terrestrial plants, is stimulated by fertilization (Boyd, 1971; Jordan *et al*., 1999; Mueleman *et al*., 2002). When a wetland becomes the recipient of waters with higher nutrient content than those it has been experiencing, there is a response of the vegetation, both in species composition and in total biomass. This response has been detailed for the Houghton Lake wetland by Kadlec and Alvord (1989). The increased availability of nutrients produces more vegetation during the growing season, which in turn means more litter during the nongrowing season. This litter requires several years to decay, and hence the total pool of living and dead material grows slowly over several years to a new and higher value. A significant quantity of structural components are thus retained in the wetland.

Primary productivity of wetland plants is increased by the availability of water, light, and nutrients. Adding wastewater to wetlands generally increases the availability of water and nutrients and consequently results in the stimulation of gross and net primary productivity of these ecosystems. Figure 3.8 illustrates the typical plant growth response curve to increased concentrations of nitrogen and phosphorus. The maximum rate of plant growth is attained as nutrient levels are initially increased. However, at higher nutrient levels, plant growth levels off while luxury nutrient uptake continues, and at higher nutrient concentrations, phytotoxic responses are observed.

Figure 3.9 gives an example of this fertilizer response for soft-stemmed bulrush, *Schoenoplectus (Scirpus) validus*, grown in dairy wastewater. As the nitrogen concentration was increased, both above- and belowground biomass increases (Tanner, 1994). However, there is a suggestion of a maximum

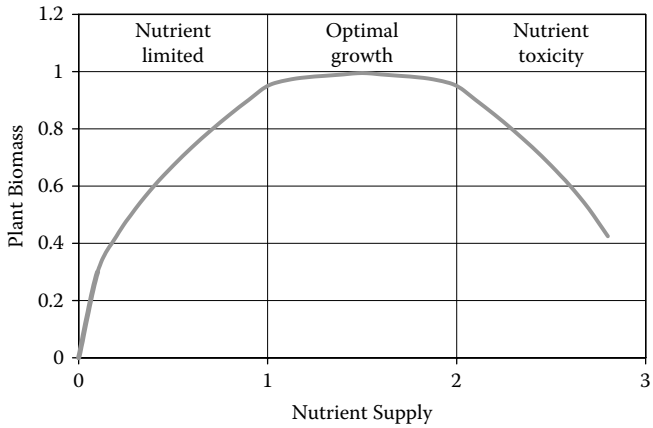



**FIGURE 3.8** General relationship between plant biomass and nutrient concentration in the water column.

at the highest concentrations. In fact, root death was noted by Tanner (1994) in plants growing in piggery wastewaters, where high ammonia concentrations (mean 222 mg/L) were at potentially phytotoxic levels. For example, ammonia concentrations of 200 mg/L are known to be detrimental to water hyacinths (de Casabianca-Chassany *et al.*, 1992). Other studies have also established similar effects for other treatment wetland plants. Hill *et al.* (1997) found dry matter production of *Typha latifolia, Phragmites australis,* and *Sagittaria latifolia* were unaffected by ammonia in the concentration range 20–80 mg/L range. Dry matter production of *Schoenoplectus* (*Scirpus*) *acutus* was found to be maximized in the 30–50 mg/L range, and then to rapidly fall off above 60 mg/L.

### Seasonal Patterns

The growth and senescence of the soft tissue macrophytes commonly used for wastewater treatment all follow a common seasonal pattern in temperate climates. In northern climates, growth begins at the time of frost disappearance (around April), and senescence begins in early autumn (around September). This autumnal decline creates standing dead aboveground plant material, which subsequently in part decomposes, and in part falls to the soil surface.

A specific case for *Typha* is shown in Figure 3.10, which is representative of other emergent macrophytes as well. New growth proceeds from small shoots that may be initiated as early as late summer of the preceding year for *Typha* (Bernard, 1999), but remain tiny and dormant over the winter season. Aboveground biomass increases rapidly in early spring, typically commencing from late February to Late April, depending on climate. Growth tapers off, causing aboveground biomass to peak in late summer to early autumn. The size of the peak standing crop varies considerably with plant species and degree of nutrient availability (see Table 3.4). Typically, there is some degree of senescence that accompanies the later portions of the growth period, so that the total peak standing crop exceeds the live peak standing crop. During autumn, more rapid senescence occurs, leaving only a residual of standing and/or prostrate aboveground dead material.

Belowground biomass follows a much more muted annual cycle. In some cases, available methods of root and rhizome biomass measurement are not accurate enough to clearly define a pattern (Figure 3.10). In other cases, a midsummer depression has been found, to about 50% of the midwinter maximum (Smith *et al.*, 1988; Mueleman *et al.*, 2002). But mid summer maxima were found for *Sparganium* and *Phragmites* in Iowa (van der Valk and Davis, 1978). When root biomass is measured, it is usually an important part of net annual plant production. NPP estimates by peak standing stock are underestimates because they do not account for biomass turnover during the growing season. For instance, a multiplier of 1.2–1.4 for aboveground cattails and *Spartina* has been reported by Cronk and Fennessy (2001).

In tropical or subtropical climates, seasonality is much more muted (Figure 3.11). There may be periods of dormancy

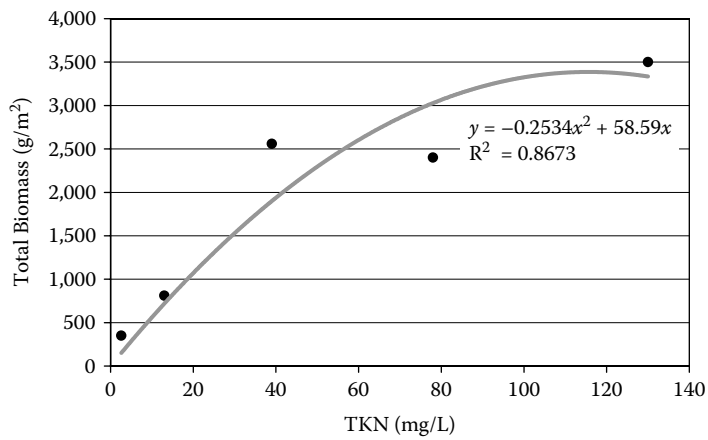



**FIGURE 3.9** Growth of *Schoenoplectus* (*Scirpus*) *validus* in dairy wastewater at various dilutions. The accompanying range of total phosphorus concentrations was 0.3–14.8 mg/L. (Data from Tanner (1994) *Aquatic Botany* 47(2): 131–153.)

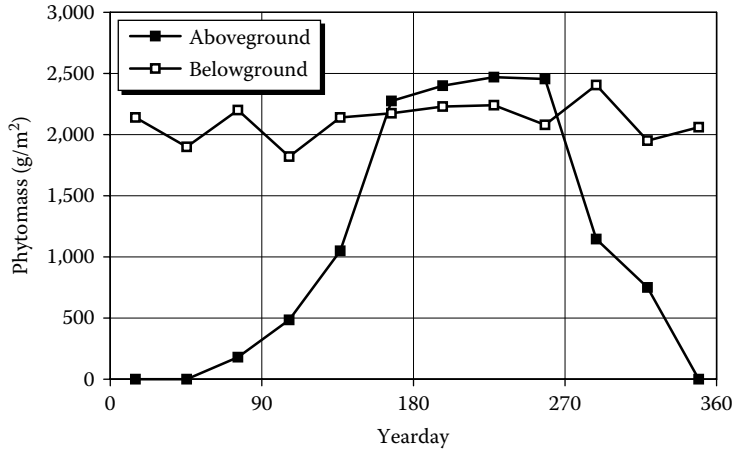



**FIGURE 3.10** Seasonal patterns of above- and belowground *Typha angustifolia* phytomass at Richardson, Texas. The climate is warm temperate. Points are averages for two years. (Data from Hill (1987). *Aquatic Botany* 27: 387–394.)

and of regrowth, but there is typically not complete senescence and death of all aboveground plant parts.

Two other factors are important in assessing the growth of wetland plants: the length of the growing season, and belowground productivity. All of the growth for the year occurs in about 100 days in high latitudes, whereas systems in the tropics grow year-round (see Table 3.4). Therefore, the instantaneous growing season rate is much higher than the annualized rate for northern systems. Belowground biomass is typically comparable to aboveground biomass, although the root-to-shoot ratio is sensitive to nutrient status and other variables. The ratio of below to aboveground biomass is generally less in a fertilized environment than in a lower nutrient (natural) environment (Mueleman *et al*., 2002). Kadlec and Alvord (1989) indicated that belowground biomass responded to fertilization differently from aboveground biomass. The initial vegetation showed greatly reduced root biomass in response to the added nutrients: 1,500 g/m$^2$ versus 4,000 g/m$^2$ at the end of the growing season. There are some reports that root growth and activity continues much longer than for aboveground plant parts (Prentki *et al*., 1978).

Roots and rhizomes persist over winter in northern climates, and therefore standing crop alone is not a measure of productivity. Estimates of turnover times are on the order of two to three years for herbaceous wetland plants. For example, Tanner (2001a) estimated a lifetime of 18 to 24 months for *Schoenoplectus* rhizomes, and Prentki *et al*. (1978) reported 1.5–2 years for *Typha* rhizomes and at least three years for *Phragmites* rhizomes. Therefore, the total growth rate for wetland plants is much higher than for aboveground parts alone.

These factors lead to the conclusion that plant growth is much higher than one standing (aboveground) crop per year. Table 3.5 presents a hypothetical illustration of factors for two climate zones. The growth of plant biomass during the

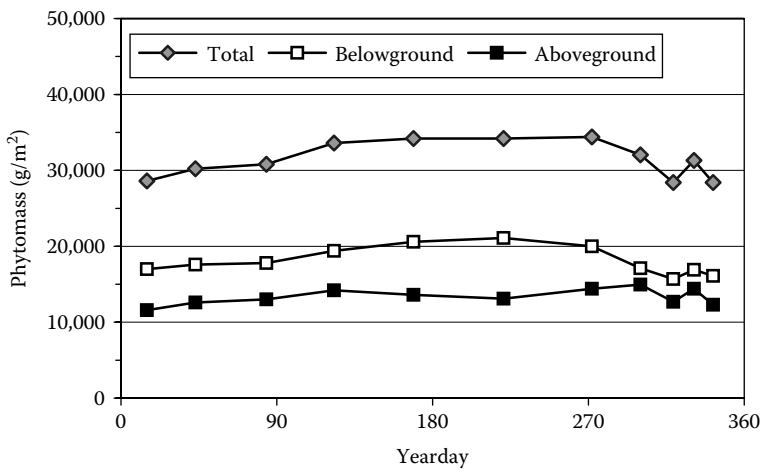



**FIGURE 3.11** The pattern of growth of *Phragmites australis* in the warm dry continental climate of Griffith, Australia. The water phosphorus concentration was 12 mg/L, and the approximate annual temperature range was from 10–23°C. (Data from Hocking (1989a) *Australian Journal of Marine and Freshwater Research* 40: 421–444.)

**TABLE 3.5**
**Hypothetical Growth Characteristics of Wetlands Growing in Temperate and Subtropical Conditions**

| Characteristic | Unit | Temperate Growing Season (M–J–J–A) | Annual | Subtropical Annual |
|---|---|---|---|---|
| Peak standing crop aboveground | $g/m^2$ | 2,000 | 2,000 | 2,000 |
| Growth (GPP/NPP = 1.3) | $g/m^2$ | 2,600 | 2,600 | — |
| Growth (4 turnovers per year) | $g/m^2$ | — | — | 8,000 |
| Growing season | days | 120 | 365 | 365 |
| Growth rate above | $g/m^2 \cdot d$ | 21.7 | 7.1 | 21.9 |
| Belowground crop (root/shoot = 1.0) | $g/m^2$ | 2,000 | 2,000 | 2,000 |
| Growth (0.5 turnovers per year) | $g/m^2$ | 1,000 | 1,000 | 1,000 |
| Growing season | days | 240 | 365 | 365 |
| Growth rate below | $g/m^2 \cdot d$ | 4.2 | 2.7 | 2.7 |
| Total growth rate | $g/m^2 \cdot d$ | 25.8 | 9.9 | 24.7 |
| Annualized instantaneous growth rate | $g/m^2 \cdot yr$ | 9,429 | 3,600 | 9,000 |
| Undepleted solar radiation | $MJ/m^2 \cdot d$ | 38 | 24 | 31 |

*Note:* These both grow at about the same rate during their respective growing seasons, which are year round in the warm climate.

respective growing seasons is about the same, but the growing season is much attenuated in northern climates. As a result, the annual growth is higher in the warmer environment.

## Start-Up: Wetland Vegetation Changes

A constructed wetland begins its existence with the vegetation placed by the constructors, and the seed bank associated with the selected soils. A natural wetland will have evolved over time to contain a mix of vegetation commensurate with the hydropattern and water quality conditions prior to wastewater addition. In either case, the wetland vegetation is likely to change over the course of time, as local adaptations to the treatment hydropattern and quality occur. The plant community that develops over time is a function of organic loading, hydrology, and climate. FWS wetlands that are heavily loaded with organic matter and nutrients will typically develop a less diverse plant community since fewer plant species are able to tolerate the reducing conditions that develop under these circumstances. In polishing wetlands with very high water quality, a diverse species composition may develop.

### Individual Plants

Plants reproduce in a two principal ways, by seeding and by vegetative reproduction. A plant starting from seed is a new individual, whereas it is not so easy to identify new individuals when new shoots arise from underground runners. Bulrushes tend to spread in a radial habit, with clumps growing in diameter. Cattails and *Phragmites* tend to spread in a linear mode, with new shoots emerging from a runner at intervals (Figure 3.12). Such runners can extend several meters in just one growing season, for both cattails and *Phragmites*.

Aboveground parts of plants in cold environments have a life span dictated by the photoperiod and frost conditions of the region. They live through one growing season, and new plants emerge the next year, from root stock or from seed. However, in warm climates, individual plants may persist for more than one year. Davis (1989) tagged individual leaves of 43 individual shoots of *Typha domingensis*, and followed their growth over their entire life history in a Florida wetland (Figure 3.13). He found that leaf growth and mortality continued throughout the life span of each tagged plant. New leaves emerged and grew, even while total biomass declined. Older leaves senesced, broke, or died even while total biomass increased. This continual growth and mortality resulted in an annual turnover of 4.4 ± 0.7 times the mean standing crop (Davis, 1989).

The concept of individual plant life history becomes important when, as is a common case, an entire wetland is planted at one time, creating a cohort of plants that will all live about the same length of time. Clearly, without regeneration this wetland will be devoid of plants after a few years. Therefore, the key to a self-sustaining wetland plant community is not only the survivability of plants in the treatment environment, but also the ability to regenerate.

### Plant Coverage

The vegetative cover of a treatment wetland refers to the area of wetland plants, and is concerned with four principal measures: (1) fraction areal coverage, (2) stem density, (3) submerged area, and (4) underwater porosity.

## Fractional Coverage

Most FWS constructed treatment wetlands are not monotypic communities, but rather contain a patchwork of open water, SAV, EAV, and FAV. In contrast, many SSF systems are in fact completely vegetated with uniform stands of EAV.



**FIGURE 3.12** *Phragmites* spreads vegetatively via linear runners. Dr. Hans Brix holds a specimen of only a few weeks' age, in a sludge drying reed bed in Denmark.

In both cases, the vegetation contributes to treatment, with greater effect at lighter pollutant loadings. For example, FWS phosphorus removal has been strongly linked to the fractional coverage of different community types (Lakhsman, 1982; Juston, 2006). Therefore, it is useful to distinguish between various degrees of vegetative completeness. Aerial photography or other remote sensing can be used to measure coverage of emergent plants, but it is more difficult to determine the presence of SAV (Rutchey and Vilcheck, 1999). If the wetland has design bathymetry including deep zones, then that information provides estimates of coverage of EAV.

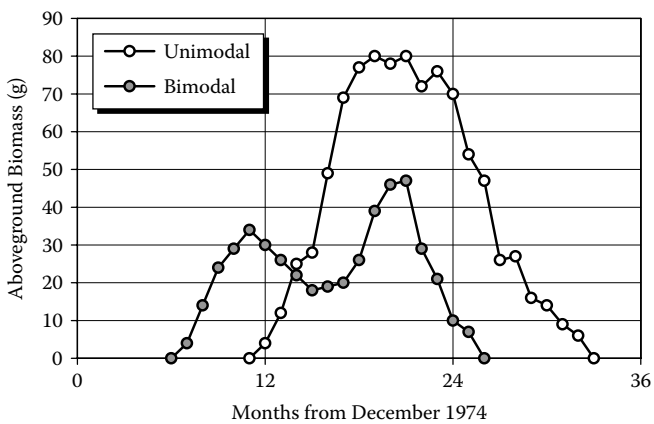



**FIGURE 3.13** Growth of single shoots of *Typha domingensis* in a subtropical wetland. (Data from Davis (1989) *Sawgrass and cattail production in relation to nutrient supply in the Everglades*. Sharitz and Gibbons (Eds.). U.S. Department of Energy Conference No. 8603101, held in Charleston, South Carolina; National Technical Information Service: Springfield, Virginia, pp. 325–341.)

## Stem Density

The stem density of wetland plants is important because the resistance to water flow is determined in part by stem density. Only a small fraction of the ultimate plant density is planted in a new wetland. Planting densities range from 1,000–40,000 plants per hectare (0.1–4.0 plants per $m^2$), depending on the rate of spread of the selected plant species and the acceptable timeframe for plant establishment. Through vegetative reproduction, these plants will eventually spread to much greater densities.

Tanner reported 1,400–1,500 stems per $m^2$ for *Schoenoplectus tabernaemontani* growing in dairy wastewater (Tanner, 2001a), and over 2,000 stems per $m^2$ for *Schoenoplectus validus* (Tanner, 1994). In contrast, stem counts for *Scirpus acutus* in the Sacramento, California, project were typically only 150 per $m^2$ in secondary effluent, although accompanied in some cases by 15–30 per $m^2$ *Typha latifolia* plants (Nolte and Associates, 1997; 1998a).

Cattails generally have many fewer stems per unit area than bulrushes. For instance, the discharge area at Houghton Lake, Michigan, had 71 ± 23 per $m^2$ for *Typha latifolia*, and 89 ± 22 per $m^2$ for *Typha angustifolia*. A nutrient-poor location at the same wetland had only 35 ± 22 per $m^2$ for *Typha latifolia*. Glenn *et al.* (1995) measured 140 per $m^2$ for *Typha domingensis* in northern Mexico. *Phragmites australis* has comparable numbers in secondary reedbeds, 70–100 per $m^2$ in the United Kingdom (Daniels and Parr, 1990; as referenced by Cooper *et al.*, 1996). However, *Phragmites australis* grows to higher densities in warm climates, around 250 per $m^2$ in Australia (Hocking, 1989a).

Hydraulic modeling has therefore adopted similar stem density numbers. For instance, Nepf *et al.* (1997) used stem (cylinders) densities of 200–2,000 per $m^2$ in constructed flume experiments, to represent *Juncus roemerianus*. Hall and Freeman (1994) studied hydraulics in constructed flumes,

with bulrush plants, at densities of 400 and 800 per $m^2$. In laboratory flumes, Schmid *et al*. (2004b) used 12.8 stems (cylinders) per $m^2$ as representative of *Typha latifolia*.

### Submerged Area

Since microbial transformations within a FWS wetland are largely a function of area available for biofilm growth, the creation of surface area by emergent aquatic plants and associated leaf litter is an important contribution to the treatment process. One method to assess the relative contribution of the plants is to measure the amount of submerged surface area available per area of wetland (submerged specific surface area). For instance, a waste stabilization pond would have a specific surface area of 1.0 $m^2/m^2$ as the only wetted surface area is the bottom of the pond. Specific surface areas for wetlands are higher, averaging 2.8 $m^2/m^2$ at depth 30 cm for various species (Table 3.6). The depth dependence of specific surface is nearly linear (U.S. EPA, 1999).

The reader is cautioned that submerged area differs markedly from the leaf area index (LAI), the latter being commonly used in studies of photosynthesis and transpiration. LAI measures the total area of leaves in the air above water. Under most normal depths of operation, the large majority of leaf area will be above water. For instance, *Scirpus* leaves were measured to have LAI of 5.3–6.5 $m^2/m^2$, and *Typha* of 4.1–5.5 $m^2/m^2$ at the Sacramento, California wetlands (Nolte and Associates, 1998a).

### Underwater Porosity

The actual detention time in a FWS wetland is the wetland water volume divided by the volumetric flow rate. In turn, the actual water volume is less than the bathymetric value, because submerged stems take up space. The literature contains pronouncements of appropriate estimates ranging from 0.65 (Reed *et al*., 1995) to 0.95 (Kadlec and Knight, 1996). Porosity depends upon stem density and stem size. For cylindrical stems, the relationship is:

$$\varepsilon = 1 - \eta \frac{\pi D^2}{4} \tag{3.2}$$

where

$D$ = stem diameter, m
$\varepsilon$ = porosity fraction
$\eta$ = stem density, no. per $m^2$

For instance, at the Houghton Lake, Michigan, site, there were 96 *Typha latifolia* stems per $m^2$, and the mean stem diameter in the 30 cm depth was 1.2 cm. The cylinder porosity was therefore 99%. As may be confirmed from Equation 3.2, it is only when there are large numbers of stems of large diameter that porosity drops below 95%, for example, more than 100 per $m^2$ at diameter 2.5 cm. Such extreme sizes and densities are uncommon, but may be encountered in warm climates. For instance, Hocking (1989a; 1989b) reports stem densities of 250 per $m^2$, and basal diameters of one cm may be inferred from his data for *Phragmites australis* in a nutrient-rich warm climate. The corresponding cylindrical porosity is 96%.

In many circumstances in FWS, topographical "blockage" is more important than vegetative wet volume exclusion (see Chapter 2).

### Root Penetration

Early literature on HSSF wetlands contained much emphasis on the importance of root penetration depth and its effect on treatment (U.S. EPA, 1993f; Reed *et al*., 1995). The perception was that some wetland plants would have greater rooting depths, and hence provide more radial oxygen loss to conduct aerobic processes in the rhizosphere. It is indeed true that plants differ in their rooting profiles in relatively clean water, but it is now known that rooting profiles do not differ much among species in nutrient-rich waters (see Chapter 2, Figure 2.29). Roots are predominantly in the upper 20–30 cm of the media in both HSSF and FWS wetlands.

**TABLE 3.6**
**Submerged Surface Area in Ponds, and Wetlands at Depth 30 cm**

| Treatment System | Vegetation | Submerged Area ($m^2/m^2$) |
|---|---|---|
| Waste stabilization pond | None | 1.0 |
| Water hyacinth pond | *Eichhornia crassipes* | 2.2 |
| Arcata, California | *Scirpus acutus* | 4.5 |
| Arcata, California | *Typha latifolia* | 2.0 |
| Benton, Kentucky | *Scirpus cyperinus* | 3.1 |
| Benton, Kentucky | *Typha latifolia* | 2.1 |
| Houghton Lake, Michigan | *Typha latifolia* | 2.1 |
| Houghton Lake, Michigan | *Typha angustifolia* | 2.7 |
| Pembroke, Kentucky | *Scirpus validus* | 2.7 |
| Pembroke, Kentucky | *Typha angustifolia* | 3.2 |

*Note:* Litter and basin side walls are excluded.

*Source:* Data from U.S. EPA (1999) *Free water surface wetlands for wastewater treatment: A technology assessment*. EPA 832/R 99/002, U.S. EPA Office of Water: Washington D.C. 165 pp.; and Khatiwada and Polprasert (1999a) *Water Science and Technology,* 40(3): 83–89.

## 3.3 LITTERFALL AND DECOMPOSITION

Over the life cycle of a vascular plant, all plant tissues are either consumed, exported, or eventually recycled back to the ground as plant litter. Litterfall and the resulting decomposition of organic plant material are ecologically important functions in wetlands, and contribute to the cycling of nutrients and pollutants.

### Litterfall

Wetland plant tissues fall at variable rates depending on the survival strategy of the individual plant species. Herbaceous

plant species typically recycle the entire aboveground portion of the plant annually in temperate environments. The growth season may vary from ten or more months in subtropical regions to less than three months in colder climates. Also, most herbaceous species lose a fraction of living leaf and stem material as litter throughout the growing season, so there is a continuous rain of dead plant tissues throughout the year with seasonal highs and lows of litterfall. Woody plant species also participate in this production of plant litter through a natural pruning of small branches throughout the annual period. In the northern hemisphere, large amounts of flowers are shed during the spring, and leaves and fruiting bodies are lost during the fall.

Most herbaceous wetland plants do not directly fall to the wetland floor after senescence and death. Instead, plants remain in an upright stance until meteorological conditions cause them to topple. Wind, rain, and especially weight of snow, cause the standing dead material to fall. Terminology varies, and so dead material is sometimes called litter, regardless of whether it is upright or not. At other times, a distinction is drawn between standing dead and prone material called litter.

## Decomposition

Decomposition generally refers to the disintegration of dead organisms into particulate form (or detritus), and the further breakdown of large particles to smaller and smaller particles, until the structure can no longer be recognized and complex organic molecules have been broken down into $CO_2$, $H_2O$, and mineral components (Mason, 1977). In wetland studies, the term decomposition is mostly confined to the breakdown and subsequent decay of dominant macrophytes, which leads to the production of detritus. Most net annual aboveground production of wetlands is not consumed by herbivores but decomposes on the wetland surface. Rates of decomposition vary in wetlands and the fate of materials released and adsorbed during decomposition depends on the physical and chemical composition of material as well as environmental conditions at the site of decomposition (Vymazal, 1995).

Studies of litter decomposition are very numerous in the literature. Techniques for such studies have been compiled in books (Barlocher *et al*., 2005). Most of these studies have been concerned with aboveground plant parts.

The decomposition of litter and resultant release of nutrients involve at least two processes (Godshalk and Wetzel, 1978a) An initial loss of soluble materials is attributed to abiotic leaching (Boyd, 1970; Gosselink and Kirby, 1974; Godshalk and Wetzel, 1978b; 1978c; 1978d). This process is quite rapid and accounts for the majority of mass reduction during the early stages of decomposition. Leaching occurs very quickly under both aerobic and anaerobic conditions with most of the water-soluble organic substances being released within a few days. The rapid initial release of nutrients by leaching has been documented in many marsh plants—up to 30% of nutrients are lost by leaching alone during the first few days of decomposition (Vymazal, 1995). In submerged and floating-leaved plants, leaching accounts for up to 50% loss of dry matter within the first two to three days. Released nutrients may be incorporated into the protoplasm of decomposer organisms where activities such as respiration and denitrification account for additional nutrient losses (Mason and Bryant, 1975).

Flooding in wetlands has been found to increase the litter decomposition rate through physical leaching of inorganic and organic compounds from the plant tissues (Day, 1989; Whigham *et al*., 1989) and by providing habitat for aquatic microbes and invertebrates, which are important mediators in this process. However, if flood waters are anaerobic, biological activity is greatly reduced (Tupacz and Day, 1990) and only the leaching mechanisms and anaerobic respiration will occur.

## Patterns of Weight Loss

Chemical analysis of plant material reveals different rates of decomposition for different components of the plant material (soluble components, cellulose, hemicellulose, and lignin), and that rates of decomposition of each component change over time, such that the specific rate of decay for each fraction decreases as decomposition proceeds (Moran *et al*., 1989). The initial sharp drop in necromass is followed by a decline to an undecomposed residual. The initial drop is typically of the order of 10–20% for soft-tissue emergent macrophytes (Table 3.7).

The residual of recalcitrant substances is on the order of 5–20%, as inferred from long-term accretion studies. Rarely are decomposition studies continued to the point where such residuals can be determined. This is in major part due to the length of time required, as well as to the limitations of measurement techniques. An example of a litter residual is shown in Figure 3.14.

When these features are considered in combination, a modified first-order loss equation results:

$$\left(\frac{M - M^*}{M_o - M^*}\right) = A\exp(-kt) \tag{3.3}$$

where

$A$ = fraction remaining after initial leaching
$k$ = mass loss rate coefficient, $d^{-1}$
$M_o$ = initial mass, g
$M$ = mass remaining, g
$M^*$ = residual mass remaining, g
$t$ = time, d

In the vast majority of literature studies, the value of $M^*$ is chosen to be zero; and the value of $A$ is selected to be unity. There is then only one parameter to consider, the lumped mass loss rate coefficient, and under these special circumstances, it is here denoted by $k_1$. Chimney and Pietro (2006) provide rates of litter decomposition of 140 different wetland plant varieties (Table 3.8). Mean first-order rate coefficients ($k_1$) for emergent macrophyte leaf litter decomposition

**TABLE 3.7**
**Initial Weight Loss for Submerged Litter in Treatment Wetlands**

| Site | Wetland | Water | *Typha* | *Scirpus* | Data Source |
|---|---|---|---|---|---|
| Sacramento, California | | | | | Nolte and Associates (1998a) |
| | 1A | WW | 0.01 | 0.03 | |
| | 1B | WW | 0.15 | 0.35 | |
| | 7A | WW | 0.03 | 0.90 | |
| | 7B | WW | 0.21 | 0.56 | |
| | 9A | WW | 0.00 | 0.82 | |
| | 9B | WW | 0.17 | 0.14 | |
| | **Mean** | **WW** | **0.10** | **0.47** | |
| Sacramento, California | | | | | Nolte and Associates (1998a) |
| | 5A | Control | 0.14 | 0.00 | |
| | 5B | Control | 0.00 | 0.00 | |
| | LC3 | Control | 0.15 | 0.10 | |
| | LC4 | Control | 0.18 | 0.16 | |
| | **Mean** | **Control** | **0.12** | **0.07** | |
| Léon, Spain | | | | | Alvarez and Becares (2006) |
| | Winter | WW | 0.14 | — | |
| | Summer | WW | 0.15 | — | |
| Theresa Marsh, Wisconsin | | | | | Puriveth (1980) |
| | — | Runoff | 0.09 | 0.11 | |
| Houghton Lake, Michigan | | | | | Kadlec (1989) |
| | — | WW | 0.14 | — | |
| | — | Control | 0.06 | — | |

*Note:* WW = wastewater; values were determined by data fitting.

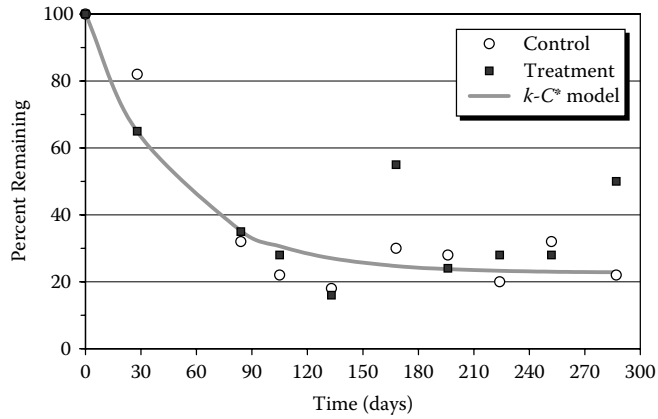



**FIGURE 3.14** Leaf litter decomposition in treatment and control wetlands at Thibodeaux, Louisiana. Species were *Fraxinus pennsylvanica, Salix nigra, Taxodium distichum, Nyssa aquatica,* and *Acer rubrum*. Two outliers removed for modeling. (Data from Rybczyk *et al*. (2002) *Wetlands* 22(1): 18–32.)

**TABLE 3.8**
**Summary of Lumped Loss Rate Coefficients for Herbaceous Plants in Various Wetlands**

| Species | Data Sets N | Mean $k_1$ ($yr^{-1}$) | Median $k_1$ ($yr^{-1}$) | Mean Half-Life ($d^{-1}$) |
|---|---|---|---|---|
| All submersed species | 107 | 17.3 | 10.2 | 15 |
| All floating species | 80 | 13.9 | 8.9 | 18 |
| All emergent species | 280 | 3.03 | 0.80 | 83 |

**TABLE 3.9**
**Values of the Lumped Loss Rate Coefficients for *Typha* in Various Treatment Wetlands**

| Species | Location | Mean $k_1$ ($yr^{-1}$) | Half-Life ($d^{-1}$) |
|---|---|---|---|
| *Typha* wastewater | Sacramento, California | 0.71 | 356 |
| *Typha* control | Sacramento, California | 0.82 | 308 |
| *Typha* wastewater, summer | Leon, Spain | 1.57 | 161 |
| *Typha* wastewater, winter | Leon, Spain | 0.73 | 347 |
| *Typha* wastewater, average | Leon, Spain | 1.15 | 220 |
| *Typha* runoff | Theresa Marsh, Wisconsin | 0.70 | 361 |
| *Typha* wastewater | Houghton Lake, Michigan | 0.50 | 506 |
| *Typha* control | Houghton Lake, Michigan | 0.71 | 356 |
| *Typha* runoff | ENRP, Florida | 1.72 | 147 |

averaged 1.4 $yr^{-1}$ for 32 studies of *Phragmites australis*, 1.7 $yr^{-1}$ for 23 studies of 10 *Scirpus* species, and 1.4 ± 0.9 for 72 studies of 8 *Typha* species. Variability for a single plant across studies is not great (Table 3.9). The half-life of the litter is equal to $0.693/k_1$.

Litter decomposition is largely mediated by vertebrates, invertebrates, and microbes living in wetlands. New litter is typically conditioned by fungi and bacteria before it is shred into smaller particles by aquatic macroinvertebrates (Merritt and Lawson, 1979). The activity of these organisms is conditioned by temperature, and therefore a temperature effect on decomposition is to be expected. Studies by Alvarez and Becares (2006) confirm this effect, as a differential in rates in summer and winter (Table 3.9). It is also true that warmer climates show higher rates of litter decomposition on an

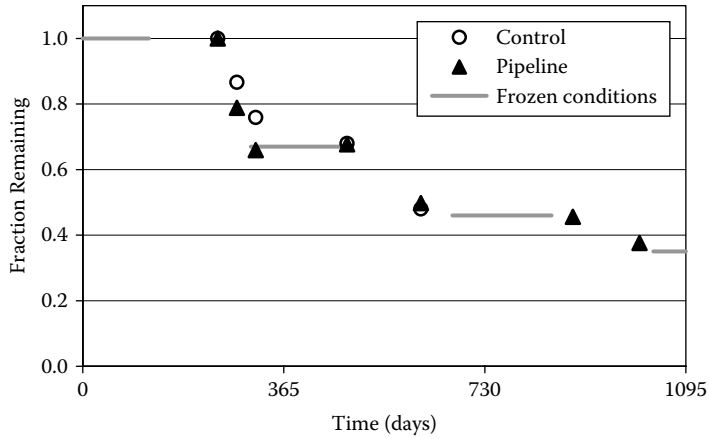



**FIGURE 3.15** Decomposition of cattail (*Typha* spp.) litter in wastewater and control areas of the Houghton Lake, Michigan, wetland. Material was placed on September 2. Freeze-up occurs around November 1, and thaw around May 1. No weight loss occurred during frozen conditions.

annual basis. However, the effect of frozen winter conditions typically interrupts the decay processes, which effectively come to a halt in frozen water and soils (Figure 3.15). Therefore, part of the variability across data sets has to do with this winter-season shutdown.

### Combined Effects of Successive Cohorts

Research has in general focused on the fate of a particular cohort of necromass, placed in a porous bag and isolated from other materials in the wetland. However, the litter layer in the wetland is the result of many such cohorts that accrue over the years, and the decomposition processes that reduce each of them over time. A conceptual model of this successive accrual and decomposition is shown in Figure 3.16, for the case of startup of a new wetland. As a simple example, consider litter which has a half-life of one year, being deposited once per year in cold temperate climate. At the end of year one, a fresh "crop" of litter of mass $M_o$ is present. At the end of year two, half that remains, and another crop of $M_o$ is added, with the total now being 1.5 $M_o$. A bit of arithmetic shows that, after a period of some years, this process will lead to an end-of-season litter crop that is twice the annual litterfall. It will take five years to build the litter to 97% of the final value. Of course, events are not so simple in a real situation, but this conceptual model serves to illustrate that a wetland has considerable "memory" via the process of litter accumulation and decomposition.

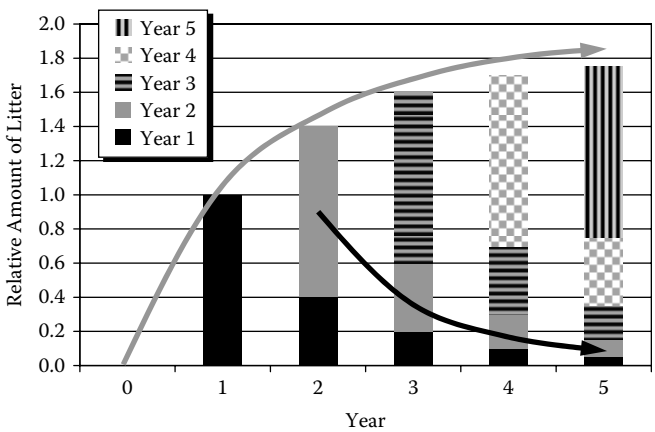



**FIGURE 3.16** Cohorts of litter accumulate and decompose during the course of time. (From Rybczyk *et al*. (2002) *Wetlands* 22(1): 18–32. Reprinted with permission.)

### Belowground Decomposition

Roots and rhizomes also undergo mortality and decomposition. Asaeda and Nam (2002) found a mean half-life of 1.2 years for *Phragmites* rhizomes of age greater than one year. Hill (1987) found 1.84 years for below ground cattail (*Typha angustifolia*). Sharma and Gopal (1982) reported 75% loss of *Typha elephantina* rhizomes decomposed in six months, in India (half-life 0.25 years). Tanner (2001a) found 1.5–2.0 years half-life for rhizomes of soft stem bulrush (*Schoenoplectus tabernaemontani*). Prentki *et al.* (1978) reported 1.5–2 years for *Typha* rhizomes and at least three years for *Phragmites* rhizomes. The fraction of this necromass which contributes to below ground soil accretion has not been determined. It seems probable that most root-rhizome necromass is recycled and only a small fraction ultimately contributes to an underground residual soil accretion. However, the rates of decomposition are slower than for aboveground litter, and therefore the belowground litter crop is much more than double the annual belowground production. It also takes much longer for the belowground litter standing crop to develop.

### Thatch

In especially hot and arid climates, treatment wetlands can accumulate excessive quantities of dead plant biomass,



**FIGURE 3.17** Thatch at the Tres Rios Hayfield wetland near Phoenix, Arizona. This standing dead material is over two meters in height, and has totally blocked light penetration to the wetland water.

regionally referred to as *thatch*. This accumulation results from the low decomposition rates occasioned by lack of water to support decomposer organisms, plus the upright orientation of the necromass, which keeps the material in the air rather than in the water. The high productivity of the litter, coupled with slow decomposition, leads to very large standing crops of standing dead thatch (Figure 3.17). Mechanical harvesting may be used to remove standing dead aboveground material (combing or thinning) or both dead above- and belowground (thatching) (Nolte and Associates, 1998b; Thullen *et al*., 2002). Controlled burning is one alternative to remove excess plant biomass in wetland treatment systems, although ash produced by burning will reintroduce nutrients into the water column. This can potentially cause a short-term decrease in treatment efficiency. Burning has been implemented at sites that permit such activities. Since accumulated plant necromass can regenerate, the benefits of removal are only temporary (Thullen *et al*., 2002).

In contrast, in cold climates the presence of standing dead material provides an excellent adjunct to insulation. Firstly, the standing material protects the wetland soil or water surface from direct exposure to the wind. This wind-break function is probably secondary to the function of catching snow, often to a depth of a meter or more. Snow is held up on dead plants, creating a zone of air spaces interlaced with plant material and captured snow (Figure 3.18). This composite is an excellent thermal insulator, and often prevents freezing in vegetated natural wetlands at times when water is deeply frozen. This function is served for both FWS and HSSF wetlands.

The litter layer on top of a HSSF wetland bed functions as mulch. Such a layer also provides air spaces and holds up the snow to form an insulating layer for the SSF bed.

### Mineral Constituents of Litter

The chemical composition of litter is not fixed during decomposition. Carbon and macronutrients (N, P, Ca, K) may be depleted or amplified at differential rates. Decomposer



**FIGURE 3.18** Standing dead wetland plants capture snow and provide thermal insulation.

organisms utilize chemicals from both water and the litter, and then contribute to the overall biomass of the litter. For example, the rate of concentration increase may exceed the rate of necromass loss, thereby creating an increase in the mass of a constituent (Figure 3.19). The additional chemicals are acquired from the wetland water. In other situations, there can be a mass loss of chemicals accompanying the loss of necromass (see, for instance Kulshreshtha and Gopal, 1982; Corstanje *et al*., 2006). There appears to be no universal pattern for the time series of litter chemical composition in natural or treatment wetlands (Chimney and Pietro, 2006).

## Accretion

Wetland ecosystems are often sites of long-term positive net primary productivity (NPP), and develop accumulations of buried organic matter in the form of peat and eventually coal. This net accumulation of organic matter is primarily because of the reduced metabolic rate of microbes in flooded wetland sediments compared to metabolic rates in well aerated, upland soils. When living and dead plant material sinks to the level of anaerobic sediments, it is protected from abundant free oxygen and from the higher rates of degradation typical of an oxygenated system.

Therefore, not all of the dead plant material undergoes decomposition. Some small portions of both aboveground and belowground necromass resist decay, and form stable new accretions. The amount of such accretion has been quantified in only a few instances for free water surface wetlands (Craft and Richardson, 1993; Reddy *et al*., 1993; Rybczyk *et al*., 2002), although anecdotal reports also exist (Kadlec, 1997a; Sees, 2005; Wang *et al*., 2006a). Quantitative studies have relied upon either atmospheric deposition markers (radioactive cesium or radioactive lead) or introduced horizon markers, such as feldspar or plaster. Either technique requires several years of continued deposition for accuracy.

The manner of accretion has sometimes been presumed to be sequential vertical layering (Kadlec and Walker, 1999; Rybczyk *et al*., 2002), but that view is likely to be overly simplified. At least two factors argue against simple layering:

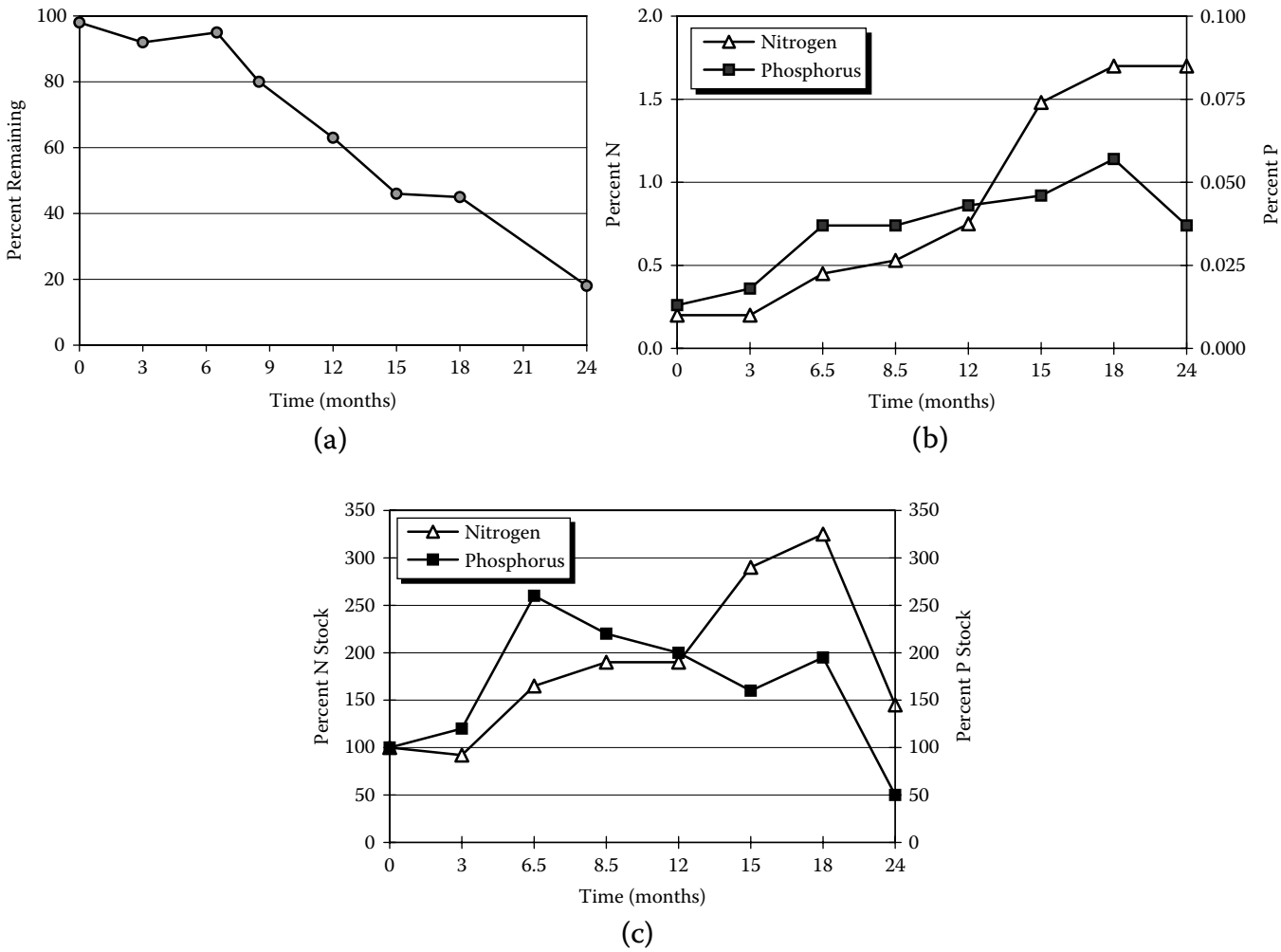



**FIGURE 3.19** Changes in amount of culm litter (a), nitrogen (N), and phosphorus (P) content (b), and N&P stock (c) for *Phragmites* over a two-year period of decomposition. (From Gessner (2000). *Aquatic Botany* 66(1): 9–20. Reprinted with permission.)

vertical mixing of the top soils and sediments (Robbins, 1986), and the injection of accreted root and rhizome residuals at several vertical positions in the root zone. Nonetheless, new residuals are deposited on the wetland soil surface, from various sources. The most easily visualized is the litterfall of macrophyte leaves, which results in top deposits of accreted material after decomposition. However, algal and bacterial processing that occurs on submersed leaves and stems results in litterfall and accretion of micro-detrital residuals.

The net result of undecomposed residuals is the buildup of new sediments and soils in the treatment wetland. These residuals are composed of both undecomposed plant parts and the remains of organisms that have caused the decay. The rate of such buildup is often in the range of 0.1–2.0 cm/yr.

### Background Concentrations

Wetland systems are dominated by plants (autotrophs), which act as primary producers of biomass. However, wetlands also include communities of microbes and higher animals, which act as grazers (heterotrophs) and reduce plant biomass. Most wetlands support more producers than consumers, resulting in a net surplus of plant biomass. This excess material is typically buried as peat or exported out of the wetland (Mitsch and Gosselink, 1993). This net export results in an internal release of particulate and dissolved biomass to the water column, which is measured as nonzero levels of BOD, TSS, TN, and TP. These wetland background concentrations are typically denoted by the term $C^*$. Enriched wetland ecosystems (such as those treating wastewater) are likely to produce higher background concentrations than oligotrophic wetlands because of the larger biomass cycling resulting from the addition of nutrients and organic carbon. Even land-locked wetland basins, which only receive water inputs through precipitation, will have nonzero background concentrations. Rainfall and dryfall contain these same substances, and therefore contribute to background concentrations.

Background concentrations are achieved when wetland inflows and outflows contain the same (low) levels of constituents. That situation typically occurs far from the inflow sources of those compounds for flow through systems, and at long times for batch systems exposed to doses of the compounds. Because of random wetland processes, background concentrations may fluctuate markedly around a mean time average value. Atmospheric deposition, uptake, and return processes are in balance (Figure 3.20). The first-order areal model for pollutant removal will be described in detail in Chapter 6, but here the ramifications of decomposition processes are briefly explored. The mass balance for background conditions is:

$$QC_i - QC_o = 0 = (kC^* - R - PC_p)A \tag{3.4}$$

where

$A$ = wetland area, $m^2$
$C^*$ = wetland background concentration, mg/L
$C_p$ = atmospheric deposition concentration, mg/L
$C_i$ = inlet concentration, mg/L
$C_o$ = outlet concentration, mg/L
$k$ = removal rate coefficient, m/d
$Q$ = flow rate, $m^3/d$
$P$ = rain rate, m/d
$R$ = return rate from decomposition, $g/m^2 \cdot d$

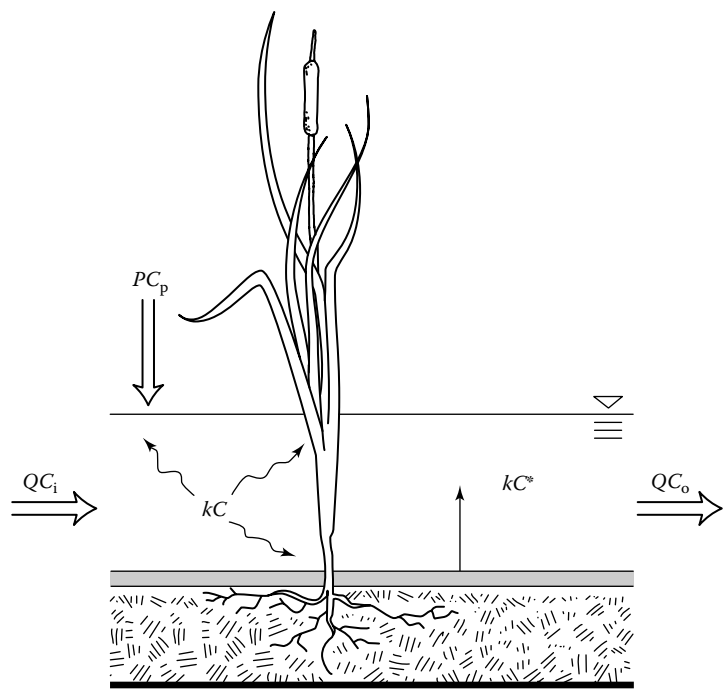



**FIGURE 3.20** The background concentration is determined by processes far from inflow effects in a flow through wetland. In that situation, $C_i = C = C_o$.

As a result, the background concentration is that required for a balance between uptake and the combination of atmospheric deposition and return flux from decomposition:

$$C^* = \frac{(R + PC_p)}{k} \tag{3.5}$$

The return fluxes for dissolved organics (BOD) and organic nitrogen are often quite large, and result in $C^* \approx 5$ mg/L and 1.5 mg/L, respectively. On the other hand, phosphorus, nitrate and ammonia are utilized by a variety of biota, and uptake often far exceeds the return flux, resulting in $C^* \approx 0$ mg/L. These values, and methods for determination, will be discussed in more detail in later chapters, by compound.

### Wastewater Stresses

Plants living in FWS and SSF treatment wetlands may be subjected to a different set of conditions than plants in natural wetlands. If the application is for domestic wastewater polishing, the incoming water quality is often as good or better than most natural wetlands. The same is true for many remediation applications, in which the chemical targets do not particularly influence nutrients or wetland biogeochemical cycling. Likewise, applications for drinking water conditioning, and crop and urban runoff treatment, do not push the boundaries of wetland water quality environments. Even if the water quality is nonthreatening, treatment wetlands have water level controls, which may be inadvertently set at water levels that are detrimental to the selected or existing wetland plants. Many wetland plants prefer water depths of less than 40 cm, and most also prefer intermittent rather than continuous flooding. Relatively stable water levels, rather than seasonal and rain-driven hydrologic regimes, may place stress on wetland vegetation. The hydrologic requirements of wetland plants are a design consideration (see Part II).

However, treatment of primary domestic wastewaters, food and animal waste, acid mine waters, and leachates, and sludge consolidation, all may create unusual and stressful water quality conditions for wetland plants. The conditions that may be created by strong wastewaters include:

- High influent oxygen demand, which leads to reducing conditions (low redox potential) in the water column and in the wetland root zone
- High nutrient loadings, which lead to increased production of plant biomass and detritus, and subsequently to a high internal oxygen demand
- High sulfur, leading to sulfide toxicity
- Extraordinarily high or low pH
- High salinity, created by large dissolved salt concentrations

### Oxygen Deficiency

Under primary or secondary domestic wastewater loading, the influent BOD, nitrogen, and phosphorus are typically much higher than in natural wetlands. Due to the additional oxygen demand from the wastewater, there is generally little or no dissolved oxygen in the FWS water column. The nutrient loadings increase biomass production, which in turn increases the amount of decaying plant material in the detritus layer. These two effects create a strongly reducing (highly anaerobic) sediment layer, and anaerobic soils beneath. The chemical gradient between the oxygen in the root tissue and the sediment is greater, leading to increased oxygen losses from the root tissue (Sorrell and Armstrong, 1994; Cronk and Fennessy, 2001). Wetland plants may develop a thick, waxy coating on mature root and rhizome tissue. However, on the newly growing root hairs (especially at the root tip), oxygen can be easily transferred from the root to the sediment due to the thinness of the cell walls.

Wetland plants attempt to minimize this oxygen loss by preferentially rooting in the uppermost sediment layers, where the least reducing conditions are present (Lockhart, 1999). Under extreme conditions, rooting may preferentially occur in the water (adventitious roots). Under oxygen deficiency, emergent plants can tolerate less flooding; typically the maximum allowable water depth for a given plant species subjected to wastewater loading is less than half of that for the same species in an oligotrophic wetland environment.

Plants living in HSSF wetlands are subjected to stresses similar to FWS wetlands, but additionally possess a relatively hostile rooting environment. Unless very fine sands or soils are used, the capillary action and moisture holding capacity of the bed media is much less than that of natural wetland sediments. Plant root networks must be submerged in order to survive (submersion is especially important during plant establishment). For HSSF systems receiving primary (septic tank) effluent, a strongly reducing (highly anaerobic) environment will develop in the bed matrix. The required nutrient supply is overabundant, and extensive, deep rooting is not necessary to acquire nutrients. Wetland plants respond by preferentially rooting in the uppermost bed layers and by reducing the overall root biomass (Lockhart, 1999). This limited root penetration can create preferential flow paths through the lower section of the gravel bed (Breen and Chick, 1995; U.S. EPA, 2000a; Whitney *et al.*, 2003). Root penetration to the bottom of the bed is likely to occur only in systems that receive low-oxygen demand waste (e.g., a nitrified influent), or have some other means of supplemental oxygen transfer (Behrends *et al.*, 1996; Lockhart, 1999).

### Sulfide Toxicity

Lamers (1998) documents that sulfate has negative effects on the growth rate of *Carex nigra*, *Juncus acutiflorus,* and *Gallium palustre*, at concentrations of 64 and 128 mgS/L. Koch and Mendelssohn (1989) report that 32 mgS/L of sulfide produced negative effects in *Panicum hemitomon* and *Spartina alterniflora*. The presence of sulfide is coupled with anaerobic conditions in the root zone, but the effects of

sulfide go beyond mere anoxia (Koch and Mendelssohn, 1989). Hydrogen sulfide apparently inhibits the activity of alcohol dehydrogenase, thereby limiting the ability of plants to avail themselves of alternative anoxic energy pathways. This effect was confirmed by measuring a reduced $^{15}N$ uptake rate in the presence of sulfide. However, the availability of free sulfide is strongly mediated by the presence of iron, because of the formation of iron sulfides.

Phytotoxicity was found to be very serious at the 45 mgS/L level in *Phragmites australis* (Armstrong *et al.*, 1996). These authors found that aeration pathways became blocked, interfering with the diffusive connection to the atmosphere, and thus reducing the plant's ability to oxygenate the rhizosphere. Smolders and Roelofs (1996) found, for *Stratiotes aloides*, an aquatic macrophyte characteristic of mesotrophic freshwater marshes, that levels of 320 mgS/L were toxic to the roots. Lamers *et al.* (2002) found root parts growing in 1.7–3.4 mgS/L of sulfate into the peaty sediment, clearly showed sulfide toxicity by becoming black, slimy, and unfit for nutrient uptake from the sediment. Free sulfide could not be detected in the surface water. They concluded that only roots in the surface water would survive. *Nuphar lutea* did not propagate in the sulfate-treated enclosures. However, the sensitivity of a wetland plant species to free sulfide not only depends on the actual sulfide levels in the rhizosphere, but also on detoxification mechanisms such as radial oxygen loss.

### Extreme Salt Content and/or pH

Acid mine drainage wetlands often operate with incoming pH less than 5, which is commonly regarded as a lower limit for aquatic resource protection (U.S. EPA, 2006), and pH 6.5 is preferred. Although there are many plants that can tolerate low pH, the diversity of treatment wetlands operating under extremes will be constrained. Indeed, natural northern bogs commonly have pH less than 5, as a result of the decomposition processes and conditions that prevail. Likewise, high pH is found in other situations, such as leachates from waste material piles from the phosphate and soda ash industries, and from construction debris. There are natural wetlands with high pH, including prairie potholes and playas in the United States. Again, there are many plants that can tolerate high pH, but the selection for these alkaline treatment wetlands will be limited.

There are major differences between the species of plants that inhabit saline wetland environments and those that live in freshwater wetlands. Treatment wetlands are almost always utilized for fresh waters, but high salt content is sometimes a feature of the incoming water. Species such as *Typha* and *Phragmites* are tolerant of a wide range of salinity, and will do well in environments with high TDS. However, some plants normally inhabit saline or brackish water, including, for example, *Spartina* spp. (cordgrasses) and *Juncus maritimus* (seaside rush). The reader is referred to the vast literature on the characteristics of salt marsh plants if a high salinity treatment wetland is contemplated.

## 3.4 VEGETATIVE COMMUNITIES IN TREATMENT WETLANDS

### Algal Systems

### Periphyton

Natural Everglades periphyton-dominated wetlands exist and function at phosphorus levels below 10 ppb. Constructed wetlands dominated by periphyton, termed periphyton stormwater treatment areas (PSTAs), have also been successful in closely approaching the 10 ppb goal in small units. Periphyton-based STAs (PSTA) and submerged aquatic vegetation (SAV) wetlands are variants on the same theme: shallow submersed aquatic vegetation that supports an active periphyton community. Both envision sparse emergent vegetation that forms an anchor and a substrate for the periphyton. Emergent vegetation must be very sparse to avoid shading of the algae, which occur in three forms: on the bottom, as floating mats, and as attached growth on submerged plant parts (Figure 3.21). The benthic mats can access residual phosphorus in the sediments and recycle accreted phosphorus. PSTA envisions sparse vegetation that forms an anchor and a substrate for periphyton. Emergent vegetation must be very sparse, if present at all, to avoid shading of the algal mats, which occur on the bottom as floating mats, and as attached growth on submerged plant parts. Accretion of residuals is needed to make this a passive sustainable process. The benthic mats can access such residuals and recycle accreted phosphorus.

It should be recognized that periphyton treating water of concentration greater than about 10 ppb would not be pristine Everglades periphyton. Extensive research has shown that pristine cyanobacterial mats do not survive at concentrations above that limit. That research shows that at higher concentrations, the periphyton contains a significant proportion of green algae. At some higher phosphorus concentration,



**FIGURE 3.21** A periphyton-dominated wetland contains sparse emergents, for protection and anchoring. The algal mats may be either floating or resting on the bottom.

approximately 50 ppb, the existence of any kind of self-sustaining, algal-dominated system is threatened.

There have been eleven constructed projects in South Florida; and, supplemented by natural system response studies, form an impressively large suite of datasets (Kadlec and Walker, 2003). The two largest of these constructed systems are 40 ha in extent.

### Algal Turf Scrubbers

Algal turf scrubbers are channels with shallow water flow, vegetated by filamentous algae. These have been utilized in Asia (Kim *et al*., 2002), Europe (Schumacher and Sekoulov, 2002), and North America (Adey *et al*., 1996; Craggs *et al*., 1996a; 1996b). The performance of algal biofilm processes is comparable with suspended algae systems. The algae grow rapidly in nutrient rich water, and adhere to available surfaces. Harvest is a necessity, else the biomass begins to slough, and effectiveness is lost. Therefore, the success of this technology is very much dependent upon the infrastucture used to support the organisms. The organisms may include individuals or mixtures of green algae (*Stigeoclonium, Oedogonium, Ulothrix, Scenedesmus*, *Spirogyra*), blue-green algae (*Oscillatoria, Lyngbya*), and diatoms.

## Submerged Plants

Submerged aquatic vegetation (SAV) such as waterweed (*Elodea* spp.), coontail (*Ceratophyllum* spp.), and naiads (*Najas* spp.) have been used to treat wastewater (Gumbricht, 1993a; 1993b). These submerged plants have parts suspended in the water column, and are sometimes rooted in the bottom sediments. Typically, their photosynthetic parts are in the water column, but certain species may grow to where their photosynthetic parts are at or just above the water surface. This category of constructed FWS wetland has not had widespread usage, but submerged plant species are present in many natural treatment wetlands, and are invaders in other constructed wetlands.

Examples are presented here to illustrate usage of this type of treatment wetland vegetation.

### Secondary Wastewater Treatment, Australia

An experimental trench, 4 × 100 m, was established and used for two years (1984–1986) to treat secondary municipal water (Bavor *et al*., 1988). The trench contained 100% cover of parrot feather (*Myriophyllum aquaticum*). Parrot feather is regarded as a "mostly" submersed plant (Collins *et al*., 2005), but with floating parts under some circumstances. Four other trenches contained emergent plants in varying proportions. The *Myriophyllum* trench had the poorest performance.

### River Treatment, Sweden

The submersed macrophyte treatment system at Snogeröd, Sweden was put in operation in 1988, and operated until 1991 (Gumbricht, 1993a). The 1.2-ha wetland contained *Elodea canadensis* and *Cladophora glomerata*, and was operated at a depth of 0.6 m and a flow of 2,400 $m^3$/d. The incoming water had TN = 9.8 mg/L and TP = 0.26 mg/L; wetland effluents averaged TN = 7.5 mg/L and TP = 0.07 mg/L. The conclusion from this field-scale project was that submersed macrophyte systems have the potential of polishing river waters and pretreated wastewaters. Gumbricht (1993b) went on to speculate that harvest could be used to improve performance.

### Municipal Wastewater Polishing, Netherlands

A treatment wetland system on the island of Texel in the Netherlands was constructed in 1994 to polish 3,400 $m^3$/d of effluent from a 45,000 PE municipality (Toet *et al*., 2005). The surface flow wetland had a total water surface of 1.3 ha. The STP effluent first entered a presettling pond, was then divided over nine parallel ditches, after which it was collected in a discharge ditch and discharged to surface water. The first half of eight ditches was 0.2 m deep and contained *Phragmites australis* or *Typha latifolia*, while the second half was 0.4 m deep and contained submerged aquatic macrophytes (*Elodea nuttallii*, *Ceratophyllum demersum*, and *Potamogeton* spp.). The SAV portion of the system removed essentially no phosphorus, because of high loading rates, but did reduce nitrogen by 45%.

### Agricultural Runoff, Florida

Cell 4 of the Everglades Nutrient Removal Project (ENRP) was a 147-ha constructed wetland that developed into a SAV system, by virtue of herbiciding competing emergents (Figure 3.22). Emergents were spot-sprayed, which required relatively small quantities of chemicals. For example, the sum of all herbicide applications in 1998 averaged only 3.0 L/ha (SFWMD, 1999a; Dierberg *et al*., 2002). The use of herbicides as a feature of treatment wetland operation and maintenance is perhaps unique to south Florida, and was exercised for the first time in the ENRP.

Recognizing the good performance of SAV in cell 4, phosphorus removal was investigated in mesocosms stocked with a mixture of taxa common to the region: *Najas guadalupensis*, *Ceratophyllum demersum*, *Chara* spp. and *Potamogeton illinoensis* (Dierberg *et al*., 2002). After eight months of operation, *N. guadalupensis* dominated the standing crop biomass and phosphorus storage. The mean inflow TP concentration of 107 µg/L was reduced to 52, 29, and 23 µg/L in the 1.5, 3.5, and 7.0 day HRT treatments, respectively.

As a result of these research and demonstration projects, SAV was specified for the outlet sections (about 50%) of all the stormwater treatment wetlands (STAs), by virtue of state law. Conversion of the outlet wetland cells is underway at the time of this writing, and been completed in large measure. Approximately 8,000 ha of SAV constructed wetlands will result.

## Floating Plants

Floating aquatic vegetation (FAV) treatment systems consist of one or more ponds in which one or more species of water tolerant, floating vascular plants are grown. The shallower depths and the presence of floating aquatic macrophytes



**FIGURE 3.22** Coots enjoy the submerged aquatic vegetation in Cell 4 of the Everglades Nutrient Removal Project, a 147-ha constructed wetland in Florida.

in place of algae are the major differences between these aquatic treatment systems and stabilization ponds. The presence of plants is of great practical significance because the effluent from aquatic systems is often of higher quality than the effluent from stabilization pond systems with no floating plants, for equivalent detention times. Floating aquatic plant wetlands are described in detail by DeBusk and Reddy (1987), Crites and Tchobanoglous (1998), and Crites *et al.* (2006). Their major application has been in the tropics and subtropics.

In FAV systems used for municipal wastewater, the carbonaceous biochemical oxygen demand (CBOD) and suspended solids (SS) are removed principally by bacterial metabolism and physical sedimentation. In systems used to treat CBOD and SS, the plants themselves bring about very little actual treatment of the wastewater. Their function is to provide components of the aquatic environment that improve the wastewater treatment capability and/or reliability of that environment. In aquatic treatment systems designed to remove nutrients (N and P), plant uptake can contribute to the removals, especially where plants are harvested frequently.

The principal floating plant species used in aquatic treatment systems are water hyacinth (*Eichhornia crassipes*), pennywort (*Hydrocotyle* spp.), and duckweed (*Lemna* spp.). These, and other floating species such as water lettuce (*Pistia stratiotes*) and mosquito ferns (*Azolla* spp.), may occur in any FWS wetland. Water hyacinths have been used in a variety of experimental and full-scale systems for treating wastewater (see, for instance, Reddy and Smith, 1987). The use of water hyacinths has been limited in geographic location to warm weather regions because of the sensitivity of water hyacinth to freezing conditions. Duckweed systems have been developed in colder climates because of the greater temperature tolerance of duckweed species. Both duckweed and water hyacinth systems have most often been used for either removing algae from oxidation pond effluents or for nutrient removal following secondary treatment.

Floating plants have their photosynthetic parts at or just above the water surface with roots extending down into the water column. Nutrients are taken up from the water column through the roots. These roots provide an excellent support medium for the growth of bacteria and for the filtration/adsorption of SS. Root development is a function of nutrient availability in the water and growth rate of the plant. Thus, in practice, the density and depth of plant roots will be affected by pretreatment, and by other factors affecting plant growth rate such as temperature and harvesting. With floating plants, the penetration of sunlight into the water column is reduced and the transfer of gas between water and atmosphere is restricted. As a consequence, floating plants tend to keep the wastewater nearly free of algae and anaerobic or nearly so.

In this book, designed FAV systems are regarded as a modification of pond or lagoon treatment technology, rather than a variety of wetland. That appears to be the decision of much of the literature on FAV systems; however, some authors classify them as wetlands, for instance, Nahlik and Mitsch (2006). FAV systems are an alternative to FWS emergent marshes and SSF systems under appropriate circumstances. They provide a better opportunity for harvesting, but are difficult to maintain in large cells.

Floating plants can invade FWS wetlands that were not designed to include such vegetation, and therefore some examples are included here.

### Volunteer FAV

Floating plants are easily able to invade open water zones of FWS wetlands. Systems in the southern United States, for example, are susceptible to larger plants, such as water hyacinths (*Eichhornia crassipes*), water lettuce (*Pistia stratiotes*), and pennywort (*Hydrocotyle* spp.), while northern systems typically experience duckweed (*Lemna* spp.).

### Duckweed

Probably the most common floating plant in constructed treatment wetlands is duckweed (*Lemna* spp.). It colonizes with great ease, and is geographically widespread. It has been advocated as a treatment system in itself (see, for example, Smith and Moelyowati, 2001; Körner *et al*., 2003; Ran *et al*., 2004), but requires a retaining grid to prevent wind-driven drift of the plants. Small patches of open water in FWS wetlands, such as intentional deep zones and muskrat eat-outs, are very often covered with duckweed (Figure 3.23).

### Pennywort

The Cobble wetlands at Tres Rios, Arizona, passed through a startup period of approximately one year, after which they operated in a stable vegetation mode for approximately two years. But in the spring of 1998, the planted wetland vegetation (bulrushes) died entirely (Kadlec, 2006b). The cause of the demise of the selected plants is not definitively known. Subsequently, the wetland was reconfigured as braided channels connecting former deep zones. The wetland then underwent a period of regrowth, and floating aquatic plants colonized, and pennywort (*Hydrocotyle ranunculoides*), eventually creating near-complete cover (Figure 3.24).

### Water Hyacinths

In warm climate FWS wetlands, water hyacinths and water lettuce are ready invaders of open water areas. For example, water hyacinths invaded most of cell 1B of STA5 of the Everglades Protection Project wetlands, and overgrew the intended SAV in the 494-ha wetland. As suggested above, hyacinths are not necessarily a bad alternative for treatment, but physical problems occurred in this case. Because of the large size of the cell, the wind fetch was an unobstructed three kilometers. Even a modest westerly (prevailing) wind caused the floating hyacinths to drift to the outlet, and jam outlet structures. A strong wind caused vegetation to pile up to depths of about a meter, half abovewater. These windrows created water backup, and badly interfered with hydraulic operation of the wetland. The hyacinths were controlled by herbiciding (SFWMD, 2004).

### Floating Mats and Rafts

Floating islands or mats are widespread vegetation formations that occur in all climatic regions of the globe (van Duzer, 2004). These range in character from the floating sedge fens of Alaska (Racine and Walters, 1991) to the papyrus swamps of equatorial Africa (Gaudet, 1977; Kansiime and Nalubega, 1999). For example, very large areas of the Mississippi River delta wetlands are floating mat systems (Sasser *et al*., 1996), comprising over 70% of the western Terrebonne Basin.

There are at least three natural formation mechanisms (Clark and Reddy, 1998):

1. The delamination and floating of unvegetated organic substrates from deeper sediment. Germination of plants occurs after emergence. This is a peat float-up process.
2. The rhizomes of aquatic plants colonize the water surface from a nucleus of aquatic vegetation that is either unattached or expanding from the shore. This is the grow-over process.
3. Units of rooted vegetation and substrate split simultaneously from the bed, and float to the wetland surface. This is a mat floating process.

Floating mats must be almost entirely organic in order to be buoyant enough to float. They derive their buoyancy from gas spaces in rhizomes (Hogg and Wein, 1987; 1988; Krusi and Wein, 1988), and also from gases generated by decomposition processes. However, floating plant mats may also be artificially fostered in aquatic or wetland systems, by use of rafts of one sort or another.

A distinction is drawn between treatment systems that contain floating plants and those that contain floating mats. If the plants can normally float as individuals, without any



**FIGURE 3.23** *Lemna* filled all open water areas in the Lake Nebagamon, Wisconsin, constructed wetland during the startup grow-in period.



**FIGURE 3.24** An unintended cover of pennywort (*Hydrocotyle* spp.) in Cobble Cell 1 (0.92 ha) at Tres Rios, Arizona.

support from a substrate or their neighbors, then the system is a floating plant system. Well-known examples of constructed floating plant systems are water hyacinths (*Eichhornia crassipes*), water lettuce (*Pistia stratiotes*), duckweed, (*Lemna* spp.) and water fern (*Azolla* spp.). In contrast, a very much larger category of plants may be established in floating systems in which supporting media and neighbors are required. A total of 67 different plants have been tested in Hungary (Lakatos, 1998).

Here we are concerned about two aspects of floating mat systems: their unintended development in treatment wetlands, and the intentional design of floating plant mats for wastewater quality improvement. Interestingly, such systems are not often considered as a constructed wetland design option.

## Unintended Floating Mats in Treatment Wetlands

In several instances, treatment wetlands have developed floating mats of vegetation, which were unplanned and unexpected. A few illustrative examples are given here. Treatment has sometimes continued to be effective; but has been impaired in other cases. Of course, the water flows under such mats, rather than over and through the litter layer. This is a major difference from the common marsh–overland flow system. Floating mat systems may be more akin to floating plant systems, such as water hyacinths, but no direct comparisons have been done.

### Kis-Balaton, Hungary

The constructed shallow treatment impoundments (wetlands) on the Zala River, as it enters Lake Balaton, function for nutrient removal. The original vegetation of the second unit consisted mainly of reed beds (*Phragmites*). As a consequence of routing the river, the reed beds were damaged. Dead rhizomes produced gas, which buoyed fragments of the reed bed to the surface as a floating mat. These mats formed a matrix for secondary succession. These floating islands are partly attached to the still-living fragments of reed beds rooted in the sediment, and have an approximate diameter of about 15 m. The approximate rhizome mat thickness of the floating islands was 0.5 m in 2001. The islands were characterized by willows (*Salix cinerea*), sedges (*Carex* spp.), cattail (*Typha angustifolia*), and ruderal species (e.g., *Bidens cernuus*) (Somodi and Botta-Dukat, 2004).

### Kinross and Houghton Lake, Michigan

These two natural wetlands developed near-monocultures of cattails on preexisting peatlands. Over the course of time, these *Typha* communities became floating mats (Kadlec and Bevis, 1990). A possible cause was the retreat of the root zone to a smaller biomass located high in the soil profile, compared to prior conditions. This physical effect, coupled with possible partial peat dissolution into the less acidic added wastewater, led to a 50-cm soil-free water zone topped by the floating mat. The mats are closely woven beds of roots, rhizomes, and sediments (Figure 3.25). These had enough strength to permit foot and small all-terrain vehicle travel on the mat. Treatment continued to be generally effective under the mat, except that early phosphorus additions to the Kinross system were later in a bleed-back mode (see discussion on woody plants)

### Lake Apopka, Florida

The constructed marshes at Lake Apopka, Florida, developed into floating mats (Stenberg *et al.*, 1998). Different vegetation strategies were employed, and all underwent significant conversion to floating mats, over the period 1990–1995. In 1995, 73% of natural succession areas contained floating vegetation mats, while 55% of planted sites were floating.



**FIGURE 3.25** A section through the floating mat at the Houghton Lake, Michigan, treatment wetland. The cattail plants have been clipped about 15 cm above the mat, which is about 30 cm thick. Note the black sediment in the rhizome and root matrix.

Belowground biomass declined as roots and rhizomes shifted to floating vegetation mats.

### STA1W, Florida

Cell 2 of Stormwater Treatment Area 1W (STA1W) was constructed on peatlands formerly in agriculture. Cattails developed early in the project life, comprising about 50% cover of the 413-ha wetland. Changes in the mode of operation of this cell after six years of operation caused greater water depths, and the cattail areas separated from the base substrate and became floating mats. Because there was less than 100% cover, these floating islands moved with the wind. The water was shallow enough that portions of the mat bottom scraped the base substrate below, creating suspension of the base soils. The water in the cell was therefore very turbid, and the associated nutrients were exported from the wetland. Performance was severely impacted. The cell was later reconfigured, and partially converted to submerged aquatic vegetation.

## Floating Mat Constructed Wetlands

Floating mat wetlands have been intentionally employed at many sites, and in great variety. In general, they do not employ SSF substrates, but may employ floating matrices for plant support. In terms of performance, these will later be included in the category of free water surface wetlands, although in fact the mat covers the water surface. The float method has several potential advantages:

- It directly takes up nutrients from water column, and does not remobilize the nutrients in sediment.
- It enables the use of a diversity of aquatic plants.
- It could be used in any water body, regardless of its depth and bottom characteristics.
- Biomass harvesting is theoretically easy.

The principal drawback for raft systems is cost: the frames are expensive. Support structures for the mat are quite varied in design. Several ideas have been patented (Balogh, 1982; Ishikawa and Mizuno, 1988; Hondulas, 1994), and complete units are commercially available (van Duzer, 2004). Here the vegetative character of such constructed mats is described via examples.

### New Zealand and Australia

The plant *Glyceria maxima* is capable of being established as a floating mat without the assistance of any support frames. Work at Hamilton, New Zealand, at mesocosm and pilot scale, showed that *Glyceria* mats could be excised in sections from existing treatment wetlands, and floated on the water in a new treatment wetland (Van Oostrom and Russell, 1994).

Wen and Recknagel (2002) implemented polyethylene foam floats, planted with parrot feather (*Myriophyllum aquaticum*), water couch (*Paspalum paspalodes*), and waterbuttons (*Ranunculus repens*). The intent was to treat irrigation drains, fields, or treatment ponds in order to eliminate dissolved phosphorus. Phosphorus removal rates in the range of 0.043–0.086 g/m$^2$·d were measured as bioaccumulation in plant tissues.

### United Kingdom and Europe

Hiley (1990) reports that "raft lagoon" systems were built at Highroyd, Bishop Wilton, Pattrington, Yorkshire, United Kingdom. These were supported by a buoyant geotextile of 5 cm mesh, and contained a variety of plants for testing purposes. Cattails (*Typha*) worked well, as did sweetflag (*Acorus*), marsh marigold (*Caltha*), and bentgrass (*Agrostis*). However, *Iris, Nuphar,* and *Spartina* were unsuccessful. Small-scale trials indicated that *Phragmites* and *Phalaris* would be good raft candidates.

London's Heathrow airport pilot tested floating rafts of both *Typha* and *Phragmites*, at the scale of 6 × 7 m, operated with detention times of less than one day. Subsequently, a 1.2-ha floating raft wetland was built, and planted with *Phragmites*. The design detention time was just over one day (Revitt *et al*., 2001; Richter *et al*., 2003; see Chapter 13).

Artificial floating meadows have been piloted near Budapest, Hungary (Lakatos, 1998). Rectangular wooden frames were filled with plastic netting, and planted with a wide variety of plants. Sixty-seven species were tested, of which 20

either died or did not do well, and 47 grew normally or better. Species found to be suitable included *Alisma, Glyceria, Sagittaria, Sparganium,* and *Typha*. Removals ranged from 40% for phosphorus to 98% for oxidized nitrogen.

The School of Agricultural Engineering of Madrid, Spain, has developed several applications of floating mat systems, which are termed *floating macrophyte filter* (FMF) systems. Systems have been installed at the communities Aviles, Coy, and Doña Inés; at a pig farm in Lorca, Spain (Figure 3.26); as well as at single-family residences. Plants that have been used are *Phragmites*, *Sparganium*, *Schoenoplectus (Scirpus)*, *Iris pseudocorus,* and *Typha*. *Typha* species have shown the best results, with high growth and treatment rates (Curt *et al*., 2005).



(a)



(b)

**FIGURE 3.26** Floating macrophytes filter on the pig farm owned by the cooperative Sociedad Cooperativa GAMUR (Ganaderos de Murcia) in Lorca, Spain. (a) Start-up with plants in floating containers. (b) System after partial grow-in. (Photos courtesy Fundación Global Nature: http://www.macrophytes.info/galeria_imagenes.html.)

### North America

A variety of floating platform wetlands have been used in the United States, mostly on an experimental, pilot basis. For example, an open water area in a peatland near Madison, Wisconsin, was covered with a matrix of logs, leaf bales, and wire mesh (Hefty, 2002). Planted species included pickerelweed (*Pontederia cordata*), bulrushes (*Scirpus acutus, Scirpus fluviatilis*), burreed (*Sparganium eurycarpum*), and arrowhead (*Sagittaria latifolia*). Muskrats ate the river bulrush, pickerelweed, and arrowhead. Floating rafts were installed in Lake Mead, Nevada, for the purpose of improving water quality (Boutwell, 2001). Cattails (*Typha domingensis*) were found to be more successful than various bulrushes (*Scirpus* spp.).

The removal of metals from mine waters may lead to accumulations in sediments that could be dangerous to sediment-foraging organisms. The use of floating systems allows the accumulation to occur in deep-water locations, as a result of processes in the root zone that drop metal-laden materials. Such systems have been implemented in several locations in Canada, including Buchanans, Newfoundland, Sudbury, Ontario, and Kitimat, British Columbia (Smith and Kalin, 2001). Frames were constructed from timber and snow fencing, buoyed by extruded polystyrene (XPS), and planted with narrow-leaved cattail (*Typha angustifolia*). These systems have been found effective for suspended solids removal.

## Woody Plants

Many natural freshwater wetlands contain a variety of woody species. In the southern United States, swamps commonly contain cypress (*Taxodium*), gum (*Nyssa*), and swamp white oak (*Quercus*). In northern North America, species include white cedar (*Thuja*), spruce (*Picea*), red maple (*Acer*), willow (*Salix*), and alder (*Alnus*). However, forested wetlands have only rarely been constructed, and then in tropical or subtropical climates. This is probably more due to the perception of a long grow-in period, rather than to any potential deficiencies in treatment capability. Greenway and Bolton (2001) suggest that this is possibly an oversight:

> Little attention has been given to the use of tree species as candidates for constructed wetlands and yet, woody species may have additional advantages such as higher nutrient uptake, higher rates of primary productivity, higher nutrient storage capacity (biomass sink potential), lower maintenance (due to greater tree longevity) and the production of useful resources. Harvesting biomass for resources also removes the accumulated nutrients, which could be recycled through mulch.

At present, the applications of forested constructed wetlands are in the following principal categories:

- Melaleuca (tea tree) systems
- Mangrove systems
- Willow systems
- Forested edges or bands in stormwater wetlands

The data from constructed woody wetlands is too sparse to analyze with the same degree of thoroughness as emergent marshes. The beginnings of a performance basis are discussed here.

### Melaleuca

The genus *Melaleuca* is found in tropical and subtropical climates. It is comprised of 250 species, and was formerly prolific in the lowlands of Australia. It is a modern-day widespread nuisance invader in Florida. It can form a dense stand that crowds out virtually all other species. It is also known as "paperbark" because of its soft, paper-like bark, and some varieties are economically valuable because of the oils that they produce. It has been shown that these trees may be fostered in constructed wetlands, and provide excellent treatment capability (Bolton and Greenway, 1997; Bolton and Greenway, 1999a; 1999b; Greenway and Bolton, 2001). *Melaleuca alterniflora* grew rapidly, and accumulated about 5,000 gdw/m$^2$ in a 21-month irrigation period with secondary effluent (Bolton and Greenway, 1997). The trees may be harvested, with regrowth occurring if a dry-down period is included.

### Mangroves

Mangroves are one of the few woody species that can tolerate saltwater environments. There are many species, including *Kandelia candel*, *Avicennia marina*, and *Rhizophor*a spp. in Asia; and *Avicennia germinans* and *Rhizophora mangel* in tropical South America. Integrated mangrove-aquaculture systems are currently practiced throughout Asia, including the traditional *gei wai* in Hong Kong and *tambak* in Indonesia (Primavera, 2000). Shrimp pond effluents are treated in pond-mangrove systems, most often involving natural mangrove stands. This practice has been criticized as causing degradation of the mangroves (Gautier, 2002). The use of natural mangrove stands has not been particularly effective in Colombia (Gautier *et al.*, 2001). Natural mangrove stands were studied at Shenzhen, China, for the purpose of treating settled sewage, and survived a moderately heavy loading (Wong *et al.*, 1997). Constructed (transplanted soils and plants) mangrove wetland mesocosms were effective in controlling metals in applied sewage (Tam and Wong, 1997). Boonsong *et al.* (2003) studied a 1.5-ha wetland planted with four species of mangroves, for the treatment of dilute, settled sewage. Good removals were found for many common wastewater parameters, at three and seven days' detention, over a year-long study. The authors cautioned that the effects on the plants remained to be assessed.

### Willows

Willows (*Salix* spp.) grow rapidly and are water-loving plants. An entire subset of treatment wetland technology has developed, in which the transpiration capabilities of willows is used to create zero-discharge wastewater treatment systems, primarily for single residences and small communities (Gregersen and Brix, 2001; Brix and Gregersen, 2002; Brix and Arias, 2005). Many willow systems are now functioning in Denmark (Figure 3.27).

Principal features are zero discharge and recycle of part of the nutrients via harvested willow biomass. Danish guidelines have been published (Gregersen *et al.*, 2003). Willow facilities generally consist of 1.5-m-deep high-density polyethylene-lined basins filled with soil and planted with clones of willow (*Salix viminalis* L.). The surface area of the systems depends on the quantity and quality of sewage to be treated and the local annual rainfall. For a single household in Denmark, the area needed typically is between 120 and 300 m$^2$. Settled sewage is dispersed underground into the bed under



**FIGURE 3.27** A willow treatment system at Pileanlag, Denmark. (Photo courtesy C. Arias.)

pressure. The stems of the willows are harvested on a regular basis to stimulate the growth of the willows and to remove some nutrients and heavy metals.

The total annual water loss from the systems is assumed to be 2.5 times the potential evapotranspiration at the location as determined by climatic parameters, and is partially compensated by precipitation. In small systems, the vegetation experiences enhanced evaporation from the "oasis effect," resulting from warmer and dry air flowing across an area of plants. In addition, there is also the "clothes-line effect," where the vegetation height is greater than that of the surroundings and may increase evaporative loss. Therefore, evapotranspiration from isolated expanses, on a per unit area basis, may be significantly greater than the calculated potential evapotranspiration.

One third or one half of the willows are harvested every year to keep the willows in a young and healthy state with high transpiration rates.

## Woody Plants in Stormwater Wetlands

Wetland-tolerant trees have been used as an internal landscaping feature in some urban stormwater wetlands. For instance, cypress (*Taxodium*) was used in the Greenwood urban wetland in Orlando, Florida, and red maple (*Acer*) in the Tollgate urban wetland in Lansing, Michigan.

## Wastewater and Natural Forested Wetlands

Natural wetlands have been used for wastewater treatment in modern times, including forested systems. These wetlands are to some degree engineered to accommodate and treat incoming wastewater. In some cases, forested wetlands have accommodated wastewater with only small effects on the preexisting ecology, while in other, more numerous cases, the original ecology has been severely disrupted. In many cases, woody species have not survived, including both shrubs and trees. Replacement communities are often soft tissue herbaceous plants, notably *Typha* spp. Forested northern bogs have not provided good long-term treatment, and have been markedly altered by wastewater additions (Guntenspergen *et al.*, 1980; Nichols and Higgins, 2000).

The water quality performance of forested wetlands can differ markedly from that of emergent marshes, submerged vegetation, and floating plant systems. For instance, Kadlec and Knight (1996) reported that the phosphorus removal rate constant for 63 emergent marsh wetlands averaged 13.1 m/yr, while for 11 natural forested wetlands, it averaged 3.1 m/yr. In contrast, the removal of nitrate in forested wetlands is quite good, averaging 96 ± 5% for five systems in the southeastern United States (Boustany *et al.*, 1997; NADB database, 1998).

This book does not focus on such systems, but the reader may wish to consult the appropriate literature if faced with the need to evaluate them. Accordingly, a few brief examples from the available literature are provided here.

### Northern North America

Forested wetlands abound in Minnesota, Michigan, and the Canadian provinces, and some have been the recipients of treated wastewater. Monitoring has been conducted at several sites. Based upon results, few if any new projects have been permitted in the past three decades.

### Kinross, Michigan

Prior to 1977, the Kincheloe Air Force Base was in full operation with an estimated population of 7,500 people. The plant treated an average wastewater flow of 2,300 $m^3$/d with primary clarifiers, trickling filters, and secondary clarifiers, followed by gravity discharge to the adjacent 300-ha wetland (Kadlec and Bevis, 1990). Approximately one third of this area was impacted by the discharge. Nitrogen removal was effective in this wetland, with about 99% reduction in both ammonia and oxidized nitrogen. $BOD_5$ and TSS were reduced 64% and 94%, respectively. Phosphorus removal was complicated by the time sequence of wastewaters and treatment. The base used very large quantities of phosphate detergents to wash airplanes, and the original wastewater treatment plant (WWTP) did not remove phosphorus. An unknown but large phosphorus load was delivered to the wetland and presumably partially trapped. Subsequently, the base closed, and the WWTP added phosphorus removal. Therefore, low-phosphorus water entered the wetland, which released considerable phosphorus and caused outflows higher than inflows.

It is probable that this peatland was at one time a shallow lake basin that had filled and developed into a palustrine acid/peat wetland system typical of the region. The original wetland probably contained sedges (*Carex* spp.), leatherleaf (*Chamaedaphne calyculata*), sphagnum (*Sphagnum* spp.), and sparse black spruce (*Picea mariana*), based on remnants and adjacent ecosystems. The wastewater promoted a shift to a monoculture of cattails (*Typha latifolia*). Peat in the near discharge area largely disappeared, leaving a floating mat of cattails over ooze. A trend to a cattail monoculture occurred, and the vicinity of the discharge now consists of a cattail monoculture surrounded by upland forest. Small, isolated duckweed ponds dot the cattail stand. Occasional remnant living and dead black spruce "islands" are present. Aerial photos from 1939, prior to wastewater addition, show no evidence of cattail (Kadlec and Bevis, 1990). The gradual expansion of the *Typha* monoculture from the discharge point downgradient continued up to 1981, at which time the entire watercourse had converted to cattail. Figure 3.28 illustrates this change-over process for another system at Biwabik, Minnesota.

### Bellaire, Michigan

The wetland is peat-based, and vegetated by white cedar (*Thuja occidentalis*), spruce (*Picea* spp.), and black ash (*Fraxinus nigra*), grading to a sedge-shrub community containing *Carex* spp., alder (*Alnus* spp.), red osier dogwood (*Cornus*



**FIGURE 3.28** Cattails (foreground) intruding on a spruce forest at Biwabik, Minnesota, after 15 years of effluent addition. The lagoon discharge is in the extreme foreground, and water flows toward the forest in the background.

*stolonifera*), and poison sumac (*Rhus vernix*; Kadlec, 1983). The wetland receives groundwater discharge from the adjacent sand hills. In the early 1970s, a sewage collection system was put in place, bringing the wastewater to a double-facultative lagoon treatment facility. Wastewater was discharged from the lagoons, in spring and again in fall into forested lake margin wetlands. These intense, high volume discharges flushed debris from the wetlands to the lake, and passed very quickly without substantial treatment. In 1976, the discharge was altered to occur throughout the entire unfrozen season, and was spread more uniformly across the wetland. In 1982, the discharge was moved to secondary locations. In 1989, the treatment process was upgraded to include sand/carbon filtration and alum addition, and the discharge was moved to a third set of locations.

Early in the project life, both nitrogen and phosphorus were reduced to low values. However, phosphorus removal became ineffective after several years (Kadlec, 1983). During the recovery period 1982–1987, the wetland then received only the groundwater discharge. The wetland removed 1,097 kg of phosphorus over the six-year period of controlled water discharge (1.02 $g/m^2 \cdot yr$), resulting from a decrease from 2.71 to 0.29 mg/L. In the six years following termination of discharge, the wetland released 64 kg of phosphorus, or 6% of the amount removed.

The trees in the first irrigation zone were killed, probably by a combination of long hydroperiod and root zone erosion. Black ash was the most resistant to destruction. The replacement community was comprised of soft-tissue plants, dominated by *Typha* and jewelweed (*Impatiens capensis*). The second irrigation area also suffered tree destruction (Figure 3.29). The third irrigation area had distribution of water in grassed strips in the forest, and to date, the forest has survived well in that area (1989–present).



**FIGURE 3.29** Tree death at the Bellaire wetland. All of the trees in the foreground, save one, are standing or fallen dead in this July photo. They have been replaced by soft-tissue plants after ten years of wastewater addition.

### Houghton Lake, Michigan

This system is the oldest continuously monitored natural wetland treatment project in the United States (U.S. EPA, 1993f; Kadlec and Knight, 1996). Wastewater from this residential community is collected and transported to two 2-ha aerated lagoons, and is then stored in a 12-ha pond for summer disposal to a 600-ha peatland locally known as the Porter Ranch Peatland. The Porter Ranch Peatland has received better-than-secondary wastewater for 30 years, since 1978, during which hydrology, water chemistry, soils, and vegetation were studied. Hydraulic loadings to the impacted area are less than 1 cm/d, and occur only in the unfrozen season. Water quality has been consistently improved by passage through the wetland. Nitrogen and phosphorus are removed to background. Phosphorus removal was found to first order with a rate constant of 11 m/yr. Five years were required to stabilize nutrient fronts (Kadlec, 1997a).

The ecosystem has changed character markedly in the zone of discharge (Kadlec, 1993). Water regime changed to a greater duration of flooding. Major community changes took place within the irrigation area. Vegetation species composition shifted from sedges (*Carex* spp.) and shrub willows (*Salix* spp.) to cattail (*Typha latifolia* and *Typha angustifolia*) and duckweed (*Lemna minor*; Figure 3.30). About 80 ha of wetland have been altered. Aboveground biomass tripled, and several centimeters of soil accreted. Some plant species—all woody species and sedge—were totally lost in the discharge area. Cattail and duckweed have totally taken over, and are increasing in abundance and range. An aspen community near the pipeline completely succumbed in 1983. A second aspen island located 500 m downgradient had also totally succumbed by 1984. The aspen on the edges of the peatland have died in backgradient and side locations where the shore slopes gradually. The alteration of the water regime was the probable cause of tree death along much of the wetland perimeter, in a band up to 50 m wide at a few locations. Long-dead timber at these locations indicates that similar events may have occurred naturally in the past.

### Southern United States

Pioneering research on wastewater irrigation to cypress domes was conducted by the University of Florida in the 1970s (Ewel and Odum, 1984). Since that time, and based in major part upon that research, guidelines and regulations have been developed at both the state and national level (see, for instance, U.S. EPA, 1985a; Schwartz, 1989).

### North Myrtle Beach, South Carolina

Pretreated wastewater from aerated lagoons is distributed to 69-ha Bear Bay, a natural forested wetland, through a series of gated aluminum pipes supported on wooden boardwalks (U.S. EPA, 1993f). The vegetation of Bear Bay is dominated by tree species, including sweet gum (*Liquidambar styraciflua*), red maple (*Acer rubrum*), pond pine (*Pinus serotina*), and loblolly pine (*Pinus taeda*). Operation of the full-scale system began in October 1990. On the basis of an estimated area of 28 ha, treated wastewater flows are equal to annual average hydraulic loading rates between 0.2 and 0.6 cm/d. The water quality entering the wetland was partially nitrified secondary. The five-year average mass removal efficiencies in Bear Bay were at least 88% for $BOD_5$, TSS, $NH_4$-N, TN, TP, and UOD (Kadlec and Knight, 1996).

Decreases in tree canopy density were observed in areas continuously flooded with wastewater. This allowed expansion of herbaceous communities dominated by pennywort (*Hydrocotyle* spp.) and duckweed (*Lemna* spp.) (Kadlec and Knight, 1996). This phenomenon was visually evident near the treated wastewater discharge area due to early and prolonged leaf fall and mortality of susceptible tree species including loblolly pine, sweetgum, American elm, red maple, and water oak.

### Reedy Creek, Florida

The 35-ha Reedy Creek natural wetland was vegetated with water ash (*Fraxinus caroliniana*), magnolia (*Magnolia virginiana*), and blackgum (*Nyssa sylvatica*), and received 12,000 $m^3$/d of better-than-secondary pretreated wastewater (Knight *et al.*, 1987; Kadlec and Knight, 1996). Removal efficiencies were 60–80% for $BOD_5$, TSS, and TN, but less than zero for TP, over a 12-year period of record, 1978–1989. There was not before-and-after sampling of the biological communities. However, in spite of about ten years of wastewater discharges, the wetland still supported a diverse and robust forested wetland plant community in 1988. Stem density and basal area were both high at 3,785 stems per hectare and 38.29 $m^2$/ha, respectively, and are typical of mature southern coastal plain swamps.

### Poinciana, Florida

The 47-ha Boot Wetland in Poinciana, Florida, was a drained and degraded forested wetland, dominated by pond cypress (*Taxodium ascendens*) and blackgum (*Nyssa sylvatica;* Martin *et al.*, 2001). In 1984 it became the recipient of 1,060 $m^3$/d of nitrified secondarily treated wastewater, which continues to the present. This is an exceedingly low hydraulic loading, amounting to less than rainfall on the wetland, and half the inflow was lost, presumably to infiltration and evapotranspiration. Concentration reductions were good for TP (69%) and TN (48%), the latter reflecting complete removal of oxidized nitrogen. Incoming $BOD_5$ (2.5 mg/L) and TSS (5.5 mg/L) were so low that there were small increases, presumably to wetland background.

This wetland was in a degraded condition prior to wastewater addition due to forestry, drainage canals, and surrounding land development. Peat oxidation had occurred, and trees were toppling. Compared to the antecedent condition, the structure and function of the system was significantly improved by wastewater irrigation (Martin *et al.*, 2001). The dominance and density of trees was increased. However, the creation of a 100% hydroperiod, and a continuous water depth of 70–90 cm, resulted in water surface

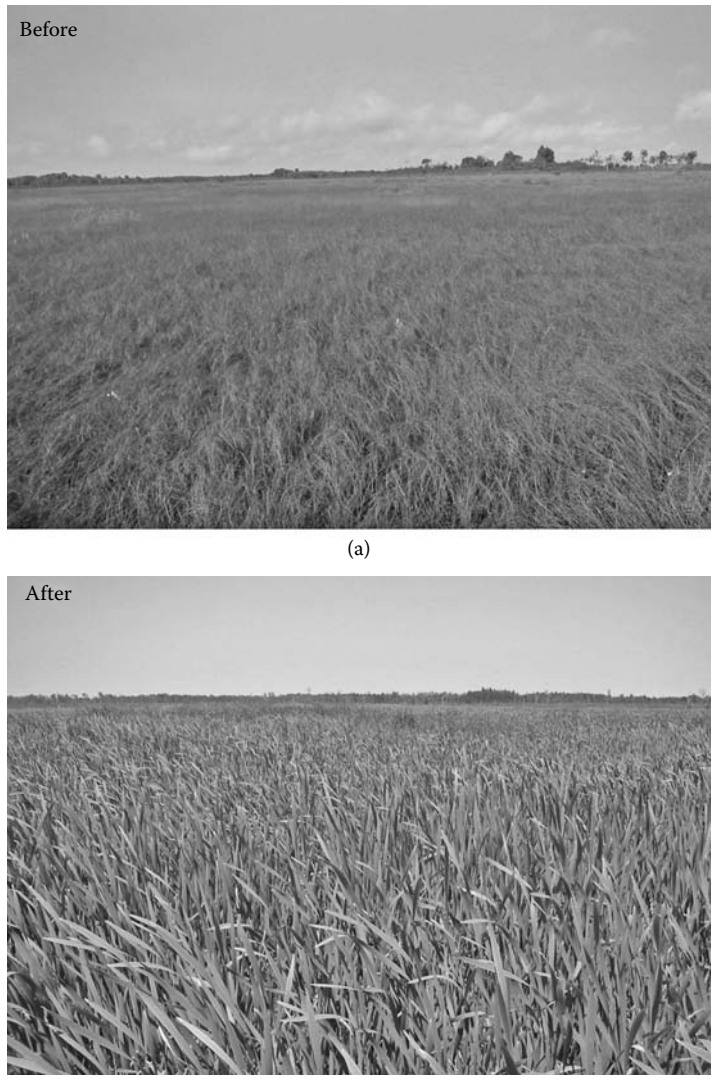



**FIGURE 3.30** Sedge meadow replacement at the Houghton Lake, Michigan, wetland. All of the sedge (a) disappeared, and was replaced by cattail (b) after wastewater addition.

cover of 100% of floating, leaved plants, including duckweed (*Lemna* spp.), frog's bit (*Limnobium spongia*), and water fern (*Salvinia rotundifolia*). Natural wetlands of the region had a much lower frequency of inundation, and consequently fostered a different understory community type, dominated by water hyssop (*Bacopa carolinianum*). Thus a healthy wetland resulted, but one with an unnatural vegetative structure.

### Thibodaux, Louisiana

The Thibodaux, Lousiana, site consists of an almost permanently flooded, subsiding, forested wetland, containing cypress (*Taxodium distichum*) and gum (*Nyssa aquatica*). Since 1992, the 231-ha wetland has received secondarily treated municipal wastewater at the average rate of 15,140 $m^3$/d. Loading amounts to about 0.27 cm/d of water, and 124 kgN/ha·yr (Boustany *et al.*, 1997). The receiving wetland had been hydrologically altered by some combination of levees, spoil banks, highways, oil and gas access roads, or railroad lines (Day *et al.*, 1999). At such low loadings, effects on the ecosystem structure are believed to be absent or at least long-delayed.

From 1992 through 1996, the mean annual reduction oxidized nitrogen, the dominant form of nitrogen in the effluent, ranged from 96% to 99%. From 1992 through 1994, the mean annual reduction of total phosphorus in the wetland ranged from 33% to 71%. High rates of accretion and burial of sediments in the subsiding system provides a permanent sink for phosphorus (Zhang, 1995). Results from several ongoing and completed studies of wastewater treatment in other forested wetlands of the region indicate that they are achieving the ecological goals of enhancing effluent water quality, stimulating vertical accretion, and increasing productivity (Day *et al.*, 2004). Economically, the savings are substantial for small communities and nontoxic industrial processors (Breaux *et al.*, 1995).

### Emergent Soft Plants

By far the largest number of treatment wetlands utilize soft tissue plants (herbaceous vegetation), as discussed in the remainder of this chapter. Emergent vegetation is the most common choice; because these plants fit a wide variety of niches in wetland ecosystems, planting stock is often available through commercial plant nurseries, and they spread through lateral rhizomes, which allows the relatively rapid development of an emergent plant canopy.

### Surface Flow Wetlands

Emergent wetland plants provide a wide range of treatment mechanisms in FWS wetlands, (Sinclair Knight Merz, 2000) including:

- Increased sedimentation by reducing wind-induced mixing and resuspension
- Additional surface area in the water column, which increases biofilm biomass and soluble pollutant uptake
- Increased surface area for particle interception
- Shade from the plant canopy over the water column to reduce algae growth
- Induced flocculation of smaller colloidal particles into larger, settleable particles

Most of these mechanisms are structural in nature. Consequently, selecting the "perfect" species is not nearly as important as establishing a functional plant canopy. As microbiological transformations within the wetland are a function of area available for biofilm growth, the creation of surface area by emergent aquatic plants and associated leaf litter is an important contribution to the treatment process. Plant species that provide structure year-round generally perform better than species that die below the water line after the onset of cold temperatures. For these reasons, fast-growing emergent species that have high lignin contents and that are adapted to variable water depths are the best suited for FWS systems. Wetland plant genera that most successfully meet these criteria include cattails (*Typha* spp.), bulrushes (*Scirpus* spp.), and common reed (*Phragmites australis*).

Vegetation types in FWS wetlands exhibit small performance differences, but these differences are often masked by other unavoidable differences in comparable wetlands. At the time of this writing, the case for superiority of a particular plant species has not been proven or disproven. The evidence points toward minimal net differences among plant species. A more diverse mix of plant species will be better able to accommodate changes in water quality and flow. In other words, a polyculture is preferable to a monoculture.

Table 3.10 shows plant species used for initial planting of FWS wetlands listed in the NADB v.2 (NADB database, 1998). That database contains only an early subset of FWS systems, and does not include many recently built systems.

**TABLE 3.10**
**Species Reported as the First Dominant in Constructed FWS Wetlands in the North American Database, Version 2.0**

| Scientific Name | Common Name | Number of Wetlands |
|---|---|---|
| *Typha* spp. | Cattail | 206 |
| *Scirpus* spp. | Bulrush | 49 |
| *Juncus* spp. | Rush | 19 |
| *Phalaris arundinacea* | Reed canary grass | 15 |
| *Phragmites australis* | Common reed | 13 |

*Source:* Data from NADB database (1998) *North American Treatment Wetland Database* (*NADB*), *Version 2.0*. Compiled by CH2M Hill, Gainesville, Florida.

### Subsurface Flow Wetlands

Compared to FWS wetlands, subsurface (SSF) systems are much less dependent on plants to sustain their treatment processes. A SSF wetland will require planting because nearby seed banks are typically lacking and the gravel media is not optimal for seed germination.

Small performance differences among vegetation types also exist for SSF wetlands (Brisson *et al.*, 2006), but since the role of plants is small in these systems, plant effects are masked by other unavoidable differences in comparable wetlands. Therefore, no conclusive results could be found in a review of 47 species studied in 27 different comparative investigations (Brisson *et al.*, 2006). Speculatively, plants that have significant root penetration into the bed media are likely to enhance treatment. Effects of plant root systems include:

- Additional surface for biofilm created by the root system.
- Oxygen diffusion from root surfaces into the water column. (However, this plant-mediated oxygen transfer is very small relative to the applied internal and external organic loadings in most SSF systems.)
- Chemical exudates used by the plants detoxify the root environment.
- Additional fungi species introduced by the plants.
- Symbiotic bacteria introduced by the plant root systems.

The combined effect of these phenomena is a larger and more diverse microbial community within the SSF bed. Comparing the results of plant investigations in different SSF wetlands does not provide compelling evidence that any particular plant species offers superior treatment (Gersberg *et al.*, 1984; DeBusk *et al.*, 1989; Van Oostrom and Cooper, 1990; Batchelor *et al.*, 1990; Knight, 1993).

Designers typically focus on plants that are easy to propagate and are able to survive in the relatively hostile

environment of a SSF wetland. The most frequently used plant species worldwide is *Phragmites australis* (common reed). This species has remarkable growth rates, root development, and tolerance to saturated soil conditions. Common reed is also known to provide some ancillary benefits in terms of wildlife habitat in the United Kingdom (Merritt, 1994).

*Phragmites* has a long history of cultural use and is used almost exclusively for wetland treatment systems in Europe. However, in many areas of the United States, *Phragmites* is considered an exotic and invasive plant species, and therefore use of this species in North America has been limited. Table 3.11 shows plant species used for initial planting of SSF wetlands listed in the NADB v.2 (NADB database, 1998). That database contains only a small and early subset of SSF systems. It does not include the current large numbers of small systems (Wallace and Knight, 2006).

## Examples of Modern Emergent Community Choices

Given the apparent lack of performance differentials, many recent wetlands have been planted with a view to biodiversity and aesthetics. In the United States, decorative plants such as canna lilies (*Canna flacida*) and yellow iris (*Iris pseudocorus*) have historically been used at a number of HSSF sites, including Denham Springs and Carville, Louisiana (Wolverton, 1989). Although it is recognized that volunteer vegetation will continue to alter these systems, they have been established as communities with a variety of compatible species. Numerous systems have been so initiated; a few examples are given here.

### FWS Systems

#### *Wakodahatchee, Florida*

The Wakodahatchee Wetlands were created by converting a series of percolation ponds into flow through marshes (Bays *et al.*, 2000; Hobbs *et al.*, 2003). The wetland area totals 16 ha of wetted acreage, with individual wetland cells ranging from 1 to 5 ha. Deep zones, consisting of ponds 2 m in depth and variable in width and oriented transverse to the direction of flow, are interspersed throughout each wetland to maximize hydraulic retention time, create aquatic habitat, and equilibrate flow. Marshes comprise about 70% of the wetland area, vegetated by native emergent, forested, and transitional wetland species designed to emulate native South Florida wetland plant communities. Emergent marsh zones are composed of bulrush (*Schoenoplectus* (*Scirpus*) *validus*, *S. californicus*), duck-potato (*Sagittaria lancifolia*), arrowhead (*Sagittaria latifolia*), spikerush (*Eleocharis cellulosa*), fireflag (*Thalia geniculata*), and pickerelweed (*Pontederia cordata*). Herbaceous species planted at the upper edge of the marsh zone include sawgrass (*Cladium jamaicense*), Fakhahatchee grass (*Tripsacum dactyloides*), and Gulf muhlygrass (*Muhlenbergia capillaris*). Forested species also planted at the marsh edge include cypress (*Taxodium ascendens*), pond apple (*Annona glabra*), Carolina willow (*Salix caroliniana*), red maple (*Acer rubrum*), and buttonbush (*Cephalanthus occidentalis*). Duckweed (*Lemna* spp.) has become naturally ubiquitous throughout the wetland, but with significant seasonal fluctuations in total cover. Treatment of the secondary influent is comparable to other systems in the region.

**TABLE 3.11**
**Typical Species Used in Constructed SSF Wetlands in the North American Database, Version 2.0**

| Scientific Name | Common Name | Number of Wetlands |
|---|---|---|
| *Typha* spp. | Cattail | 5 |
| *Scirpus* spp. | Bulrush | 55 |
| *Phalaris arundinacea* | Reed canary grass | 1 |
| *Phragmites australis* | Common reed | 9 |

*Source:* Data from NADB database (1998) *North American Treatment Wetland Database* (*NADB*), *Version 2.0*. Compiled by CH2M Hill, Gainesville, Florida.

#### *Victoria, Texas*

Wastewater from a chemical plant is treated in a train that includes a 21-ha FWS wetland (Reitberger *et al.*, 2000). The vegetation selected for wetland establishment was a varied mixture. Shallow zones included bulrushes, *Schoenoplectus* (*Scirpus*) (*actus, validus, californicus, americanus, pungens*), plus arrowhead (*Sagittaria graminea*), giant cutgrass (*Zizaniopsis miliacea*), rushes (*Juncus effusus*), spikerush (*Eleocharis* spp.), and sedges (*Carex* spp.). Deeper zones were planted with coontail (*Ceratophyllum demersum*), pondweed (*Potamogeton* spp.), pickerelweed (*Pontederia cordata*), and water lilies (*Nymphaea elegans*). The wastewater treatment process is working well, with COD removal above 99%, and complete removal of nitrate and nitrite, since start-up (Beeman and Reitberger, 2003). The wetland has exceeded expectations for polishing, buffering, and community value.

#### *Lapeer, Michigan*

The domestic wastewater from a small manufacturing facility is pretreated in a septic tank, and then discharged to a pond followed by a FWS wetland. The pond had addition of yellow water lilies (*Nuphar lutea*), and developed a fringe of cattail (*Typha* spp.). The wetland was vegetated with hardstem and softstem bulrush (*Scheoenoplectus* (*Scirpus*) *acutus* and *Scheonoplectus* (*Scirpus*) *validus*), together with sedge (*Carex lacustris*), arrowhead (*Sagittaria latifolia*), water plantain (*Alisma plantago-aquatica*), pickerel weed (*Pontederia cordata*), and giant burreed (*Sparganium eurycarpum*). Cattail was an immediate invader in the wetland. Water quality data were within the expected range.

### SSF Systems

#### *Jackson Meadow, Marine on St. Croix, Minnesota*

This conservation development uses two HSSF constructed wetlands to treat wastewater from a 64-home residential subdivision. Each 0.1-ha wetland cell is designed to treat up to



**FIGURE 3.31** Vegetative community of a HSSF wetland at Jackson Meadow, Marine on St. Croix, Minnesota. This system features a vegetative community dominated by river bulrush (*Schoenoplectus* (*Scirpus*) *fluviatilis*), an unplanted colonizer—reed canary grass (*Phalaris arundinacea*)—and New England aster (*Aster novae-angliae*).

34 $m^3$/d of domestic wastewater. Primary treatment is provided by septic tanks, and wetland effluent is infiltrated back into the soil for recharge of the surficial aquifer. Both systems have consistently met permit limits established by the Minnesota Pollution Control Agency. To protect against freezing in the cold Minnesota winters (temperatures below –30°C are possible), the wetland cells are insulated with a layer of peat mulch 15 cm thick (Wallace and Nivala, 2005). The presence of the peat mulch creates an unsaturated rooting zone that shifts the competitive advantage way from obligate wetland plants towards facultative wetland plants (U.S. Army Corps of Engineers, 1987).

The wetland cells were planted with a variety of native plants. Although the two systems are only 0.8 km apart, they were established in different years (1999 and 2002), and development of the vegetative communities has proceeded along different lines. The north treatment system features a vegetative community dominated by river bulrush (*Schoenoplectus* (*Scirpus*) *fluviatilis*), an unplanted colonizer (reed canary grass [*Phalaris arundinacea*]), and New England aster (*Aster novae-angliae*; Figure 3.31). The south treatment system features a vegetative community dominated by woolgrass (*Scheonoplectus* (*Scirpus*) *cyperinus*), giant burreed (*Sparganium eurycarpum)*, arrowhead (*Sagittaria latifolia*), and prairie cordgrass (*Spartina pectinata*). Cattail (*Typha angustifolia*) was planted in both systems but did not persist beyond the first growing season.

### *Lutsen Resort, Lutsen, Minnesota*

This lodging company operates a variety of resort properties along the North Shore of Lake Superior. Two HSSF and two VF systems have been built over the period from 1997 to 2005; the systems are small (less than 400 $m^2$), and design flows range from 10 to 80 $m^3$/d. All systems consistently meet their permit limits. The systems are insulated with a layer of peat to prevent freezing in the cold Minnesota winter. A variety of native wetland plants were used in the systems. Early projects utilized planting of dormant rhizomes in the fall. This was generally unsuccessful due to grazing pressure from whitetail deer (*Odocoileus virginianus*) over the winter months (Wallace *et al.*, 2000).

In the first system (a HSSF wetland planted in 1997) the plant community has evolved over time to include cattail (*Typha angustifolia*) and bulrush (*Schoenoplectus* (*Scirpus*) *fluviatilis*); prairie cordgrass (*Spartina pectinata*) and iris (*Iris pseudocorus*) have also become dominant (Figure 3.32).



**FIGURE 3.32** Vegetative community of a HSSF wetland at Lutsen, Minnesota. Plants in the inlet zone are dominated by arrowhead (*Sagittaria latifolia*), with cattail (*Typha angustfolia*) in the background. The vegetative community is approximately eight years old at the time of this photo.

Undesirable plant species have invaded the system over time (see the subsequent section on "weeds"), including willows (*Salix* spp.) and Canada thistle (*Cirsium arvense*), necessitating occasional management control (see Part II).

The second wetland (a HSSF system planted in 1998) has been dominated by woolgrass (*Schoenoplectus* (*Scirpus*) *cyperinus*) and green bulrush (*Schoenoplectus* (*Scirpus*) *atrovirens*). The two VF wetlands (planted in 2004 and 2005) still have juvenile plant communities dominated by biennials, such as black-eyed susan (*Rudbeckia hirta*). Due to their unsaturated flow conditions, it is likely that the VF wetlands will develop plant communities very different than the saturated flow HSSF wetlands, despite the fact that all systems are located in the same climatic conditions within 8 km of one another.

## 3.5 WEEDS

A "weed" in a constructed wetland is a plant that has not been intentionally planted and possesses one or more characteristics viewed as undesirable. Many species can quickly invade and colonize new treatment wetlands, and may be regarded as weeds, depending upon local opinion. As varied opinions exist around the world, it is not possible to generalize an overall list of "weed" species. For instance, in the Czech Republic, common reed (*Phragmites australis*) and reed canary grass (*Phalaris arundinacea*) are often used for SSF wetland systems because they are native to the country (Vymazal, 1998). In the United States, resource managers often discourage the use of these plants because they are considered as nonnative in most regions of the United States. In addition, these plants are very aggressive, and considerable effort is required to remove them from SSF systems.

The question is whether or not the weeds deteriorate the treatment efficiency of the system. It seems that in most cases where weeds occur or took over the originally planted species, the treatment effect is not hampered. The problem with herbaceous weeds is the aesthetics and, especially in the United States, the occurrence of unwanted not-native species. The problem with woody weeds could be more serious especially in subsurface systems that are commonly lined with plastic liners. In this case there is a danger of root penetration through the liner and subsequent water leakage.

However, in well established stands the weedy species are usually limited to the wetland margins because plants typically used in constructed wetlands are quite robust, and it is difficult to outcompete these plants once they are established. Regardless of the location of the wetland, there is always the potential that some type of "undesirable" plant will introduce itself, especially those spreading easily by seeds. Consequently, any operations and maintenance plan for a wetland treatment system should address removal and management of undesirable plant species. Typical plant species invading constructed wetlands include purple loosestrife (*Lythrum salicaria*), cottonwood (*Populus deltoides*), willow (*Salix* spp.), stinging nettle (*Urtica diocia*), and in North America, common reed (*Phragmites australis*).

### Examples of Weeds in Treatment Wetlands

The submersed macrophyte hydrilla (*Hydrilla verticillata*) is native to the warm areas of Asia. It was first discovered in the United States in 1960 (Langeland, 1996). This plant is well adapted to life in submersed freshwater environments, and has spread rapidly through portions of the United States, where it has become a serious weed. Where the plant occurs, it displaces native aquatic plant communities, and adversely impacts freshwater habitats by forming a dense surficial cover. *Hydrilla* has invaded hundreds of hectares of the Florida stormwater treatment areas (STAs). Its performance for water quality improvement is somewhat poorer than other SAV species (DB Environmental, unpublished results, 2006).

*Phragmites australis* is a widely distributed clonal grass species, ranging all over Europe, Asia, Africa, America, and Australia. Extensive reed beds are protected in Europe because of their important ecological functions. In contrast, the rapid expansion of *P. australis* in North America, particularly along the Atlantic coast, is considered a threat to biodiversity. Although *P. australis* was a component of marshes in New England several thousand years ago, genetic evidence (Saltonstall, 2002; Blossey *et al.*, 2002) has now confirmed that a more aggressive genotype has been introduced to North America, probably in the late 1800s. Dense *Phragmites* stands in North America have decreased native biodiversity and quality of wetland habitat, particularly for migrating waders and waterfowl species. The closest related species is *Arundo donax*, also an invasive introduced species.

Purple loosestrife (*Lythrum salicaria*) is a herbaceous perennial of Eurasian origin that became established in northeastern North America in the early 1800s (Thompson *et al.*, 1987). By the late 1800s it had spread throughout the northeastern United States and southeastern Canada, reaching as far north and west as Manitoba. Since then, it has steadily expanded its local distribution and now poses a serious threat to native emergent vegetation in shallowwater marshes throughout northern North America. Thompson *et al.* (1987) observe:

> It is no small irony that after 50 years of struggle to find some means of breaking up monotypic stands of cattails (*Typha* spp.) to increase wildlife diversity and abundance, wetland managers must now cope with a foreign species that replaces cattail, but unfortunately creates another monospecific community of greatly diminished wildlife value.

The impact of purple loosestrife on native vegetation in North America is disastrous, with almost the entire biomass of some wetland communities displaced. Monospecific blocks of this weed have survived for at least 20 years. Impacts on wildlife indicate serious reductions in waterfowl and aquatic furbearer productivity. Several declining species of vertebrates are threatened with further degradation of their breeding habitats with the continued expansion of purple loosestrife.

Woody plants, especially willows, are opportunistic invaders in some constructed treatment wetlands (Figure 3.33).

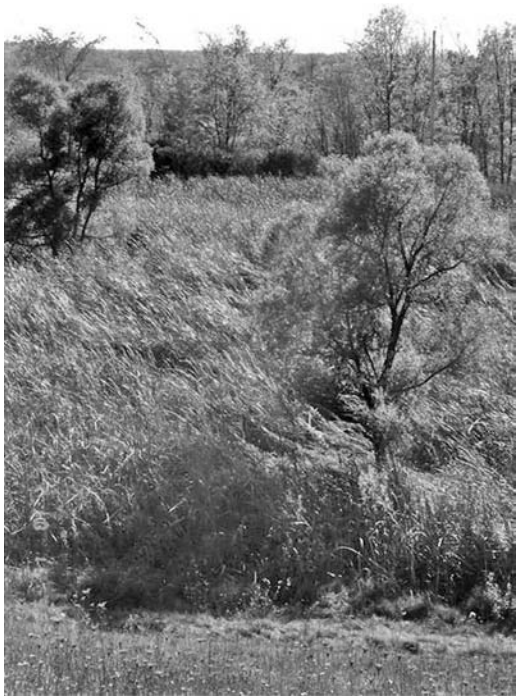


**FIGURE 3.33 (A color version of this figure follows page 550)** Trees growing in the Vermontville, Michigan, constructed treatment wetland after 15 years of operation.

Although there are no documented failures of wetlands due to tree growth, there is a perception that potential problems could occur. For instance, tree roots can compromise the integrity of containment berms if the berms are of small cross-section. There is also the possibility that tree roots could puncture wetland liners, either plastic membranes or clay layers, and allow wastewaters to seep to groundwater. Another possibility is that large trees would be susceptible to falling over during high wind events if the wetland liner prevents normal propogation of the root systems of the tree.

In contrast to the idea of invasive takeover and destruction of biodiversity, there is the sometimes-held concept that wetlands should contain only the plants selected in design. The view of the Constructed Wetland Association (CWA) of the United Kingdom is that reed beds should be *Phragmites* monocultures as designed. A study of 255 reed bed sites built by Severn Trent Water found that "weed infestation" was a problem at 130 of the sites (51%), defined as more than 25% cover (Cooper *et al*., 2006a). The perception of CWA is that *Phragmites* is necessary to alleviate clogging of the bed, by several mechanisms including "windrock." Other plants are believed to be capable of functioning only for some weeks or months.

## SUMMARY

Wetland environments support a wide variety of bacteria, fungi, algae, and macrophytes (submerged, floating, and emergent). Treatment wetlands have been implemented that use periphyton, algae, submerged macrophytes, floating vegetation, and woody plants, although emergent macrophytes remain the most common choice. In order to survive in a flooded environment, emergent macrophytes transport oxygen from their leaves through their stalks to the root tissues. The majority of this oxygen is used for plant respiration, although some is used to detoxify the rooting environment. For wetlands treating primary effluents, such as many HSSF wetlands, the amount of oxygen that passes into the water column from the plant roots is negligible compared to the wastewater loading, and majority of the root biomass is in the top 20 cm of the wetland bed.

The growth, death, and decay of plant biomass is an important biogeochemical cycle in treatment wetlands and imposes a seasonal cycle on many internal processes. During the growing season, nutrients such as nitrogen and phosphorus are taken up by the plants, and temporarily stored in the plant canopy. This uptake is significant for juvenile ecosystems where the plant canopy being established, and for periods of peak plant growth. At the end of the growing season, nutrients are returned to the system after the emergent portion of the plants die back. The decay of plant biomass imposes nonzero background concentrations for many constituents in treatment wetlands and is important in some treatment processes, such as denitrification. Some portion of the phytomass is resistant to degradation, leading to a net accretion of refractory organic matter in treatment wetlands.

A wide variety of plant species have been used in treatment wetlands, and initial plant selection is a function of hydropattern, climate, and cultural choices. Regardless of the initial planting, the plant community will self-organize over time as additional plant species invade the system. If the project goal is to maintain a specific plant community, human intervention will be required to remove plants that are viewed as being undesirable.

# 4 Energy Flows

Water temperatures in treatment wetlands are driven by energy flows (gains and losses) that act on the system. During warm conditions, the largest energy gain is solar radiation, and the largest energy loss is evapotranspiration. Energy flows are cyclical and act on both daily (diurnal) and seasonal time scales.

As water flows through the wetland, energy gains and losses drive the water temperature towards a balance point temperature, at which energy gains equal energy losses. This results in a longitudinal change in water temperatures as the system trends towards the balance point. The balance point temperature may be warmer or cooler than the influent water temperature, depending on the relative magnitude of the energy flows.

Because temperature exerts a strong influence on some chemical and biological processes, it is important to wetland design. In cold climates, freezing of the wetland may be an operational concern. Successful design requires that forecasts be made for expected or worst-case operating conditions, which implies prediction rules and equations. This chapter reviews the data on treatment wetland water temperatures, and explores the tools available to wetland designers to predict water temperatures that result from energy flows within treatment wetlands.

The water temperature in treatment wetlands is of interest for several reasons:

1. Temperature modifies the rates of several key biological processes.
2. Temperature is sometimes a regulated water quality parameter.
3. Water temperature is a prime determinant of evaporative water loss.
4. Cold-climate wetland systems have to remain functional in subfreezing conditions.

In the first instance, there is extensive literature supporting the strong effect of temperature on microbial nitrogen processing, with doubling of rates over a temperature range of about 10°C. In the second case, cold-water fishes, such as salmonids, are sensitive to water temperature, and cannot survive or breed in warm environments. In the third case, net water loss (and associated increases in total dissolved solids) is a detriment in arid climates, where water rights and water return credits are of increasing importance. Additionally, water temperature is strongly connected to evapotranspiration, which in turn is a major factor in the water budget for the wetland. Finally, freezing of the wetland can create operational problems in cold-climate applications unless the system is designed to avoid freeze-up failure.

## 4.1 WETLAND ENERGY FLOWS

The energy flows that determine water temperature and the associated evaporative losses are shown in Figure 4.1. These processes are driven and dominated by solar radiation. Incoming solar radiation is partially reflected, with the remainder intercepted by the vegetative canopy and water column. Solar radiation intercepted by the vegetative canopy drives transpiration in plants. The remaining solar radiation is absorbed by the wetland water, and drives evaporation. The combined water loss is termed *evapotranspiration*, and is commonly abbreviated as *ET*.

Convection and diffusion carry water away from the surface, and transfer heat from the air to the wetland. The driving force for convective and diffusive heat transfer is the temperature difference between the wetland and the air above. For water vapor transport, the driving force is the water partial pressure difference between the wetland and the air above. Additionally, heat is radiated from the wetland. Heat may also be transferred from soils to the wetland, but that contribution is usually very small. The net effect of these processes will be a difference between the sensible heats of incoming and outgoing water flows.

Wetland energy flows are the proper framework to interpret and predict not only evaporative processes, but also wetland water temperatures. The energy balance equations involve time-step calculations, and are in general only amenable to computer spreadsheets. However, those calculations are now available from Internet sources, and the wetland designer can readily use this approach. The required input information consists of meteorological information. There are many versions of the energy balance equations that have been put forth, and the interested reader may pursue details in the literature, including the comparative study of *ET* predictive methods for a Florida treatment wetland (Abtew and Obeysekera, 1995). A brief summary of the model will serve to explain these data needs.

### Energy Balance Terms

Here the methods for calculating each of the quantities in the wetland energy balance are illustrated. The magnitudes of the various energy flows are given in Table 4.1, for FWS wetlands near Phoenix, Arizona (Kadlec, 2006c), in the balance condition. These wetlands were large enough to consider as driven by regional climatic variables. However, freezing conditions are virtually nonexistent at that location. Cold climate wetland considerations are considered in subsequent sections, as are modifications for HSSF systems.

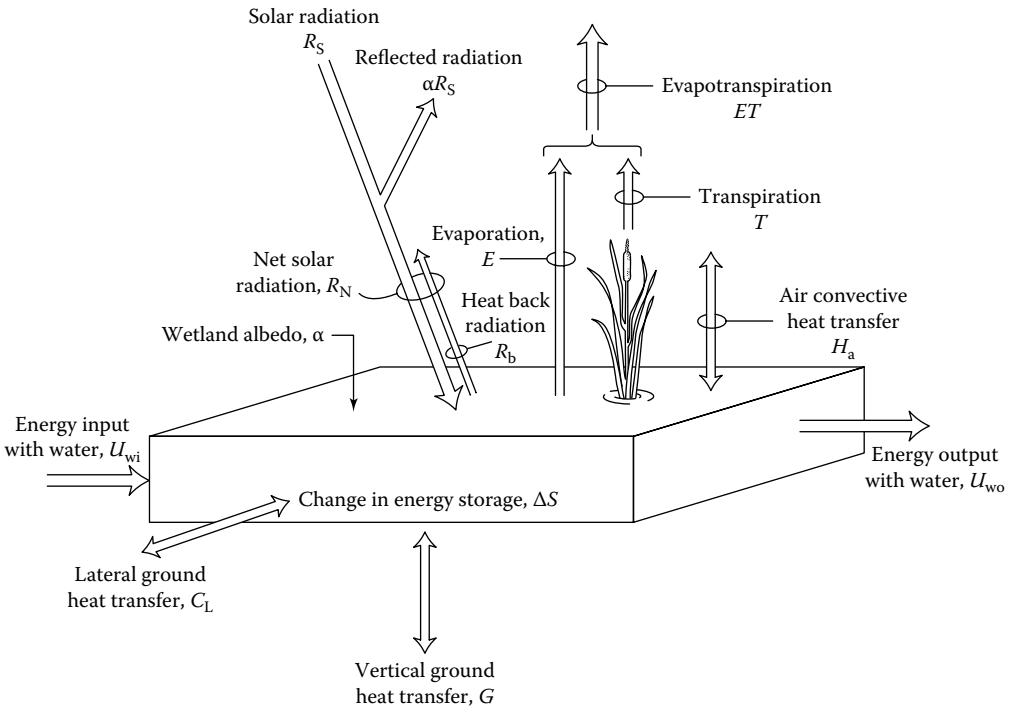



**FIGURE 4.1** Components of the wetland energy balance. (From Kadlec and Knight (1996) *Treatment Wetlands*. First edition, CRC Press, Boca Raton, Florida.)

The system for the energy balance is here taken to be the wetland water body and the associated phytomass (Figure 4.1).

$$\textit{Energy Inputs} - \textit{Energy Outputs} = \textit{Change in Energy Storage}$$

$$[R_N + H_a + U_{wi}] - [\lambda_m \rho ET + U_{wo} + G + C_L] = \Delta S \qquad (4.1)$$

where

$C_L$ = lateral heat loss to ground, MJ/m$^2$·d
$G$ = vertical conductive loss to ground, MJ/m$^2$·d
$ET$ = water lost to evapotranspiration, m/d
$H_a$ = convective transfer from air, MJ/m$^2$·d
$R_N$ = net radiation absorbed by wetland, MJ/m$^2$·d
ΔS = energy storage change in the wetland, MJ/m$^2$·d

**TABLE 4.1**
**Heat Budget Elements (MJ/m$^2$·d) for a Portion of a FWS Wetland in Phoenix, Arizona, in the Balance Condition**

| Month | Radiation Net In | Heat Gain from Air | Sensible Heat from Water | Surface Flux from Ground | Total In | Heat Loss from *ET* | Thermal Back Radiation | Total Out |
|---|---|---|---|---|---|---|---|---|
| Jan | 10.5 | 0.4 | 0.0 | 0.2 | **11.2** | 4.7 | 6.4 | **11.2** |
| Feb | 13.2 | –0.1 | 0.0 | 0.1 | **13.1** | 6.5 | 6.6 | **13.1** |
| Mar | 16.7 | –0.2 | 0.0 | 0.0 | **16.5** | 9.7 | 6.8 | **16.5** |
| Apr | 20.4 | 0.7 | 0.0 | –0.2 | **20.9** | 13.9 | 7.0 | **20.9** |
| May | 22.9 | 2.1 | 0.0 | –0.3 | **24.8** | 17.8 | 6.9 | **24.8** |
| Jun | 23.9 | 3.3 | 0.0 | –0.3 | **26.9** | 20.1 | 6.8 | **26.9** |
| Jul | 22.9 | 3.6 | 0.0 | –0.2 | **26.3** | 19.8 | 6.5 | **26.3** |
| Aug | 20.2 | 3.1 | 0.0 | –0.1 | **23.1** | 16.9 | 6.2 | **23.1** |
| Sep | 16.5 | 2.2 | 0.0 | 0.0 | **18.7** | 12.5 | 6.2 | **18.7** |
| Oct | 13.0 | 1.5 | 0.0 | 0.2 | **14.6** | 8.4 | 6.2 | **14.6** |
| Nov | 10.4 | 1.1 | 0.0 | 0.3 | **11.8** | 5.5 | 6.3 | **11.8** |
| Dec | 9.5 | 0.9 | 0.0 | 0.3 | **10.7** | 4.3 | 6.3 | **10.7** |

*Note:* The hydraulic loading rate is 15 cm/d.

*Source:* From Kadlec (2006c) *Ecological Engineering* 26: 328–340. Reprinted with permission.

$U_{wi}$ = energy entering with water, MJ/m²·d
$U_{wo}$ = energy leaving with water, MJ/m²·d
$\lambda_m$ = latent heat of vaporization of water, MJ/kg (2.453 MJ/kg at 20°C)
$\rho$ = density of water, kg/m³

It is informative to examine these terms, with a view to understanding the magnitude of the various heat fluxes.

## Solar Radiation

The net incoming radiation reaching the surface of the wetland may be calculated through a series of steps which estimate the absorptive and reflective losses from incoming extraterrestrial radiation, $R_A$, shown in Figure 4.1. The amount of radiation which makes it through the outer atmosphere is solar radiation:

$$R_S = \left(0.25 + 0.5\frac{S}{100}\right) R_A \tag{4.2}$$

where
$R_A$ = extraterrestrial radiation, MJ/m²·d
$R_S$ = solar radiation, MJ/m²·d
$S$ = percent daily sunshine

Solar radiation ($R_S$) is the quantity reported by the several climatological data services as discussed below. The data scatter about an annual sinusoidal trend (Figure 4.2). The upper limit of the data envelope represents cloud-free conditions ($S$ = 100), and individual days may have lesser amounts of incoming radiation.

A fraction $\alpha$, the wetland albedo, of this radiation is reflected by the wetland. A value of $\alpha$ = 0.23 is commonly used for green crops (ASCE, 1990). Priban *et al.* (1992) present seasonally variable values for wetlands, with summer values of 0.18–0.22, and an autumn value of 0.10.

## Back Radiation (Radiative Heat Loss)

Net outgoing long wave (heat) radiation is computed based on atmospheric characteristics of cloud cover, absolute temperature, and moisture content:

$$R_b = \left[0.1 + 0.9\left(\frac{S}{100}\right)\right]\left[0.34 - 0.139\sqrt{P_w^{sat}(T_{dp})}\right]\sigma(T+273)^4 \tag{4.3}$$

where
$R_b$ = net outgoing long wave radiation, MJ/m²·d
$P_w^{sat}(T_{dp})$ = water vapor pressure at the dew point, kPa
$T$ = air temperature, °C
$\sigma$ = Boltzmann's constant = $4.903 \times 10^{-9}$ MJ/m²·d

In combination, the net incoming radiation is:

$$R_N = 0.77R_S - R_b \tag{4.4}$$

For example, net radiation at Phoenix ranges from (9.5 × 0.77 − 6.3) = 1.0 MJ/m²·d in December, to (23.9 × 0.77 − 6.8) = 11.6 MJ/m²·d in June (see Table 4.1).

## Convective Losses and Gains to Air

Although lumped together in Equation 4.1, there are two major and distinct components of heat exchange with air. Wind blows through the wetland plant canopy, and either warms or cools the leaves. In the process, it removes the water transpired through the leaves. Secondarily, this air also may heat or cool the water or gravel bed underlying the canopy.

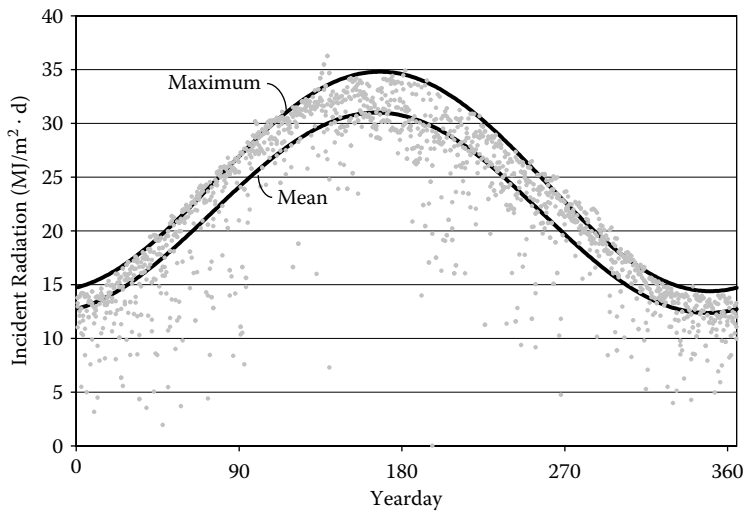



**FIGURE 4.2** Solar radiation as a function of season for Phoenix, Arizona. Mean and maximum trendlines are shown, along with data from 1995–1999.

The relative proportions depend upon the extent of vegetative cover, and the relative areas of leaves and water (bed). The effect in the canopy is to control transpiration, whereas the effect in the wetland below is to control evaporation and water temperature.

Accompanying the heat transfer in the canopy, there will be a corresponding mass transfer of water vapor from the leaves to the air passing through. In FWS, there will be a corresponding mass transfer of water vapor from the water surface to the air. However, in HSSF systems, this transfer from water is blocked by dry surface media and also mulch, if used.

Calculations utilize the known relations between the transfer rates and wind speed. For instance, according to ASCE (1990), the vapor flow is calculated as a mass transfer coefficient times the water vapor pressure difference between the water or leaf surface and the ambient air above the wetland:

$$ET = K_e[P_w^{sat}(T_w) - P_{wa}] = K_e \Delta P_w \quad (4.5)$$

where

$K_e$ = water vapor mass transfer coefficient, m/d·kPa
$P_{wa}$ = ambient water vapor pressure, kPa
$P_w^{sat}(T_w)$ = saturation water vapor pressure at $T_w$, kPa
$T_w$ = water temperature, °C

Typically, the amount of water in the ambient air is a known quantity, calculated as the relative humidity times the saturation pressure of water at the ambient air temperature:

$$P_{wa} = RH \cdot P_w^{sat}(T_{air}) \quad (4.6)$$

where

$RH$ = relative humidity, fraction
$T_{air}$ = air temperature, °C

The water transport coefficient has been found to be a linear function of the wind velocity, the following correlation being one of several in common use (ASCE, 1990):

$$K_e = \frac{(4.82 + 6.38u)}{\lambda} = (10^{-3})(1.965 + 2.60u) \quad (4.7)$$

where

$u$ = wind speed at two meters elevation, m/s
$\lambda = \rho\lambda_m$ = volumetric latent heat of vaporization of water (2,453 MJ/m$^3$)

The convective heat transfer from the water to the air is likewise represented as a heat transfer coefficient times the temperature difference:

$$H_a = U_{air}\,[T_w - T_{air}] = U_{air}\,\Delta T \quad (4.8)$$

where

$U_{air}$ = heat transfer coefficient, MJ/m$^2$·d·°C

The relation between heat and mass transfer in the air–water system has resulted in an accurate, calibrated relation between the heat and mass transfer coefficients (ASCE, 1990):

$$U_{air} = \gamma \lambda K_e = (0.0666)(2453)K_e = 163.3K_e \quad (4.9)$$

where

$\gamma = \dfrac{c_p P}{[0.622]\lambda}$ = the psychrometric constant, kPa/°C

$\gamma$ = 0.0666 at 20° C and 1 kPa and (0.622 = 18/29 = molecular weight ratio of water to air)
$c_p$ = heat capacity of air, MJ/kg°C
$P$ = ambient air pressure, kPa

thus

$$U_{air} = (0.0666)(4.82 + 6.38u) = 0.321 + 0.425u \quad (4.10)$$

For the Phoenix example, exchanges with air range from slight losses of −0.2 MJ/m$^2$·d in March, to gains of 3.6 MJ/m$^2$·d in June (Table 4.1). The corresponding heat transfer coefficients were $U_{air}$ = 0.60 ± 0.07 MJ/m$^2$·°C·d. For the NERCC, Minnesota HSSF wetlands, $U_{air}$ = 0.31 ± 0.03 MJ/m$^2$·d·°C (Kadlec, 2001b). These values are consistent with the widely accepted value of the heat transfer coefficient in stagnant air above evaporating vegetated surfaces, which is $U_{air}$ = 0.37 MJ/m$^2$·d·°C (ASCE, 1990). Crites et al. (2006) provide best judgment estimates of $U_{air}$ = 0.13 MJ/m$^2$·d·°C for dense marshes, 0.86 for open water in still air, and 2.15 for windy conditions in open water.

The energy exchange between water and air in winter in cold climates requires more detailed calculations involving the insulating properties of mulches, ice, and snow. That situation will be discussed separately below.

## Conduction Losses and Gains from Soils

In general, lateral energy transfers, horizontally from the wetland edges, are small enough to be negligible. Lateral losses at the Grand Lake, Minnesota, wetland were found to be 0.001–0.003 MJ/m$^2$·d.

The vertical energy gains and losses from soils below the water are also usually negligible compared to radiation and *ET* during summer, but are of considerable importance in winter, when they are the only gains. Approximate calculations may be based on the vertical temperature gradient below ground:

$$G = k_g\left(-\frac{dT}{dz}\right) \quad (4.11)$$

where

$G$ = energy gain, MJ/m$^2$·d
$k_g$ = thermal conductivity of ground, MJ/m·d·°C
$T$ = soil temperature, °C
$z$ = vertical distance upward, m

The thermal conductivities of soils vary with type, with a typical range of 30–190 kJ/m°C·d (Table 4.2). The maximum vertical temperature gradients below treatment wetlands have

**TABLE 4.2**
**Thermal Conductivities of Wetland Solid Materials**

| Material | Thermal Conductivity (MJ/m·d·°C) |
|---|---|
| Air | 0.0021 |
| Milled peat | 0.0043 |
| Granular peat | 0.0053 |
| Dry litter (straw) | 0.009 |
| New snow | 0.007 |
| Dry LECA | 0.010 |
| Wet LECA | 0.015 |
| Old snow | 0.022 |
| Dry gravel | 0.026 |
| Dry sand | 0.030 |
| Soil | 0.045 |
| Water | 0.051 |
| Saturated peat | 0.052 |
| Clay | 0.112 |
| Dry sand | 0.152 |
| Ice | 0.190 |

*Note:* These are generic materials with considerable variability in property values, and the numbers are therefore approximate.

been measured to be in the range of 5–15°C/m, decreasing upward in the winter, and decreasing downward in summer. Accordingly, the heat additions (winter) or losses (summer) reach extremes of 0.15–2.9 MJ/m²·d.

The vertical conduction process has been modeled as transient heat conduction, and fits data quite well for FWS and HSSF systems (Priban *et al.*, 1992; Kadlec, 2001b). The temperature profiles $T(z, t)$ in the (unfrozen) soils below a wetland are governed by the unsteady-state heat conduction equation, together with the boundary condition of a fixed temperature mean annual temperature, a constant at deep locations:

$$\frac{\partial^2 T}{\partial z^2} = \frac{1}{\alpha}\frac{\partial T}{\partial t} \tag{4.12}$$

$$T(\infty, t) = T_s \tag{4.13}$$

For a sinusoidal surface temperature, the solution to this periodic, dynamic heat balance is (Priban *et al.*, 1992):

$$T(z,t) = T_s + A\exp\left(-\frac{z}{H}\right)\cos\left[\omega(t - t_{max}) - \frac{z}{H}\right] \tag{4.14}$$

where

$$H = \sqrt{\frac{2\alpha}{\omega}} \tag{4.15}$$

$$\alpha = \frac{k}{\rho_s c_s} \tag{4.16}$$

and

$A$ = amplitude of surface temperature cycle, °C
$c_s$ = soil heat capacity, MJ/kg·°C
$k$ = soil thermal conductivity, MJ/m·d·°C
$t$ = time, Julian day
$t_{max}$ = time of maximum surface temperature, Julian day
$T$ = temperature, °C
$T_s$ = mean annual temperature of the soil surface, °C
$z$ = vertical depth, m
$\alpha$ = thermal diffusivity of soil, m²/d
$\rho_s$ = soil density, kg/m³
$\omega$ = annual cycle frequency = 2 /365 = 0.0172 d⁻¹

The penetration depth ($H$) is the depth at which the mean annual temperature swing is 63.2% of that at the soil surface ($A$). The heat flux into the water from the soil is then:

$$G = \left[\frac{kA}{H}\right]\cdot\left[\cos\left(\omega(t - t_{max})\right) - \sin\left(\omega(t - t_{max})\right)\right] \tag{4.17}$$

It may be shown that the heat flux ($G$) achieves a maximum 46 days (one eighth of an annual cycle) before the day of minimum water temperature, which is also 136 days after the day of maximum water temperature. It may also be shown that the total heat gain from the soil over the 182-day heating half cycle ($G_{half}$) is:

$$G_{half} = \left(2\sqrt{2}\right)\frac{kA}{\omega H} \tag{4.18}$$

The maximum daily heat gain may be shown to be a factor $\pi/2 = 1.57$ times greater than the average rate over the heating half of the year.

This model provides an accurate description of the temperature gradients below the Grand Lake and NERCC, Minnesota, treatment wetlands (Kadlec, 2001b), as well as the Jackson Meadow, Minnesota, and Houghton Lake, Michigan, treatment wetlands (Table 4.3). In addition to the sinusoidal surface water temperature parameters, only one further constant is needed, the penetration depth ($H$).

### Heating or Cooling of the Water

As water passes through the treatment wetland, it may either cool or warm, depending on meteorological conditions. The energy associated with the water (sensible heat) is a relative quantity, requiring a reference temperature:

$$U = \rho\, c_p\, Q(T_w - T_{ref}) \tag{4.19}$$

where

$c_p$ = heat capacity of water, MJ/kg·°C
$Q$ = water flow, m³/d
$T_w$ = water temperature, °C

**TABLE 4.3**
**Regression Parameters for the Under-Wetland Soil Temperature Heat Conduction Model**

| Parameter | NERCC 1, Minnesota HSSF | NERCC 2, Minnesota HSSF | Grand Lake, Minnesota HSSF | Jackson Meadow, Minnesota HSSF | Houghton Lake, Michigan FWS |
|---|---|---|---|---|---|
| Data years | 4 | 4 | 4 | 2 | 4 |
| Number of depths | 4 | 4 | 4 | 4 | 5 |
| Soil | Mineral | Mineral | Mineral | Mineral | Wet peat |
| Surface temperature amplitude, °C | 8.23 | 8.23 | 8.02 | 10.11 | 8.15 |
| Surface temperature maximum, Julian day | 213 | 213 | 217 | 219 | 195 |
| Penetration depth, m | 2.05 | 2.24 | 2.17 | 0.61 | 0.95 |
| Thermal diffusivity, $m^2/d$ | 0.0361 | 0.0432 | 0.0407 | 0.0032 | 0.0078 |
| Correlation coefficient, $R^2$ | 0.87 | 0.89 | 0.88 | 0.92 | 0.89 |
| Upward heat flux maximum, Julian day | 350 | 349 | 353 | 356 | 332 |
| Maximum heat flux, $MJ/m^2 \cdot d$ | 0.274 | 0.250 | 0.250 | 1.189 | 0.772 |
| Half-year heat gain, $MJ/m^2$ | 31.8 | 29.1 | 28.9 | 138 | 89.8 |

$T_{ref}$ = reference temperature, °C
$U$ = energy flow with water, MJ/d
$\rho$ = density of water, $kg/m^3$

The sensible heat increase or decrease from inlet to outlet, per unit area of wetland, is:

$$\Delta U = \rho c_p q(T_{wo} - T_{wi}) \tag{4.20}$$

where
$q$ = hydraulic loading rate, m/d
$T_{wi}$ = inlet water temperature, °C
$T_{wo}$ = outlet water temperature, °C

The energy associated with a 5°C increase in water temperature, at a hydraulic loading rate of 5 cm/d, is 1.04 $MJ/m^2 \cdot d$.

## Changes in Storage: Thermal Inertia

Energy is absorbed as the entire wetland heats up, or released as it cools down. Maximum seasonal rates of temperature change are of the order of 0.5°C/d. The energy absorbed in increasing the wetland temperature is:

$$\Delta S = \rho c_p h_w \left( \frac{dT}{dt} \right) \tag{4.21}$$

where
$h_w$ = water depth, m
$\Delta S$ = stored energy increase in one day, $MJ/m^2 \cdot d$
$dT/dt$ = water temperature increase rate, °C/d

The heat capacity of the wetland, at a depth of 0.45 m, is (4.182)(0.45) = 1.88 $MJ/m^2 \cdot °C$. The energy associated with a 0.5°C/d increase in mean FWS wetland water temperature is 0.94 $MJ/m^2 \cdot d$.

A HSSF wetland has greater thermal inertia, or storage potential, because of the presence of the gravel matrix. The heat capacity of the wetland is comprised of water and gravel contributions:

$$(\rho ch)_{wetland} = [\varepsilon(\rho c)_{water} + (1 - \varepsilon)(\rho c)_{gravel}]h \tag{4.22}$$

where
$h$ = depth of the bed, m
$\varepsilon$ = porosity of bed, unitless

For a 45-cm deep bed at porosity 0.4, with gravel heat capacity 0.2 times that of water, which is typical of nearly all stone materials:

$$(\rho cV)_{wetland} = [0.4(1{,}000)(4{,}182) + (1 - 0.4)(2{,}500)(840)](0.45)$$
$$= 1.32\ MJ/m^2 \cdot °C$$

Here the density of the media has been selected as 2.5 times that of water. The maximum energy storage rate is then is 0.66 $MJ/m^2 \cdot d$.

Shoemaker *et al.* (2005) investigated the role of storage on fluctuations in energy balances for FWS wetlands in Florida. They found that the magnitude of changes in stored heat energy generally decreased as the time scale of the energy balance increased. Daily fluxes of stored heat energy accounted for 20% or more of the magnitude of mean daily net radiation for about 40% of their data, whereas weekly fluxes of stored heat were 20% of mean weekly net radiation for about 20% of the same data. Thus, storage plays a role in dampening short-term energy flow variations.

## Heat of Vaporization

Evaporated and transpired water require the input of considerable energy to accomplish the phase change from liquid, in the water column or in the leaves of the canopy, to the vapor form in the air above. As indicated in Equation 4.1, this is computed as the specific heat of vaporization times the

evapotranspiration rate, $\lambda_m \rho ET$, where $\lambda_m$ = 2453 MJ/kg. Wetland *ET* varies seasonally, from minimum values in winter to maxima in summer. Peak midsummer *ET* rates range upward from about 5 mm/d, depending upon wetland size. The peak midsummer energy required therefore ranges upward from 12.3 MJ/m$^2$·d. In Phoenix, heat loss to *ET* ranges from 4.3 to 20.1 MJ/m$^2$·d (see Table 4.1). In temperate climates, in winter, *ET* drops to close to zero. The existence of frozen conditions and snow cover requires additional considerations, given below.

## 4.2 EVAPOTRANSPIRATION

Water losses to the atmosphere from a wetland occur from the water and soil (evaporation, *E*), and from the emergent portions of the plants (transpiration, *T*). The combination of the two processes is termed evapotranspiration (*ET*). This combined water vapor loss is primarily driven by solar radiation for large wetlands, but may be significantly augmented by heat transfer from air for small wetlands. It is governed by the same wetland energy balance equations that describe wetland water temperatures.

Evapotranspiration is the primary energy loss mechanism for the wetland, and serves to dissipate the majority of the energy. In this context, evapotranspiration can be thought of as the cooling system for the treatment wetland. Without the attendant energy loss through the latent heat of vaporization of water, the "wetland" temperature would increase to a hot, desert-like condition since incoming solar radiation could not be effectively dissipated. Although evapotranspiration is best thought of in terms of the wetland energy balance, sometimes only the water volume lost through *ET* is of concern, and the attendant energy flows associated with *ET* can be ignored. As a result, there are a variety of methods to estimate *ET*. Some estimation methods rely on energy balance calculations, while others rely on surrogate measurements.

### Methods of Estimation for *E*, *T*, and *ET*

There are several related measurements of lake and wetland water losses. These measurements are not interchangeable, and indiscriminate use can lead to confusion. Information that can be used to estimate *ET* includes the following:

1. Lake evaporation, which is the loss from large, unvegetated water bodies (*E*).
2. Transpiration, which is the loss of water through above-water (or aboveground) plant parts (*T*).
3. Wetland evapotranspiration, which is the loss from vegetated water bodies (*ET*). Vegetation may be rooted or floating, emergent or submergent.
4. Class A pan evaporation, which is the water loss from a shallow pan of specific design, situated on a specified platform ($E_A$)
5. Evaporation from closed-bottom lysimeters (pans) of varying design ($E_P$), containing only water. These may be place in stands of emergent vegetation ($E_{PV}$) or in areas of open water, with or without submergent or floating plants ($E_{PO}$).
6. Evapotranspiration from closed-bottom lysimeters (pans) of varying design, which contain soil, plants and water ($ET_P$). These are placed in stands of comparable vegetation.
7. Regional, large scale, water loss computed from meteorological information, for a reference crop and the assumption of standing water or saturated soil surface ($ET_o$). Computations may follow one of several energy balance methods, such as Penman–Montieth (Monteith, 1981) or Priestley–Taylor (Priestley and Taylor, 1972).

### Energy Balance Methods

For large wetlands, the principal driving force for *ET* is solar radiation. A good share of that radiation is converted to the latent heat of vaporization. About half the net incoming solar radiation is converted to water loss on an annual basis. Reported values include: 0.49, (Bray, 1962); 0.47, (Christiansen and Low, 1970); 0.51, (Kadlec *et al.*, 1987); 0.64, (Roulet and Woo, 1986); 0.54, (Abtew, 1996; 2003). If radiation data from the central Florida area are used to test the concept for the Clermont wetland (Zoltek *et al.*, 1979), the value is 0.49.

Equation 4.1 and its variants are widely used in the literature to predict *ET*. Its use is dependent on equations relating the quantities in Equation 4.1 to meteorological and environmental variables. Incoming radiation depends upon latitude, season, and cloud cover. Incident radiation data are typically readily available from weather stations or summary service organizations, such as the National Climatological Data Center (NCDC) in the United States (http://www.ncdc.noaa.gov), which monitors radiation at 237 stations across the country.

Water losses to the atmosphere from a wetland occur from the water and from emergent vegetation. Convective eddies in the air, associated with wind, swirl water vapor and sensible heat from the water and vegetation upward to the bulk of the overlying air mass. The driving force for water transfer into the air is the humidity difference between the water surface (assumed saturated) and the bulk air. This humidity difference is strongly dependent upon water temperature, via the vapor pressure relationship.

One simple *ET* calculation procedure for large regional wetlands was described in the first edition of this book. It is not repeated here because there are now short cuts available to the treatment wetland designer.

### The Reference Crop $ET_o$ Spreadsheet Method

For large wetlands, a common assumption is that *ET* may be represented by the reference crop $ET_o$ computation. The Environmental and Water Resources Institute (EWRI) of the American Society of Civil Engineers (ASCE) established a benchmark reference evapotranspiration equation that standardizes the calculation of reference evapotranspiration

(Allen *et al*., 2000); (http://www.kimberly.uidaho.edu/water/asceewri/). The intent was to produce consistent calculations for reference evapotranspiration for use in agriculture. A spreadsheet program, PMday.xls, is available (Snyder and Eching, 2000; Snyder, 2001). Inputs include the daily solar radiation (MJ/m$^2$·d), air temperature (°C), wind speed (m/s), and humidity (e.g., dew point temperature (°C) or relative humidity (%)). The program calculates $ET_o$ using the Penman–Monteith equation (Monteith, 1965) as presented in the United Nations FAO Irrigation and Drainage Paper by Allen *et al*. (1998).

This procedure has been calibrated and verified for a green alfalfa crop, with a fetch of at least 100 m. Other cover types may vary, due to changes in albedo and convective transport and other factors. It is critical to recognize that small wetlands will have significantly greater convective heat transfer and, consequently, *ET* is amplified in small wetlands.

### Reference Crop $ET_o$ from Reporting Services

In the United States, arid states provide extensive documentation of $ET_o$ in support of agricultural irrigation, such as the California Irrigation Management Information System (CIMIS, http://wwwcimis.water.ca.gov/cimis/welcome.jsp), the Arizona Meterological Service (AZMET), and the Washington State University Public Agricultural Weather System (PAWS) (http://paws.prosser.wsu.edu/). A comparable system in the United Kingdom is the Meteorological Office Rainfall and Evaporation Calculating System (MORECS) (Fermor *et al*., 1999). These services provide the results of energy balance model calculations, usually on a daily time step, for current and recent weather conditions. Figure 4.3 shows an example of the annual pattern of $ET_o$ computed for Phoenix, Arizona. Such annual patterns vary with latitude, as indicated in Figure 4.4.

Direct calibrations and checks have been conducted in wetland environments (Abtew, 1996; German, 2000). As a first approximation, $ET = ET_o$ for large FWS wetlands; however, crop coefficients are required for small systems, as shown in Equation 4.23:

$$ET = K_c \cdot ET_o \tag{4.23}$$

where

$K_c$ = wetland crop coefficient, dimensionless

Lafleur (1990) recommended using the energy balance $ET_o$ estimate as the independent variable in linear regression for specific vegetation types. In agriculture, this approach leads to crop coefficients that influence *ET* at a specific site. This approach has the advantage of retaining the energy balance used in other ecosystems, but modifying it slightly for site-specific circumstances.

### Pan Factor Methods ($E_A$)

The Class A evaporation pan is a convenient reference, because there are many long-term data stations in the United States. The pan is placed on a platform above ground, and therefore evaporates more water than a lake or large wetland. (ASCE, 1990). Each state operates pans at a few stations, and data are reported in Climatological Data, a publication of the National Oceanic and Atmospheric Administration (NOAA), National Climatic Data Center, and available at (http://www.ncdc.noaa.gov).

Wetland evapotranspiration, *ET*, over at least the growing season, can be approximated as about 0.70–0.85 times Class A pan evaporation, $E_A$, from an adjacent open site. The Class A pan integrates the effects of many of the meteorological variables, with the notable exception of advective effects. A multiplier of about 0.8 has been reported in several studies, including: northern Utah, (Christiansen and

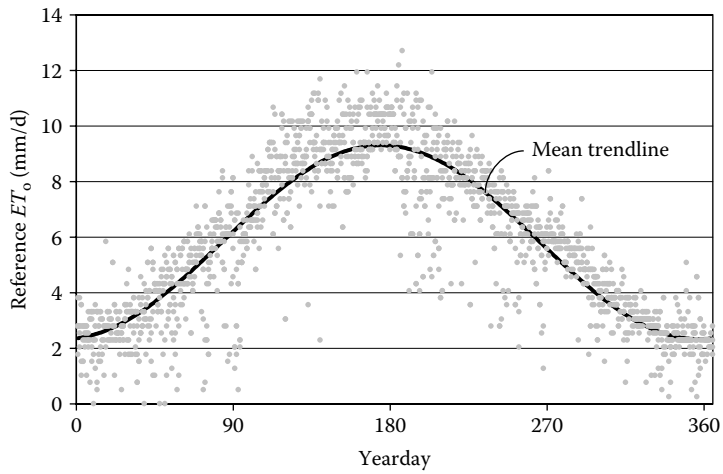



**FIGURE 4.3** Reference evapotranspiration ($ET_o$) as a function of season for Phoenix, Arizona. The mean trendline is shown, along with data from 1995–1999.

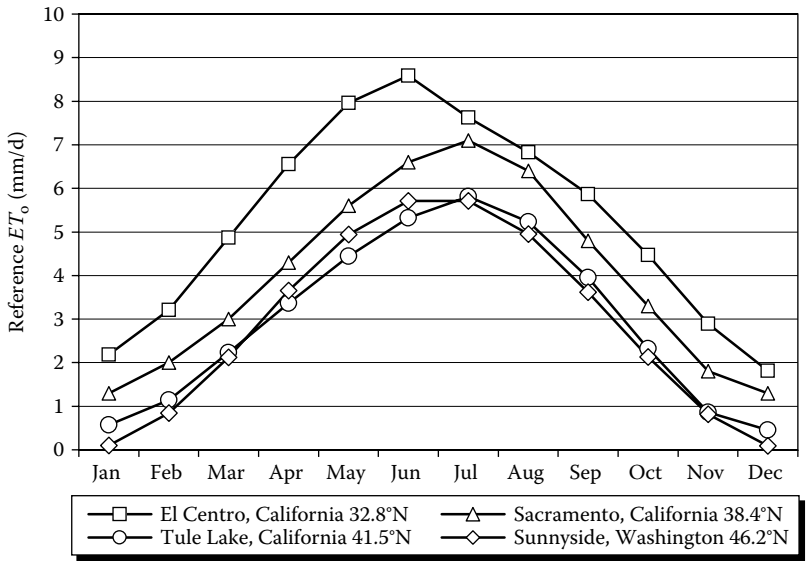



**FIGURE 4.4** Reference $ET_o$ as a function of latitude in the western United States.

Low, 1970), western Nevada, (Kadlec *et al.*, 1987), and southern Manitoba (Kadlec, 1986). The stipulation of a time period in excess of the growing season is important, because the short-term effects of the vegetation can invalidate this simple rule of thumb. The effect of climate is apparently small, as the Florida data of Zoltek *et al.* (1979), for a wastewater treatment wetland at Clermont, are represented by 0.78 times the Class A pan data from the nearby station at Lisbon, Florida, on an annual basis. This multiplier is the same as that recommended by Penman (1963) for the potential evapotranspiration from terrestrial systems.

### Surface Flow Wetlands

The presence of vegetation retards evaporation in FWS wetlands. This is to be expected for a number of reasons, including shading of the surface, increased humidity near the surface, and reduction of the wind at the surface. The presence of a litter layer can create a mulching effect that reduces open water evaporation ($E$). The reported magnitude of this reduction is on the order of 50%. A sampling of reduction percentages for open water evaporation includes: (Bernatowicz *et al.*, 1976): 47%; (Koerselman and Beltman, 1988): 41–48%; (Kadlec *et al.*, 1987): 30–86%. However, these data should not be interpreted as meaning that the wetland conserves water, because transpiration ($T$) can more than offset this reduction.

With plant transpiration offsetting reductions in open-water evaporation, large FWS wetland evapotranspiration and lake evaporation are roughly equal. Roulet and Woo (1986) report this equality for a low arctic site, and Linacre's (1976) review concludes: "In short, rough equality with lakes is probably the most reasonable inference for bog evaporation." Eisenlohr (1966) found that vegetated potholes lost water 12% faster than open water potholes, but Virta (1966) (as cited by Koerselman and Beltman, 1988) found 13% less water loss in peatlands. There is a seasonal effect that can invalidate this general observation in the short term.

The seasonal variation in evapotranspiration shows the effects of both radiation patterns and vegetation patterns. The seasonal pattern of evapotranspiration resembles the seasonal pattern of incoming radiation. During the course of the year, the wetland reflectance changes, the ability to transpire is gained and lost, and a litter layer fluctuates in a mulching function. Agricultural water loss calculations include a crop coefficient to account for the vegetative effect. This is in addition to effects due to radiation, wind, relative humidity, cloud cover, and temperature, and may be viewed as the ratio of wetland evaporation to lake evaporation. The result is a growing season enhancement, followed by winter reductions.

The type of vegetation is not a strong factor in determination of water loss for large, regional wetlands. Bernatowicz *et al.* (1976) found relatively small differences among several reed species, including *Typha*. Koerselman and Beltman (1988) similarly found little difference among two *Carex* species and *Typha*. Linacre (1976) concludes: " ... it appears that differences between plant types are relatively unimportant ... " More recently, Abtew (1996) operated vegetated lysimeters for two years in marshes with three vegetation types: (1) *Typha domingensis*; (2) a mixture including *Pontederia cordata, Sagittaria latifolia,* and *Panicum hemitomon*; and (3) submerged aquatics *Najas guadalupensis* and *Ceratophyllum demersum*. The annual average water losses ($ET_p$) were 3.6, 3.5, and 3.7 mm/d, respectively.

### Subsurface Flow Wetlands

When the water surface is below ground, a key assumption in the energy balance approach is no longer valid: the transfers of water vapor and sensible heat are no longer similar. Water vapor must first diffuse through the dry layer of gravel,

and then be transferred by swirls and eddies up through the vegetation to the air above the ecosystem. Heat transfer to the water must now pass through a porous media in addition to the eddy transport in the air for convective transport, or in addition to radiative transport to the gravel surface. The heat storage capacity of the media is also directly involved because it is in the water. The energy balance approach is still valid, but there are no estimates of the transport coefficients within the porous media. It is therefore necessary to rely on wetland-specific information.

Water budgets were used by Bavor *et al.* (1988) to estimate HSSF gravel bed wetland *ET* for 400 $m^2$ wetlands in New South Wales, Australia. The correlations with pan measurements were (mm/d):

Gravel (no plants)

$$ET = 0.0757\ E_A - 0.028 \text{ mm/d} \quad R^2 = 0.15 \quad 12°C < T_{air} < 25°C \qquad (4.24)$$

Cattails/Gravel (*Typha* spp.)

$$ET = 1.128\ E_A + 0.072 \text{ mm/d} \quad R^2 = 0.72 \quad 12°C < T_{air} < 25°C \qquad (4.25)$$

Bulrush/Gravel (*Schoenoplectus* spp.)

$$ET = 0.948\ E_A - 0.027 \text{ mm/d} \quad R^2 = 0.93 \quad 12°C < T_{air} < 25°C \qquad (4.26)$$

Comparing the gravel (no plants) *ET* results (Equation 4.24) to the vegetated (*Typha* and *Schoenoplectus)* systems (Equations 4.25 and 4.26) clearly shows the strong influence of plant transpiration on *ET* in HSSF wetlands. The gravel effectively cuts off almost all of the evaporative component. Also note that $E_A = 1.25\ ET_o$, so that the annualized crop coefficients ($K_c$ in Equation 4.23) are 1.41 for cattails and 1.19 for bulrushes.

George *et al.* (1998) measured *ET* in HSSF wetlands at Baxter, Tennessee, 6.0 $m^2$ in area and vegetated with *Schoenoplectus validus*. Water loss was reported as 1.2 times $E_A$ for healthy vegetation, but drastically less for heavily damaged vegetation. Noting that $E_A = 1.25\ ET_o$, the annual average crop coefficient ($K_c$) for the Baxter project is estimated to be 1.5.

Fermor *et al.* (1999) investigated *ET* losses from wastewater reed beds (Himely, United Kingdom, 864 $m^2$) and runoff reed beds (Teeside International Nature Reserve, United Kingdom), and computed four types of crop coefficients, based upon different methods of determination of $ET_o$. The regional estimate of $ET_o$ was based upon the assumption of the Penman–Montieth equations, as utilized by the Meteorological Office Rainfall and Evaporation Calculating System (MORECS) in the United Kingdom, calibrated to grass systems on a 40 km × 40 km grid. Results for the Himely HSSF system after maturity are shown in Table 4.4. Water losses are greater than $ET_o$ by a considerable margin, especially in the autumn.

**TABLE 4.4**
**Crop Coefficients for the Himely, United Kingdom, System for 1996**

| Month | *ET* (mm/d) | $ET_o$ (mm/d) | $K_c$ |
|---|---|---|---|
| April | 1.38 | 1.81 | 0.76 |
| May | 2.41 | 2.69 | 0.90 |
| June | 3.84 | 3.10 | 1.24 |
| July | 4.99 | 3.10 | 1.61 |
| August | 6.19 | 2.86 | 2.16 |
| September | 6.30 | 1.86 | 3.38 |
| October | 2.96 | 1.49 | 1.98 |
| **Season** | **4.01** | **2.42** | **1.66** |

*Source:* Data from Fermor *et al.* (1999) In *Nutrient Cycling and Retention in Natural and Constructed Wetlands.* Vymazal (Ed.), Backhuys Publishers, Leiden, The Netherlands, pp. 165–175.

## Size Effects on *ET*

Because many constructed water treatment wetlands tend to be small, it is reasonable to enquire at what size this effect becomes important. There is very little information available on the size effect. The Koerselman and Beltman (1988) study was on a wetland of "less than one hectare," and displayed no large differences from similar studies on larger wetlands. Studies at Listowel, Ontario (Herskowitz, 1986), indicated that lake evaporation was a reasonable estimator of wastewater wetland evapotranspiration for wetlands that aggregated about 2 ha. However, as size is decreased, the advective air energy terms in the energy balance become important at some point, and regional methods are no longer adequate. Ratios to pan and lake evaporation, and to radiation would not be expected to hold.

The use of energy balance information to estimate regional wetland *ET* is predicated on the assumptions of uniform, equilibrated water temperature, and negligible effects of energy contributions from the air passing through the canopy. There are consequently two factors that may increase water losses from treatment wetlands, in comparison to large scale wetlands in the same locality. The first is the potential for incoming warm water to evaporate to a greater extent than regional waters at ambient conditions. This enhancement is greatest at the point of entry, and diminishes along the flow direction. This effect is more fully discussed next; here, it is noted that the change in water temperature to ambient values (95%) typically occurs in about three or four days' nominal travel time for a FWS wetland. A typical detention time for

FWS systems is seven days. Therefore, for warm incoming waters, enhanced *ET* may be expected over the majority of the flow length.

The second factor has to do with the microclimate created by the wetland. Small wetlands are subject to the "clothesline" and "oasis" effects, in which warm dry air can contribute to heat input and to water loss, well in excess of the loss driven by radiation alone. Indeed, this is the entire basis for the Danish willow systems, which are zero-discharge SSF wetlands (Gregersen and Brix, 2001; Brix and Gregersen, 2002; Brix, 2004; Brix, 2006). This effect has also been reported for other FWS and HSSF wetlands. Estimation of the magnitude and distance scale of this effect may be done by considering the energy balance on the air passing through the canopy of the wetland. If the prevailing wind broadsides the wetland, there is convective transfer of heat to the canopy until the air has lost its heat excess over the regional wetland ambient air. Factors such as the leaf area index (LAI), canopy height, and air temperature and humidity influence the energy balance on the air as it moves through the wetland vegetation. Typical wetland widths for the dissipation of the incoming temperature excess and humidity deficit are on the order of 50 to 100 m (Figure 4.5; Brix, 2006).

The crop coefficient $K_c$ represents the ratio of *ET* for a given wetland to potential $ET_o$, which represents the regional large system that is always wet. Values of $K_c$ greater than 1.0 mean that the wetland is losing more water than predicted from radiation via the energy balance. For instance, Bavor *et al*. (1988) found *ET* enhanced by a factor of two over pan evaporation in an open-water, unvegetated wetland 4 m wide by 100 m long. Typically, additional *ET* losses are the greatest for the smallest systems, namely microcosms and mesocosms. Rozkošný *et al*. (2006) studied water losses from *Phragmites* and *Typha* in 0.2 $m^2$ SSF mesocosms (essentially potted plants), which contained 3,000–6,000 g dw/$m^2$ of vegetation. An unvegetated mesocosm with a free water surface (FWS) was the reference. The values of $K_c$ were found to be 5.4 for *Typha*, and 7.3 for *Phragmites*. Mesocosm studies (Snyder and Boyd, 1987) displayed a strong effect of vegetation and its rate of growth (Table 4.5) This is not unexpected, because the plants exhibit strong edge effects in mesocosms, due in large part to canopy overhang for emergent vegetation. However, convective processes are also magnified in mesocosms, and hence even floating plant systems show species differences in water loss rates. For instance, mesocosm studies by DeBusk *et al*. (1983) showed that open water and *Lemna minor* systems had similar annual average water loss (4.5 and 4.1 mm/d, respectively), but *Eichhornia crassipes* was greater (7.5 mm/d). For such small systems, vegetative overgrowth augments meteorological enhancement.

Wetlands with tall vegetation with large leaf area (LAI) will intercept more dry wind, and exhibit larger $K_c$. Therefore, willows with a height of 3–4 m will exhibit $K_c$ up to 2.5 (Danish systems). And, for HSSF wetlands, no vegetation causes a virtual elimination of *ET* (Equation 4.24). It is clear that most HSSF wetlands are small enough to exhibit enhanced evapotranspiration, compared to regional energy balance estimates.

## Timing of ET Losses

The loss of water from the wetland does not occur uniformly over the course of the day, but rather occurs during the daytime hours. This is occasioned by (1) the radiation driving force is only operative during daylight hours, and (2) wind and dry conditions usually also operate during the daytime. As a consequence, *ET* is nearly zero except for a period of about 12 hours at temperate latitudes in summer. During that period, it displays a parabolic curve, with a maximum at

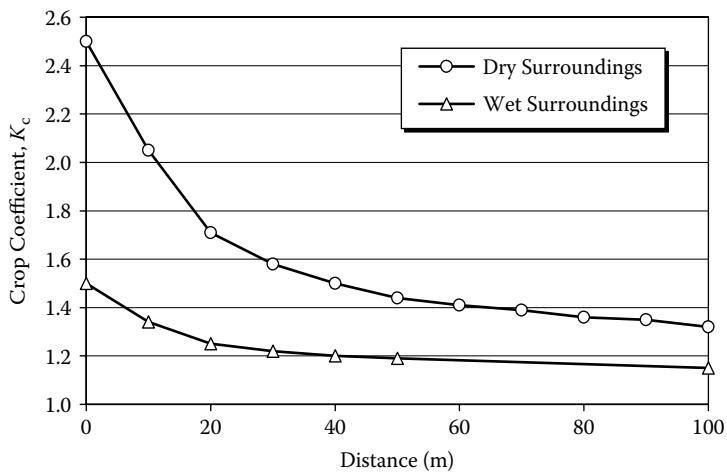



**FIGURE 4.5** Enhanced evapotranspiration for small wetlands due to cross-flow winds. $K_c$ is the crop coefficient, or multiplier on regional evapotranspiration for large wetlands. Conditions of wind and humidity are those typical of Denmark in the warm season. (Data from Brix (2006) Course Notes: Onsite treatment of wastewater in willow systems. Aarhus, Denmark, Department of Biological Sciences, Aarhus University.)

**TABLE 4.5**
**Water Loss from Cattail Wetlands**

| | Open Water *E* (mm/d) | Low Fertilization | | High Fertilization | |
|---|---|---|---|---|---|
| | | *ET* (mm/d) | $K_c = ET/E$ | *ET* (mm/d) | $K_c = ET/E$ |
| May | 5.6 | 7.2 | 1.3 | 7.6 | 1.3 |
| June | 6.2 | 9.9 | 1.6 | 12.0 | 1.9 |
| July | 4.8 | 8.5 | 1.8 | 12.0 | 2.5 |
| August | 4.8 | 7.2 | 1.5 | 10.4 | 2.2 |
| September | 4.7 | 5.7 | 1.2 | 8.0 | 1.7 |
| October | 3.7 | 3.8 | 1.1 | 5.3 | 1.4 |
| **Season** | **5.0** | **7.1** | **1.4** | **9.2** | **1.8** |

*Note:* High fertilization produced peak aboveground biomass of 1,000 g dw/m$^2$ and LAI = 6.5; low fertilization 500 g dw/m$^2$ and LAI = 3.5. Means of triplicate 6 m$^2$ mesocosms.

*Source:* Adapted from Snyder and Boyd (1987) *Aquatic Botany* 27: 217–227.

about midday, reaching about triple the mean daily *ET* loss (Scheffe, 1978; Kadlec *et al*., 1987; Snyder and Boyd, 1987). The result can be strong diel trends in the outflow from the wetland (see Figure 2.5).

### Transpiration: Flows into the Root Zone

Vertical flows of water in the upper soil horizon are driven by gravity and by plant uptake to support transpiration. In an aquatic system, without emergent transpiring plant parts, vertical downflow will be driven solely by gravity. Water infiltration flow is then computed from the water pressure (hydraulic head) gradient between the saturated soil surface and the receiving aquifer, multiplied by the hydraulic conductivity of the soil. If the hydraulic conductivity of the soil layers beneath the root zone is very low, then percolation to groundwater is effectively blocked.

In aquatic and wetland systems with fully saturated soils or free surface water, the meteorological energy budget requires the vaporization of an amount of water sufficient to balance solar radiation and convective losses. Some of this vaporization is from the water surface (evaporation); some is from the emergent plants (transpiration). Emergent plants "pump" water from the root zone to the leaves, from which water evaporates through stomata, which constitutes the transpiration loss (Figure 4.6). Water for transpiration must move through the soil to the roots. That movement is vertically downward from overlying waters in most FWS wetland situations, whereas it is directly from the flowing water in HSSF wetlands. In temperate climates, *ET* ranges from 60 to 200 cm/yr, but is concentrated in that part of the year with greatest solar radiation. Thus, transpiration has the potential to move on the order of one meter per year of water vertically downward to the root zone. This vertical flux of water carries with it the pollutant content of the overlying water, together with soluble materials formed in the root zone. This transpiration-driven pollutant transfer is far greater than the diffusion fluxes (Kadlec, 1999a).

The supply of terrestrial plant nutrients is well known to correlate strongly with this vertical movement of water (Vrugt *et al*., 2001; van den Berg *et al*., 2002; Novak and Vidovic, 2003). Novak and Vidovic (2003) state that "It is important that the transpiration flow that drives nutrient transport can be estimated relatively easily ... " Therefore, to understand wetland nutrient removal, it is necessary to separate the processes of wetland evaporation and transpiration.

This situation is well described in the literature (Nobel, 1999), by considering the canopy and water as separate components of the wetland ecosystem for energy budget purposes. Measurements of the two components of *ET* have shown that shading reduces surface water evaporation, while transpiration continues from the canopy (Kadlec *et al*., 1987). Herbst and Kappen (1999) report that transpiration accounted for 64 ± 6% of *ET* in a *Phragmites* stand, measured over a four-year period. Kadlec (2006c) found approximately 70% of *ET* was due to transpiration in an arid region FWS wetland on an annual basis, but monthly proportions ranged from 45% to 85%.

In a densely vegetated FWS wetland, and in HSSF wetlands, transpiration dominates the combined process of evapotranspiration (Kadlec *et al*., 1987). The fraction *T*/*ET* varies with vegetation density, which in this context is usually characterized by the leaf area index (LAI), defined as the leaf area per unit land/water surface area. Values of the LAI range from less than 1.0 m$^2$/m$^2$ in sparsely vegetated systems, to over 5.0 m$^2$/m$^2$ in densely vegetated systems (Koch and Rawlick, 1993; Nolte and Associates, 1997; Herbst and Kappen, 1999). The corresponding fractions are $0.5 < T/ET < 0.9$ (Shuttleworth and Wallace, 1985). Figure 4.7 shows the LAI dependence of the *T*/*ET* ratio for subtropical conditions (Shuttleworth and Wallace, 1985).

The effects of transpiration and evaporation on wetland pollutant processing in FWS are quite different.

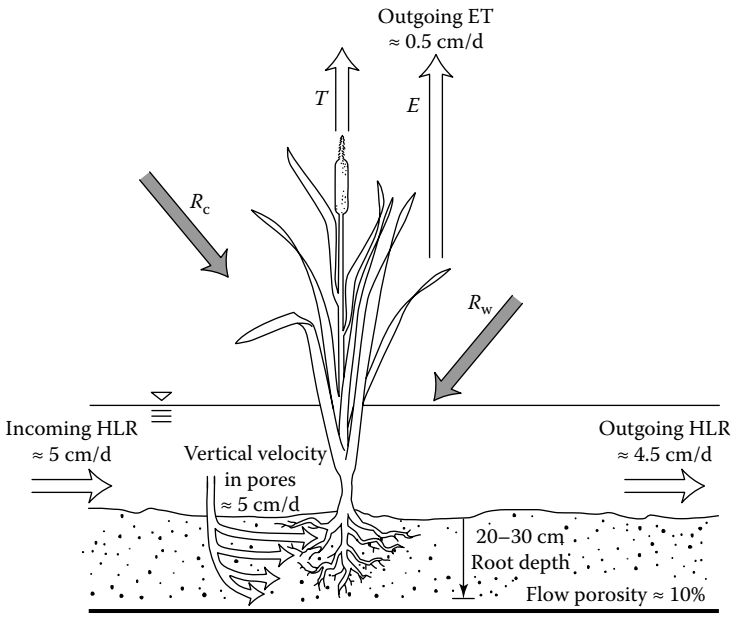



**FIGURE 4.6** Transpiration flows create a vertical flux of water that transports phosphorus from the litter-benthic mat zone down into the root zone. The vertical location of water extraction is dependent on the vertical position and density of the imbibing roots.

Transpiration pulls water into the root zone, and into roots, and therefore overcomes transfer resistances. The water loss occurs at the leaves, and hence heat effects are located in the canopy. On the other hand, evaporation concentrates pollutants in the flowing water, and draws the energy directly from the water column, contributing to wetland water cooling. The transpiration flow may be a minor fraction of wetland throughflow in the case of heavily loaded wetlands. For instance, if the hydraulic loading rate is 5 cm/d, and $T = 0.75ET = 0.75 \times 0.5 = 0.375$ cm/d, then $T/q = 7.5\%$. However, for lightly loaded wetlands, transpiration may be much a more important fraction. For instance, if the hydraulic loading rate is 0.5 cm/d, and $T = 0.75ET = 0.75 \times 0.5 = 0.375$ cm/d, then $T/q = 75\%$.

## 4.3 WETLAND WATER TEMPERATURES

The energy flows that determine water temperature and the associated evaporative losses are shown in Figure 4.1 for a FWS wetland. A treatment wetland may contain one or two thermal regions, depending on water loading (detention time). For long detention times, there is an inlet region in which water temperatures adjust to the prevailing meteorological conditions, and an outlet region in which that adjustment is complete (Figure 4.8). After adjustment, temperature does not change further with distance, or detention time. The value reached is determined by the balance of energy flows and is termed the *balance temperature*. For short detention

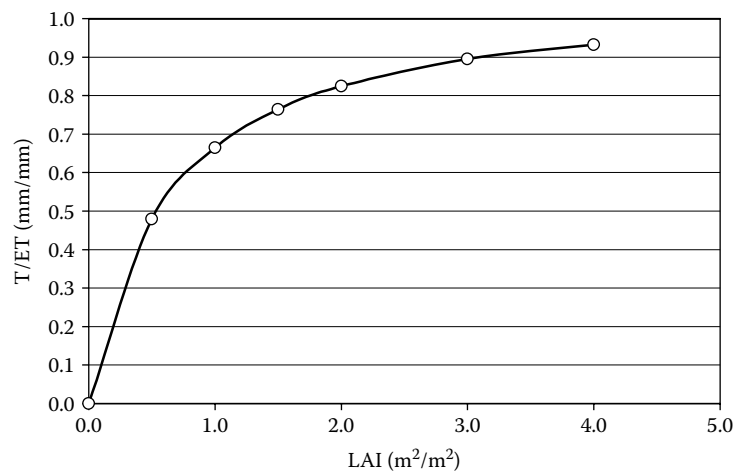



**FIGURE 4.7** Fraction transpiration versus leaf area index (LAI) according to the energy partition model of Shuttleworth and Wallace (1985).

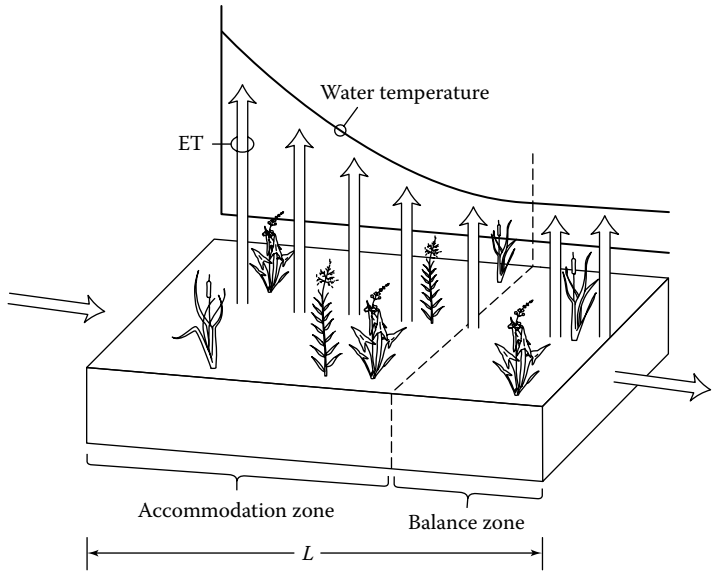



**FIGURE 4.8** Gradients in temperature and evapotranspiration in a wetland. (From Kadlec (2006c) *Ecological Engineering* 26: 328–340. Reprinted with permission.)

times, near the wetland inlet, the adjustment may not be completed, and the balance temperature is not reached. In this adjustment or accommodation region, there will be a difference between the sensible heats of incoming and outgoing water flows; in contrast, they are equal in the balance region. In the balance region, sensible heat of the flowing water is therefore not a factor in the energy budget.

To a very rough approximation, wetland water balance temperatures are linearly related to air temperatures during the unfrozen season (Figure 4.9). In winter, the balance point is just above freezing, as long as liquid water is present. However, this approximation is insufficient to support either the design of wetlands for temperature modulation, or for the determination of the temperature effects on microbial processes. Additionally, the incoming water may have quite a different thermal condition, depending upon the type of pretreatment. Lagoon pretreatment leads to water nearly at wetland temperature, whereas activated sludge effluents are likely to be much warmer in winter. Therefore, in many instances, the inlet section of a treatment wetland will contain water that is at a different temperature than the balance point temperature.

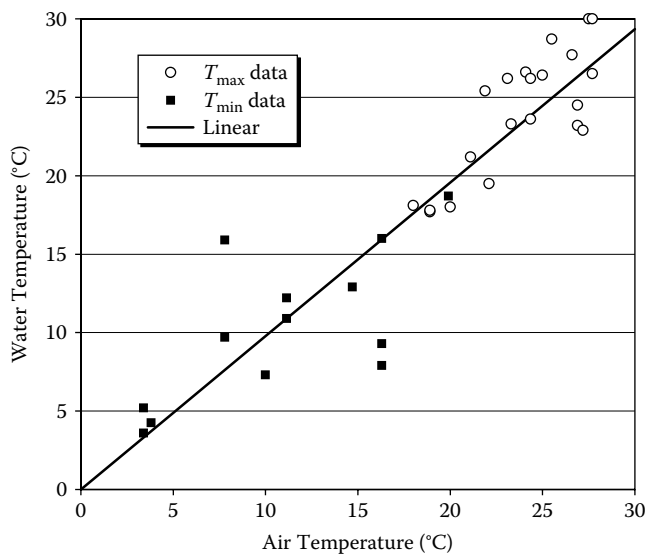



**FIGURE 4.9** Relation between annual maximum and minimum water and air temperatures for FWS wetlands. In general, arid climate systems lie below the line, and humid climate systems lie above. $T_w = 0.98T_a$; N = 36; $R^2 = 0.84$; standard error in $T_w = 3.3$°C.

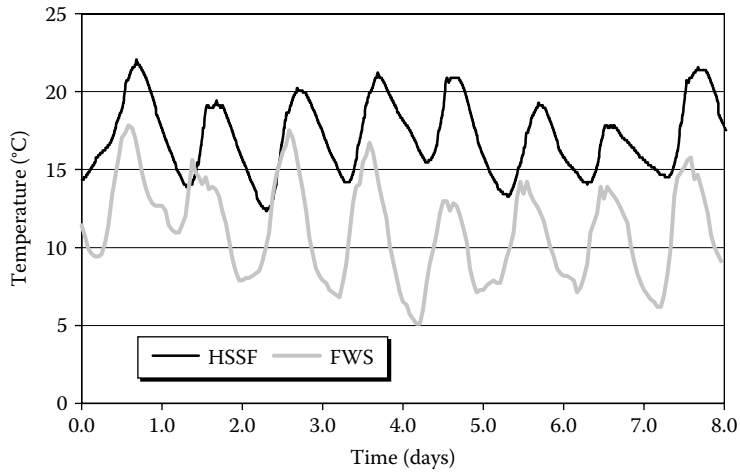



**FIGURE 4.10** Diurnal temperature fluctuations in treatment wetlands. The subsurface flow system was treating dairy wastewater (November 21–27). (Tanner, unpublished data). The FWS wetland was treating municipal lagoon effluent (June 1–8). (Kadlec, unpublished data.)

Clearly, simple rules of thumb are not adequate to characterize wetland temperatures. More detail is developed via the observations and models presented below.

### Short-Term Cycles

The amplitude of the daily water temperature swing depends on the type of wetland in question, and the type and density of vegetation (Figure 4.10). The general pattern is a marked diurnal swing in water temperature, which can be as large as 8 to 10°C in the warm months. Ordinarily, these daily cycles may be averaged to interpret wetland performance, but there are some exceptions. For instance, daily monitoring at the Tres Rios demonstration project was routinely conducted in the early daylight hours, because of the extreme heat later in the day in southern Arizona (Wass, 1997). Interpretation of the diurnal variation indicated that those morning values were about 2°C lower than the daily average. Determination of the temperature coefficients for microbial processes was therefore based upon adjusted temperatures.

### Annual Cycles

The annual cycle of wetland water temperatures in mild to warm climates follows a sinusoidal pattern, with a summer maximum and a winter minimum. In northern climates, the onset of frozen conditions typically is accompanied by under-ice water temperatures of 1–2°C. The sinusoidal model, truncated for frozen conditions, is:

For the unfrozen season ($t_1 < t < t_2$):

$$T_w = T_{avg}\left(1 + A \cdot \cos\left[\omega(t - t_{max})\right]\right) \qquad (4.27)$$

For the frozen season ($t_2 < t < t_1$):

$$T_w = T_o \qquad (4.28)$$

where

$A$ = fractional amplitude of the sinusoid, unitless
$\omega$ = yearly cycle frequency = $2\pi/365 = 0.0172\ d^{-1}$
$t$ = time, Julian day
$t_1$ = ice-out time, Julian day
$t_2$ = freeze-up time, Julian day
$t_{max}$ = time of annual maximum temperature, Julian day
$T_w$ = water temperature, °C
$T_{avg}$ = annual average water temperature, °C
$T_o$ = under-ice water temperature, °C

The various quantities associated with this time series model are illustrated in Figure 4.11. Model fits for two example datasets are shown in Figures 4.12 and 4.13. The Imperial, California, FWS cycle does not require truncation, and the weekly data fit has $R^2 = 0.97$. The Grand Lake, Minnesota, HSSF cycle requires truncation, and the daily data fit has $R^2 = 0.94$.

Three parameters are required for Equation 4.27: $T_{avg}$, $A$, and $t_{max}$. Three are also required for Equation 4.28: $t_1$, $t_2$, and $T_o$. Data from several free water surface (FWS) wetlands were regressed to a truncated, sinusoidal time series model (Table 4.6). Data from two to eight years at each site were folded into a composite annual pattern. From this information, it is seen that the time of maximum wetland water temperature is essentially fixed at $t_{max} = 200 \pm 4$ days (mean ± std. dev., N = 14). Data from HSSF systems is likewise well fit by Equations 4.27 and 4.28 (Table 4.7). For these HSSF wetlands, the time of maximum wetland water temperature is at $t_{max} = 210 \pm 6$ days (mean ± std. dev., N = 12). The difference may be attributed to the thermal lag associated with the gravel media in the SSF wetlands. The under-ice temperature is also in a very narrow range of $1 < T_o < 2$°C. It is therefore acceptable to presume an average value of about 1.5°C as an estimation.

The remaining four parameters are site-specific. The treatment wetland designer will be able to find or estimate the

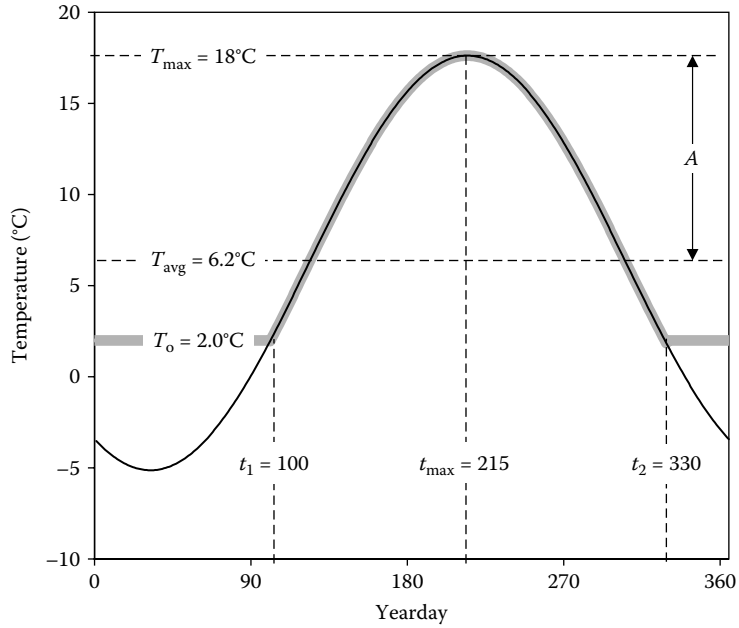



**FIGURE 4.11** Sinusoidal model for cyclic annual time series of wetland water temperatures. (*Note:* This example is for purposes of defining terms, and does not represent any real system.)

times of freeze-up and thaw for the site in question. However, there is not a lot of variability for the time of freeze-up for north temperate climates, $t_2 = 332 \pm 21$ days (Table 4.6 mean ± std. dev., N = 10). There is more variation in the time of spring thaw, with $28 < t_2 < 112$ days. Values of $A$ and $T_{avg}$ are given in Tables 4.6 and 4.7 for a number of treatment wetland sites. In qualitative terms, $T_{avg}$ increases and $A$ decreases as the site moves to warmer latitudes. Because of the symmetry of the sinusoid around $t_{max}$, there is a necessary relation between $t_1$ and $t_2$:

$$(t_{max} - t_1) = (t_2 - t_{max}) \tag{4.29}$$

The remaining two parameters, $A$ and $T_{avg}$, depend upon site climatic conditions. These pertain to the sinusoidal portion of the temperature time sequence, and not to the entire annual profile in the case of truncated profiles. In the case of the truncated annual time series, one further parameter is most conveniently the maximum wetland temperature. The maximum sinusoidal value is then:

$$T_{max} = T_{avg}(1 + A) \tag{4.30}$$

where

$T_{max}$ = maximum wetland water temperature, °C

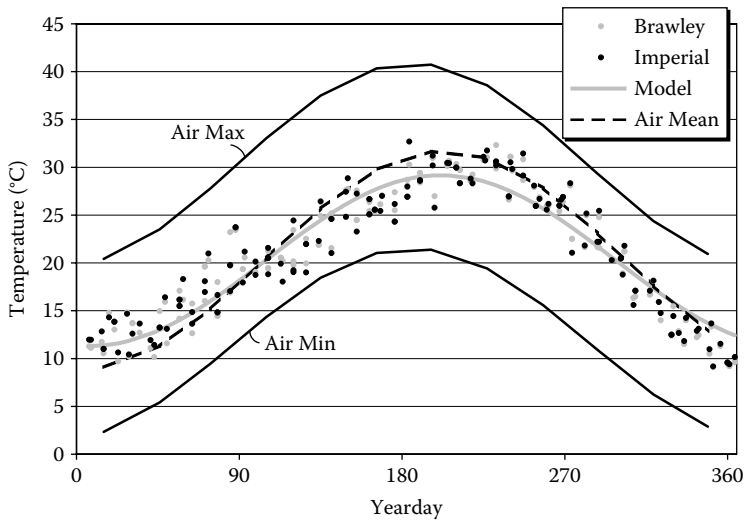



**FIGURE 4.12** Annual pattern of water temperatures in the Imperial and Brawley, California, FWS treatment wetlands, compared to air temperatures. Data spans the four-year period of record. (Unpublished data from Imperial, California, Irrigation District.)

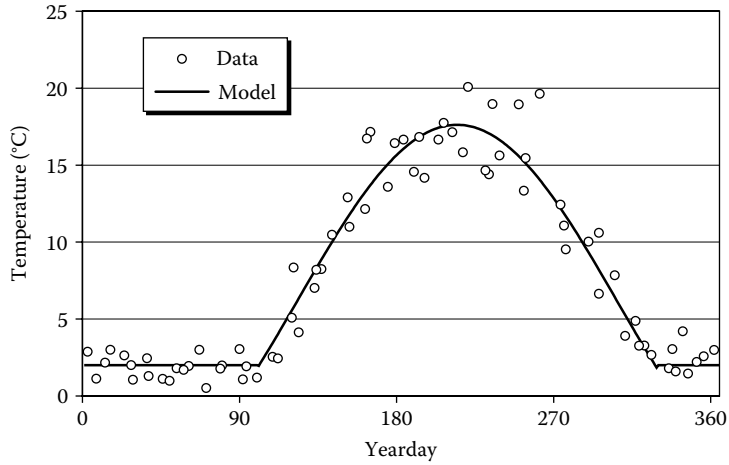



**FIGURE 4.13** Annual pattern of water temperatures in the Grand Lake, Minnesota, HSSF treatment wetland.

**TABLE 4.6**
**FWS Water and Air Temperature Regression Parameters**

| Site | Location | Wetland | $T_{mean}$ | $A$ | $A \cdot T_{mean}$ | $R^2$ | $T_{max}$ | $T_{min}$ | $T_o$ | $t_{max}$ | $t_1$ | $t_2$ |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Musselwhite | Ontario | Wetland | 1.9 | 8.43 | 16.0 | 0.72 | 18.1 | 2.0 | 2.0 | 206 | 112 | 300 |
| Musselwhite | Ontario | Air | −1.0 | −19.96 | 20.0 | 0.99 | 18 | −20.1 | — | 205 | — | — |
| Minot | North Dakota | Wetland | 5.5 | 2.55 | 14.0 | 0.91 | 19.5 | 1.0 | 1.0 | 205 | 90 | 317 |
| Minot | North Dakota | Air | 5.2 | 3.23 | 16.9 | 0.99 | 22.1 | −11.7 | — | 201 | — | — |
| Listowel | Ontario | 3 | 6.3 | 1.80 | 11.3 | 0.95 | 17.7 | 1.5 | 1.5 | 204 | 88 | 320 |
| Listowel | Ontario | 4 | 6.6 | 1.69 | 11.2 | 0.94 | 17.8 | 1.5 | 1.5 | 205 | 86 | 324 |
| Listowel | Ontario | Air | 7.2 | 2.65 | 19.1 | 0.99 | 18.9 | −8.5 | — | 206 | — | — |
| Brighton | Ontario | Wetland | 8.7 | 1.47 | 12.7 | 0.97 | 20.4 | 1.0 | 1.0 | 206 | 80 | 332 |
| Brighton | Ontario | Air | 7.2 | 1.95 | 13.9 | 0.99 | 21.1 | −6.8 | — | 205 | — | — |
| Connell | Washington | Wetland | 9.7 | 0.86 | 8.3 | 0.55 | 18.0 | 1.5 | 1.5 | 201 | 37 | 365 |
| Connell | Washington | Air | 8.5 | 1.36 | 11.6 | 0.98 | 20.0 | −3.0 | — | 199 | — | — |
| Hillsdale | Michigan | Wetland | 13.5 | 0.88 | 11.9 | 0.98 | 25.4 | 2.0 | 2 | 196 | 28 | 365 |
| Hillsdale | Michigan | Air | 8.7 | 1.51 | 13.1 | 0.95 | 21.9 | −4.5 | — | 209 | — | — |
| Commerce | Michigan | Wetland | 11.8 | 0.89 | 10.5 | 0.96 | 22.3 | 2.0 | 2 | 204 | 44 | 365 |
| Commerce | Michigan | Air | 10.6 | 1.19 | 12.6 | 0.99 | 23.3 | −2.1 | — | 202 | — | — |
| Columbia | Missouri | Wetland | 14.3 | 0.68 | 9.7 | 0.99 | 24.1 | 4.6 | — | 201 | — | — |
| Columbia | Missouri | Air | 13.0 | 1.04 | 13.6 | 0.99 | 26.6 | −0.5 | — | 201 | — | — |
| Benton | Kentucky | 1 | 13.4 | 0.73 | 9.8 | 0.87 | 23.2 | 3.6 | — | 196 | — | — |
| Benton | Kentucky | 2 | 14.8 | 0.65 | 9.6 | 0.86 | 24.5 | 5.2 | — | 195 | — | — |
| Benton | Kentucky | Air | 15.1 | 0.78 | 11.8 | 1.00 | 26.9 | 3.4 | — | 200 | — | — |
| New Hanover | North Carolina | Wetland | 18.7 | 0.48 | 9.0 | 0.96 | 27.7 | 9.7 | — | 199 | — | — |
| New Hanover | North Carolina | Air | 17.2 | 0.54 | 9.3 | 0.97 | 26.6 | 7.8 | — | 217 | — | — |
| Imperial | California | Wetland | 20.2 | 0.44 | 11.3 | 0.97 | 29.2 | 11.3 | — | 201 | — | — |
| Imperial | California | Air | 20.3 | 0.56 | 11.4 | 0.95 | 31.7 | 8.9 | — | 204 | — | — |
| Tres Rios | Arizona | Wetland | 16.5 | 0.55 | 9.1 | 0.95 | 25.7 | 7.3 | — | 194 | — | — |
| Tres Rios | Arizona | Air | 21.3 | 0.53 | 11.3 | 0.87 | 32.6 | 10.0 | — | 202 | — | — |
| ENR | Florida | Wetland | 24.4 | 0.23 | 5.6 | 0.77 | 30.2 | 18.7 | — | 196 | — | — |
| ENR | Florida | Air | 24.3 | 0.18 | 4.4 | 0.98 | 28.7 | 19.9 | — | 207 | — | — |

*Note:* Arranged in order of increasing mean air temperature. ***Important***: $T_{mean}$ and $A$ refer to the sinusoidal portion of the annual time series, and are not the overall annual means for truncated times series sinusoids.

**TABLE 4.7**
**Annual HSSF Wetland Water Temperature Cycle Parameters for Systems in Several Geographic Regions**

| Site | Latitude | Years | $T_{mean}$ (°C) | Amplitude | Freeze-Up (Julian day) | Thaw (Julian day) | $t_{max}$ (Julian day) | $R^2$ |
|---|---|---|---|---|---|---|---|---|
| Haugstein, Norway | 60N | 5 | 6.4 | 3.07 | 320 | 100 | 209 | 0.94 |
| Grand Lake, Minnesota | 47N | 4 | 8.0 | 2.73 | 330 | 100 | 215 | 0.94 |
| NERCC 2, Minnesota | 47N | 4 | 7.9 | 2.72 | 330 | 100 | 215 | 0.96 |
| NERCC 1, Minnesota | 47N | 4 | 8.0 | 2.77 | 325 | 100 | 214 | 0.95 |
| Minoa, New York | 43N | 2 | 10.7 | 0.91 | 350 | 80 | 217 | 0.98 |
| Valleyfield 2, Scotland | 56N | 2 | 10.0 | 0.49 | N | N | 208 | 0.85 |
| Valleyfield 3, Scotland | 56N | 2 | 10.5 | 0.47 | N | N | 211 | 0.85 |
| Valleyfield 4, Scotland | 56N | 2 | 10.5 | 0.45 | N | N | 211 | 0.84 |
| Valleyfield 1, Scotland | 56N | 2 | 10.6 | 0.47 | N | N | 205 | 0.83 |
| Benton, Kentucky | 37N | 1 | 13.9 | 0.68 | N | N | 195 | 0.88 |
| Richmond, NSW, *Schoenoplectus* | 34S | 2 | 18.2 | 0.34 | N | N | 214 | 0.86 |
| Richmond, NSW, *Typha* | 34S | 2 | 18.3 | 0.32 | N | N | 208 | 0.88 |
| Richmond, NSW, gravel only | 34S | 2 | 18.5 | 0.38 | N | N | 212 | 0.86 |

*Note:* Systems with freezing conditions all regressed to winter water $T = 2.0$°C, which pertained to the period from freeze-up to thaw. During unfrozen periods, regression was to a sinusoidal pattern. Julian days at southern latitudes are advanced to correspond to northern latitudes.

The sinusoid is then fully defined by choosing $A$ so that $T = T_o$ at $t = t_2$:

$$T_o = T_{avg}\left(1 + A\cos\left[\omega(t_2 - t_{max})\right]\right) \tag{4.31}$$

Solving for $A$ and $T_{avg}$ gives:

$$A = \frac{T_{max} - T_o}{T_o - T_{max}\cos\left[\omega(t_2 - t_{max})\right]} \tag{4.32}$$

and

$$T_{avg} = \frac{T_{max}}{(1 + A)} \tag{4.33}$$

This cyclic model allows quantification of existing data sets, so that information from a variety of wetlands may be compared. It is, however, not predictive, because $T_{max}$ depends upon site climatic conditions.

### Predicting Wetland Water Balance Temperatures

The energy balance also determines the equilibrium water surface temperature (Monteith, 1981), but that aspect of the energy balance is not routinely described or reported in connection with $ET_o$ calculations. However, this temperature is easily retrieved, from any energy balance estimate of $ET_o$. The $ET_o$ loss depends on the difference in water partial pressures between the water or leaf surface and the ambient air above:

$$ET_o = K_e\left[P^{sat}(T_w) - RH \cdot P^{sat}(T_a)\right] \tag{4.34}$$

where

$K_e$ = water vapor mass transfer coefficient, m/d·kPa
$P^{sat}(T_a)$ = saturation water vapor pressure at $T_a$, kPa
$P^{sat}(T_w)$ = saturation water vapor pressure at $T_w$, kPa
$T_a$ = air temperature, °C
$RH$ = relative humidity, fraction
$T_w$ = water temperature, °C

Equation 4.34 shows that the water vapor driven off by solar radiation must be convected into the air according to a water partial pressure difference from the water or leaf surface to the ambient air. The water content of the air is determined by both the air temperature and the relative humidity. At high humidity, water temperatures must be high to sustain the mass transfer gradient; conversely, at low humidity, water temperatures are lower.

The air transport coefficient depends on wind speed, and may be represented as a linear function of the wind velocity. For instance, (ASCE, 1990) suggests:

$$K_e = 1.96 + 2.60u \tag{4.35}$$

where

$u$ = wind speed at two meters elevation, m/s

Equations 4.34 and 4.35 combine to give:

$$P^{sat}(T_w) = P^{sat}(T_a) + \frac{ET_o}{(1.96 + 2.60u)} \tag{4.36}$$

The saturation temperature corresponding to a given vapor pressure may be determined from:

$$P^{sat} = 19.0971 - \frac{5349.93}{(T + 273.16)} \tag{4.37}$$

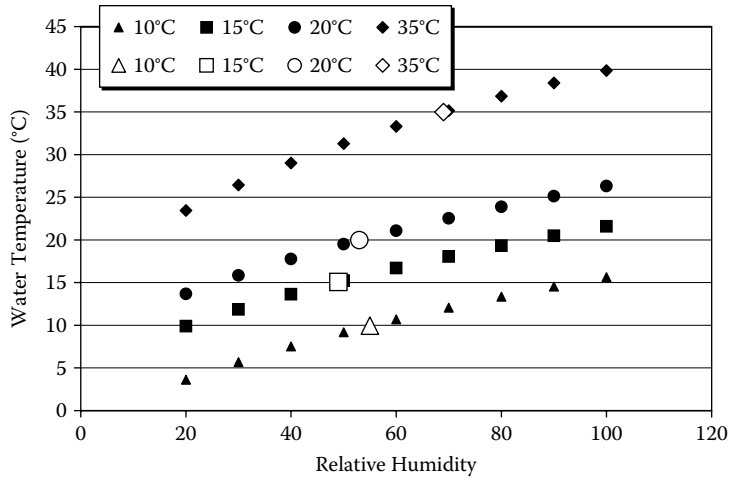



**FIGURE 4.14** Variation of wetland balance temperature with relative humidity and air temperature. These four examples use ($T_a$, $ET_o$) = (20°C, 5 mm/d); (15°C, 4 mm/d); (10°C, 2.5 mm/d); and (35°C, 8 mm/d), with a wind speed of 1.0 m/s. Open points show the humidity at which the balance temperature equals the air temperature.

Equations 4.36 and 4.37 combine to permit estimation of the balance water temperature. Example calculations show that balance water temperatures are approximately equal to air temperatures for relative humidities of about 50% (Figure 4.14). But, in arid regions water may experience significant evaporative cooling upon transit through the wetland (Kadlec, 2006c).

In some instances, such as densely vegetated wetlands in hot climates, the separate energy balances for the above-water canopy and the water may be needed to obtain a reasonable model for wetland water temperatures (Kadlec, 2006c).

### Water Temperature Variability

The deterministic trend expressed in Equations 4.27 and 4.28 represents the central tendency of water temperatures, but there are also stochastic variations. Daily meteorological variations in air temperature, cloudiness, windiness, and relative humidity cause responses in water temperatures, as do changes in incoming water temperatures (see Figures 4.12 and 4.13). Together, these factors create the need to add variability to the trend:

$$T = T_{avg}\left(1 + A \cdot \cos\left[\omega(t - t_{max})\right]\right) + E \qquad (4.38)$$

where

$E$ = stochastic contribution to water temperature, °C

The values of $E$ follow a distribution that is nearly normal for either FWS or HSSF wetlands (Figures 4.15 and 4.16). The breadth of the scatter does not change materially during the course of the year, so that $E$ does not depend upon time ($t$). However, the breadth of the E distribution does depend upon sampling frequency. The standard deviation of the daily Columbia, Missouri, FWS distribution (Figure 4.15) is 2.8°C, whereas for monthly means it is 1.6°C. The standard deviation of the monthly Grand Lake, Minnesota, HSSF distribution (Figures 4.13 and 4.16) is 1.5°C.

### Vertical Temperature Stratification

Water density is a function of temperature; with the unusual property that the maximum density is achieved at 4°C (Lide, 1992). Changes in water temperature may result in layers of water with different densities, and partition the water column into discrete density/temperature layers. Thermal stratification is frequently observed in temperate-climate lake systems. Waste stabilization ponds and lagoons, which have depths in excess of 2 m, often exhibit marked stratification during most portions of the year (Torres *et al.*, 2000; Abis, 2002). These phenomena are thoroughly described in the literature on limnology (Wetzel, 2001).

In the summer, solar radiation raises the temperature of the surface water, reducing its density. The less-dense surface water is buoyant relative to the cooler (and denser) water layer underneath. While thermally-induced vertical stratification in lakes is typically thought of in terms of long-term seasonal effects, daily stratification can also occur due to the diurnal fluctuation in solar radiation.

There are three potential regimes for vertical temperature profiles that have been observed in wetlands and shallow ponds. There may be no vertical profile at all, a condition of no thermal stratification. The second situation is no vertical profile during the night, but the development of surface heating during the daytime hours. This is termed diurnal mixing. The third case is the existence of a vertical temperature gradient throughout the entire 24-hour period, called *stratification*.

Breen and Lawrence (1998) suggest that wind speed is the primary determinant for stratification of shallow ponds in subtropical conditions. They suggest that winds less 0.6 m/s lead to stratification, 0.6–2 m/s lead to diurnal mixing, and greater than 2 m/s provide for full mixing.

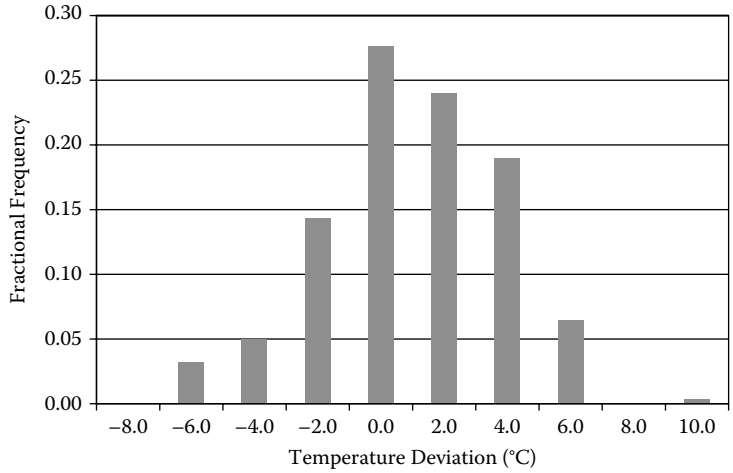



**FIGURE 4.15** Deviations of daily temperatures from the sinusoidal trend for the Columbia, Missouri, FWS treatment wetland.

Condie and Webster (2001) present a criterion for stratification based on pond/wetland models and data from a shallow unvegetated Australian billabong. This criterion is based upon the dimensionless group:

$$S = \frac{\rho c_p u^3}{g \alpha h R_N} \tag{4.39}$$

where

$c_p$ = heat capacity of water, $4.182 \times 10^6$ J/kg·°C
$g$ = acceleration of gravity, 9.8 m/s$^2$
$h$ = water depth, m
$R_N$ = net solar radiation, J/m$^2$·s
$S$ = stratification group, unitless
$u$ = wind speed at 2 m elevation, m/s
$\alpha$ = thermal expansion coefficient of water, $2 \times 10^{-4}$ °C$^{-1}$
$\rho$ = density of water, 1,000 kg/m$^3$

The stratification conditions were found for two different models and the data, to separate as follows:

| | |
|---|---|
| $S > 10^8$ | no stratification |
| $10^8 > S > 10^7$ | diurnally mixed |
| $10^7 > S$ | always stratified |

Condie and Webster (2001) also present an argument that mixing caused by flow through is negligible compared to that caused by even light winds. For conditions of operation of FWS treatment wetlands, these criteria predict no stratification.

The presence of vegetation promotes turbulence induced by water flow, but suppresses mixing caused by wind shear. Emergent vegetation canopies intercept a significant fraction of incident radiation, and thus prevent heating of the top

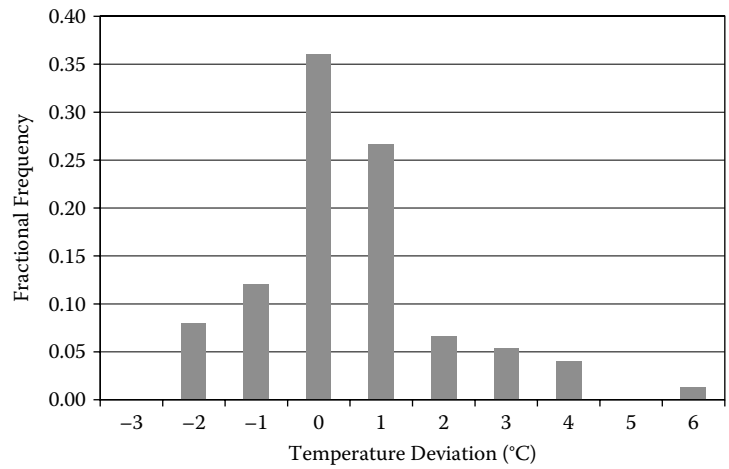



**FIGURE 4.16** Deviations of weekly temperatures from the sinusoidal trend for the Grand Lake, Minnesota, HSSF treatment wetland.

layer of water. Therefore, the most extreme case would be expected for submerged aquatic vegetation (SAV), which can efficiently intercept radiation within the top layer of the water column, due to submerged leaves, yet inhibit wind and flow induced mixing. That is indeed the case for wetlands studied by Chimney *et al.* (2006). The surface of SAV beds was about 2.5°C warmer than water at 40–60 cm depth, based on average profiles over 18 months of the study. In contrast, surface temperatures and those at 40–60 cm depth differed by less than 0.5°C in *Typha* beds.

In HSSF wetlands, vertical stratification is inhibited by the thermal inertia of the wetland bed media. Further, solar radiation does not impinge directly on the water body, but is intercepted by the canopy and top layer of the gravel. As a consequence, stratification is minimal.

In general, temperatures in both FWS and HSSF wetlands are nearly uniform vertically. Although slight thermal stratification does exist in these treatment wetlands, the degree of temperature differential is usually small, and the top-to-bottom variation is typically not more than 1°C (Table 4.8).

In VF wetlands, the flow direction is perpendicular (normal) to vertical stratification mechanisms. The water column experiences a significant fraction of the cyclical soil temperature profiles that produce the dominant heat flux during the cold season. Vertical temperature gradients are not large (Table 4.8). Results from pilot scale VF wetlands indicate that the annual water temperature cycle is not much different from those for HSSF and FWS wetlands. The outlet water temperature is sinusoidal, with a 2°C winter minimum (Figure 4.17). Energy balance models for VF wetlands have been presented by Smith *et al.* (1997).

## The Accommodation Zone

The inlet zone of a treatment wetland exhibits temperature changes, as the water approaches the balance temperature

**TABLE 4.8**
**Vertical Temperature Profiles in Treatment Wetlands**

| HSSF Systems | Bed Depth (cm) | Bottom (cm) | Mid (cm) | Top (cm) |
|---|---|---|---|---|
| **Grand Lake, Minnesota** | 60 | 53 | 23 | 8 |
| Winter | T,°C | 5.0 | 4.9 | 5.9 |
| Summer | T,°C | 16.5 | 17.9 | 21.8 |
| **NERCC, Minnesota** | 45 | 40 | 23 | |
| Spring | T,°C | 5.9 | 5.9 | — |
| Summer | T,°C | 16.2 | 16.1 | — |
| Fall | T,°C | 7.6 | 8.4 | — |
| **Minoa, New York** | 84 | 70 | 40 | 10 |
| *Planted* | | | | |
| Winter | T,°C | 2.7 | 2.5 | 2.0 |
| Spring | T,°C | 8.2 | 8.3 | 8.9 |
| Summer | T,°C | 19.3 | 19.4 | 20.3 |
| Fall | T,°C | 17.7 | 17.7 | 17.7 |
| *Unplanted* | | | | |
| Winter | T,°C | 5.0 | 4.9 | 4.4 |
| Spring | T,°C | 8.1 | 8.1 | 8.2 |
| Summer | T,°C | 20.3 | 20.1 | 20.1 |
| Fall | T,°C | 12.4 | 12.3 | 12.3 |
| **FWS Systems** | **Water Depth (cm)** | **Bottom (cm)** | **Mid (cm)** | **Top (cm)** |
| **ENR, Florida** | | | | |
| July— Cattail | 70 | 60 | 30 | 20 |
| | T,°C | 28.43 | 28.29 | 28.41 |
| July—Open Water | 70 | 60 | 40 | 20 |
| | T,°C | 29.55 | 29.67 | 29.66 |
| October—Open Water | 70 | 60 | 40 | 20 |
| | T,°C | 24.94 | 25.08 | 25.13 |
| **VF Systems** | **Water Depth (cm)** | **Bottom (cm)** | **Mid (cm)** | **Top (cm)** |
| **Niagara-On-The-Lake, Ontario** | | | | |
| March | 90 | 90 | 30 | 0 |
| | T,°C | 3.0 | 1.5 | 1.0 |

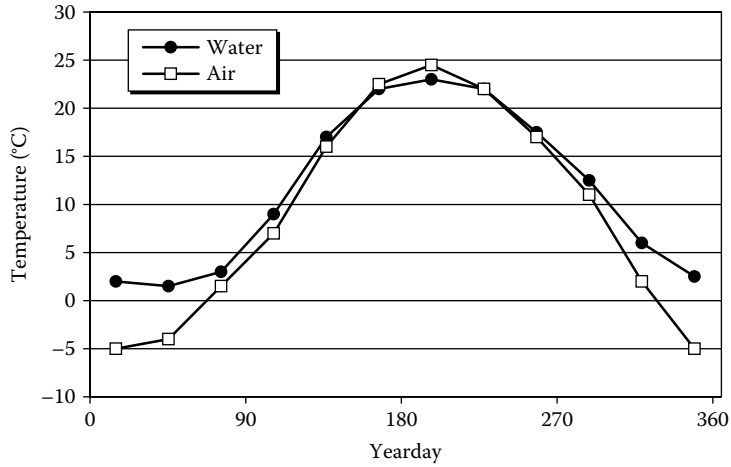



**FIGURE 4.17** Annual progression of temperatures at the Niagara-on-the-Lake, Ontario, VF wetland. The measurement point was at 60-cm depth in a 90-cm downward flow path. The wetland was flood-dosed six times per day, totaling 6.0 cm/d. (Data from Lemon *et al.* (1996) *SWAMP pilot scale wetlands: Design and performance, Niagara-on-the-Lake, Ontario, Canada*. Presented at Constructed Wetlands in Cold Climates: Design, Operation, Performance Symposium; The Friends of St. George: Ontario, Canada.)

(see Figure 4.8). For short detention times (typically less than three days for FWS, and less than one day for HSSF), the adjustment may not be completed, and the balance temperature is not reached (Kadlec, 2006c).

Although the energy budget procedure (see Equation 4.1) is capable of providing a good representation of temperature variation with detention time, it is awkward to use because of the requirement for extensive meteorological data. Further, the partition between transpiration and evaporation is not known *a priori*, and it is only the evaporation component that affects water temperature; transpiration affects canopy temperature. An empirical exponential model may be easily calibrated and used to describe the approach to the balance temperature:

$$\begin{aligned} T_w &= T_b + (T_{wi} - T_b)\exp\left(-\frac{\eta t}{\rho c_p h}\right) \\ &= T_b + (T_{wi} - T_b)\exp\left(-\frac{t}{\tau_A}\right) \end{aligned} \tag{4.40}$$

where

$c_p$ = heat capacity of water, $4.182 \times 10^6$ J/kg·°C
$h$ = water depth, m
$T_w$ = wetland water temperature, °C
$T_b$ = wetland balance temperature, °C
$T_{wi}$ = inlet water temperature, °C
$\eta$ = accommodation coefficient, MJ/m²·d·°C
$\rho c_p$ = volumetric heat capacity of water, MJ/m³·°C
$t$ = nominal detention time to an internal point, d

The quantity $\tau_A = \rho c_p h/\eta$ represents characteristic accommodation time for the wetland water on its travel through the system, during which 63.2% of the change from inlet to balance temperature has been achieved. At $3\tau_A$, 95.0% of the change has been accomplished.

The energy budget analysis suggests that the accommodation coefficient is comprised of radiative, evaporative, and convective components, with the radiative and evaporative portions being dominant (Kadlec and Knight, 1996). Therefore, although the accommodation coefficient is analogous to a convective heat transfer coefficient, and has the same units (MJ/m²·d·°C), it is not predictable from convection correlations as has been presumed in other literature (Reed *et al.*, 1995; Crites *et al.*, 2006), because those correlations ignore radiation, which is the principal heat input in summer, and soil heat retrieval, which is the major energy source in winter.

A further difficulty with previous wetland thermal literature is the reliance upon the assumption that the balance temperature is the air temperature, which is clearly not the case except in summer when the relative humidity is approximately 50%. It is further not the case in winter, when water temperatures are driven to within a degree or two of the freezing point, and not lower. A FWS wetland example illustrates this effect.

### Warm-Up or Cool-Down?

The Tres Rios, Arizona, demonstration project operated 12 research wetlands (0.12 ha) and 4 pilot scale wetlands (about 1.0 ha). The research wetlands were operated at three detention times, approximately quadruplicated. Transects were monitored along the flow direction in the pilot wetlands. Consequently, on any given transect day, data were available for both distance and loading variations of detention time. Water temperatures coming from the advanced treatment plant were warm year-round, varying from 21–34°C.

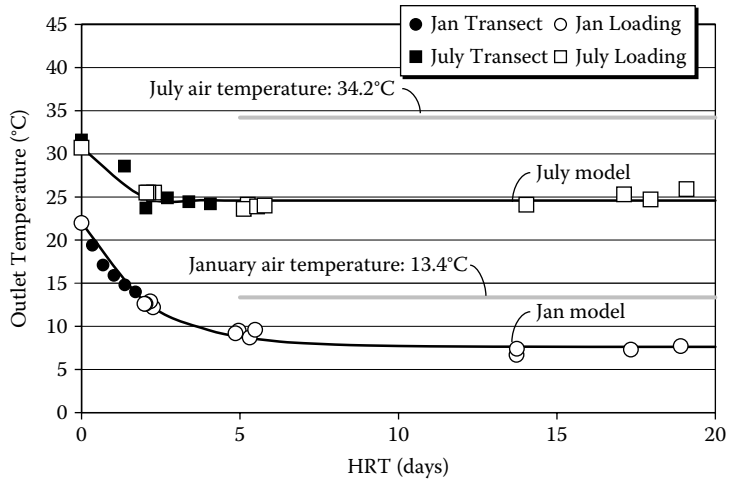



**FIGURE 4.18** Wetland water temperature profiles through various Tres Rios, Arizona, FWS wetlands. Closed symbols represent a transect in wetland H1. Open symbols represent research wetlands operated at different detention times (loadings). The upper data and curves are for July 2; the bottom data and curves are for January 27.

The water cooled on passage through wetlands in both winter and summer (Figure 4.18).

Water temperatures display exponentially decreasing trends from the inlet water $T_i$ to a balance temperature $T_b$. Balance temperatures were 5–10°C lower than the ambient air temperature, due to evaporative cooling in summer, and to evaporation and convection in winter. In summer, the Reed *et al.* (1995) convective model would suggest that the effluent at 31°C should warm up to the air temperature of 34°C, whereas operating data show that it cools to 25°C. An energy balance analysis (not shown) predicted a balance temperature of 26°C. In the summer, the relative humidity at the Tres Rios site is about 30%. Referring to Figure 4.14, it is seen that the corresponding prediction based upon $ET_o$ (Equations 4.36 and 4.37) is 26°C.

This example represents an extreme of very hot arid conditions. Although there are no known temperature transect datasets for wet climates, it is to be expected that wetland balance temperatures would exceed air temperatures under such conditions. This is apparently true for the Hillsdale, New Hanover, and ENR datasets presented earlier, in Table 4.6, which all have long detention times, of about 20 days. Their effluent temperatures would then be balance temperatures. As further evidence of wetland water warm-up, Andradottir and Nepf (2000) found a 1–3°C temperature increase in littoral wetlands in the Boston area.

## How Large Is the Adaptation Zone?

The wetland designer or data interpreter needs to know whether there is an adaptation zone, and if so, how much of the wetland it may occupy. This may be assessed either through estimates of $\eta$, the accommodation coefficient, or through $\tau_A$, the time for 63.2% accommodation (see Equation 4.40). Data for FWS wetlands indicates that adaptation takes on the order of one to three days' detention (Table 4.9). This implies that many FWS treatment wetlands will totally contain the temperature adaptation gradient if the incoming water is colder or warmer than the balance point. As a result, very short detention wetlands may never reach the balance temperature, but most FWS systems will have an exit zone at the balance temperature.

The situation is different for HSSF wetlands, because of the thermal inertia of the media. Under arid conditions, for instance, evaporation has to cool the gravel as well as the water. Further, transpiration is probably more important than evaporation in HSSF systems than in FWS systems, as suggested by comparing Equation 4.24 to Equations 4.25 and 4.26. Nonetheless, HSSF water temperatures adapt during transit if there is a disparity between the incoming water temperature and the wetland balance temperature.

**TABLE 4.9**
**Accommodation Coefficients (MJ/m²·d·°C) for FWS Wetlands and 63% Change Detention Times ($\tau_A$) for Tres Rios, Arizona; Orlando, Florida, Easterly; and Sacramento, California, Wetlands**

| Wetland | Mean ($\eta$) | $\tau_A$ (days) | N |
|---|---|---|---|
| Tres Rios Research | 0.97 | 1.47 | 240 |
| Tres Rios Hayfield 1 | 0.57 | 1.80 | 23 |
| Tres Rios Hayfield 2 | 0.62 | 1.67 | 22 |
| Tres Rios Cobble 1 | 0.27 | 1.70 | 10 |
| Tres Rios Cobble 2 | 0.43 | 1.69 | 11 |
| Sacramento 3 | 2.50 | 0.78 | 2 |
| Sacramento 5 | 1.33 | 0.98 | 3 |
| Sacramento 7 | 1.45 | 2.11 | 4 |
| Sacramento 9 | 0.65 | 3.70 | 2 |
| Orlando Easterly | 0.61 | 3.07 | 4 |

*Note:* N = number of transects or wetland months (research cells).

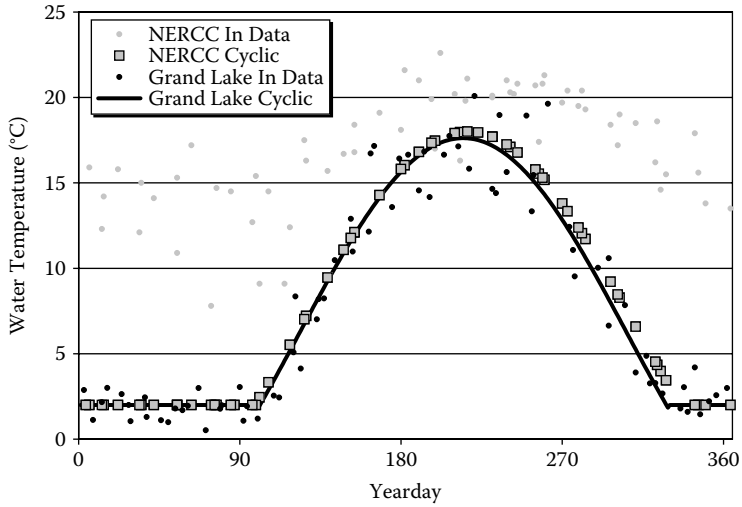



**FIGURE 4.19** Annual temperature pattern for water in the Grand Lake and NERCC, Minnesota, HSSF wetlands. (From unpublished data; for more information, see Kadlec (2001b) *Water Science and Technology* 44 (11/12): 251–258.)

This was the case at the NERCC wetlands in Minnesota, which had warm water entering. The neighboring Grand Lake wetland received water at the local soil temperature. Both produced the same temperature effluents (Figure 4.19) due to similar energy flows. The NERCC HSSF wetlands had accommodation coefficients (η values) averaging 0.55–0.70 MJ/m$^2$·d°C, corresponding to 95% adaptation in three to four days' detention. These values are similar to those for FWS systems (Table 4.9).

Longitudinal profiles were measured in the HSSF wetlands at Minoa, New York (Liebowitz *et al.*, 2000). In addition to measurements of temperatures at points along the flow path, three wetlands were operated in parallel at different hydraulic loading rates, hence different detention times. There is an exponential decline in temperature with nominal travel time (Figure 4.20). Cell 3 had short detention, and was entirely in the accommodation mode. Cells 1 and 2 had longer detention, and were mostly in the balance mode. Note that although the profile is for February, with an air temperature of about −4°C, the profiles trend to a balance temperature of +2°C.

Data for horizontal subsurface flow wetlands indicates that adaptation takes on the order of one day's detention (Table 4.10). This implies that many HSSF treatment wetlands will totally contain the temperature adaptation gradient if the incoming water is colder or warmer than the

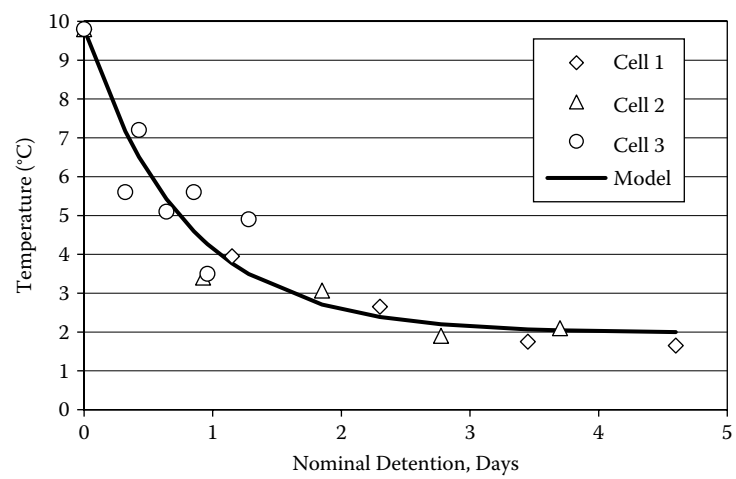



**FIGURE 4.20** Temperature decrease in the flow direction through the three cells of Minoa, New York, HSSF wetlands on February 15, 1996. The three cells were operated in parallel at different detention times. Data points are averages for two depths, and one to three cross-flow positions. (Data from Liebowitz *et al.* (2000) *Subsurface flow wetland for wastewater treatment at Minoa, New York*. New York State Energy Research and Development Authority: New York.)

**TABLE 4.10**
**Accommodation Coefficients (MJ/m²·d·°C) and 63% Change Detention Times ($\tau_A$) for HSSF Wetlands**

| Wetland | Season | Balance $T$ (°C) | $\tau_A$ (days) | Mean ($\eta$) | N |
|---|---|---|---|---|---|
| **NERCC 1** | Spring | 7.7 | 1.86 | 1.0 | 21 |
| | Summer | 16.0 | 1.47 | 1.3 | 7 |
| | Autumn | 9.8 | 1.29 | 1.5 | 9 |
| | Winter | 1.7 | 1.03 | 1.8 | 20 |
| **NERCC 2** | Spring | 7.3 | 1.38 | 1.4 | 21 |
| | Summer | 15.7 | 1.69 | 1.1 | 6 |
| | Autumn | 10.1 | 0.82 | 2.3 | 9 |
| | Winter | 1.5 | 0.91 | 2.1 | 20 |
| **Minoa** | Spring | 7.1 | 0.82 | 3.9 | 1 |
| | Summer | — | — | — | — |
| | Autumn | 16.7 | 0.93 | 3.4 | 1 |
| | Winter | 2.0 | 0.78 | 4.1 | 1 |
| **Sacramento** | Spring | — | — | — | — |
| | Summer | 21.3 | 1.58 | 1.6 | 8 |
| | Autumn | 12.19 | 1.14 | 2.2 | 8 |
| | Winter | 10.81 | 0.39 | 6.5 | 3 |

balance point. As a result, most HSSF wetlands will operate over most of their length at the balance temperature.

## 4.4 COLD CLIMATES

Treatment wetlands that operate in cold (subfreezing) environments face several unique design challenges. During periods below freezing, the water temperature can no longer be approximated by air temperatures once an ice layer forms on the wetland. Effluent water temperatures will be 1 to 2°C, and the thickness of the ice layer becomes a design consideration. The formation of an ice layer will reduce the depth of the water column, reducing detention times, unless the water level is increased in the fall to accommodate the anticipated thickness of the ice layer. As a result, FWS wetlands in cold climates are often designed with additional freeboard in order to accommodate the anticipated layer of ice. Energy balance calculations are required to determine the extent of ice formation.

Ice thickness can vary significantly from year to year due to variations in snowfall and temperature. The principal factor is the insulation provided by the snow layer. Areas of emergent wetland vegetation are much more effective in trapping snow than unvegetated areas. Therefore, the thickness of ice in wetlands is much less than in adjacent lakes or frost depths in nearby uplands. Due to the spatial variability within the wetland, and year-to-year variations in winter conditions, simplifying assumptions are typically used to estimate ice formation.

The options that may be used for FWS treatment wetlands in cold climates include:

- Full year-round discharge, allowing for ice formation
- Restricted winter discharge accompanied by partial pond storage, and accelerated discharge through FWS treatment wetlands during the unfrozen season
- Storing water in ponds over the frozen season, and discharge through FWS treatment wetlands during the unfrozen season

These design options are explored in Chapter 17. HSSF wetlands provide further options, including:

- Added insulation, supported by the bed media or standing dead plants and thus kept out of the water. Mulch is one option (Wallace *et al.*, 2001), and is discussed in detail in this chapter. Straw may be used to supplement the standing dead plant material. Blankets, supported by the standing dead plant litter, have also been used.
- Lowered water levels, to create a layer of dry media (Jenssen *et al.*, 1994a).
- An ice layer on top of dry media. This is accomplished by raising water levels slightly above the media at the time of freeze-up. After the surface water freezes, the water level is dropped below the media surface, creating a dry media gap sealed by ice (Jenssen *et al.*, 1994a; Mæhlum, 1999).
- Using deep beds that allow for ice formation and retain capacity to pass water under the ice (Jenssen *et al.*, 1996).

In this section, methods for estimating the extent of ice formation are presented. Ice cover in wetlands causes the energy balance to split into a balance on the canopy and a separate balance on the water and ice below. It is the latter that is of interest in understanding the degree of ice formation. Radiation and vaporization are no longer factors for the water-side balance, because the ice layer blocks these processes from the underlying water.

**TABLE 4.11**
**Example of the Cumulative Effect of Insulation Layers for an HSSF Wetland**

| | Thickness (cm) | Thermal Conductivity (MJ/m·d·°C) | Resistance (MJ/m·d·°C$^{-1}$) |
|---|---|---|---|
| Air above/in canopy ($U = 0.3$) | — | — | 3 |
| Snow | 25 | 0.010 | 25 |
| Peat mulch | 10 | 0.005 | 20 |
| Dry gravel | 5 | 0.026 | 2 |
| **Total** | — | — | **50** |



**FIGURE 4.21** The inlet zone of this FWS wetland at Hillsdale, Michigan, remains unfrozen in this February 1, 2001 photo. Downstream portions of the wetland are fully ice and snow covered. Note the preferential channel extending outward to the muskrat mound.

## Spatial Extent and Distribution of Ice in FWS Wetlands

The point discharge introduction of warm water into either a constructed or natural treatment wetland causes an unfrozen, un-snow-covered inlet area to persist even in the event of extremely cold air temperatures. As the water moves out into the wetland, the incoming exotherm is dissipated, and a snow and ice cover becomes possible. This cover may consist of snow, ice, or a combination, depending on the vegetation density. If the discharge is into an unvegetated inlet zone, snow trapping is not possible, and ice covers the inlet pond in areas away from the discharge point. Flow then proceeds away under or over the ice. If the inlet zone is densely vegetated, snow may be held up above the water by standing dead vegetation or by a floating litter layer. In that event, flow from the unfrozen, uncovered inlet area proceeds away under a snow blanket.

When warm water enters a FWS wetland during freezing conditions, the incoming sensible heat may be sufficient to maintain an ice-free zone near the inlet, for at least part of the frozen season (Figure 4.21). At the Hillsdale, Michigan, wetlands, the unfrozen zone advances and retreats during freezing temperatures, depending upon short-term meteorological conditions. A complete ice cover may form under very cold conditions, only to have open water appear during less-cold, but freezing conditions. The incoming water is from venting groundwater, which has an annual temperature cycle from 4 to 21°C, with a mean of 12.5°C. It is this sensible heat that is capable of overcoming freezing for a short period of travel time.

This adaptation zone is not easily amenable to model calculations, because all of the terms in the energy balance (see Equation 4.1) are operative. A rough approximation may be obtained from Equation 4.40. An assumption of the accommodation coefficient must be made, based on Table 4.9, for instance. It is also necessary to assume that the balance temperature is approximately equal to the air temperature for the subfreezing conditions:

$$T = T_a + (T_i - T_a)\exp\left(-\frac{U}{\rho c_p v}y\right) \tag{4.41}$$

Equation 4.41 is solved for the distance at which the water temperature reaches 0°C. For instance, suppose $v = 4$ m/d, $T_a = T_b = -5$°C, $T_i = 5$°C, and $U = 1.0$ MJ/m²·d·°C. Then:

$$0 = -5 + (5 - (-5))\exp\left(-\frac{1}{(4.182)(4)}y\right)$$

$$y = 12 \text{ m}$$

These conditions approximate those for Figure 4.21, and correctly predict a small zone of open water at the inlet. Further down the flow path, for about 90% of the wetland, ice is present, and the energy balance changes markedly for under-ice conditions. The balance temperature increases to 1°C, because heat losses are reduced by the ice and snow cover.

The ability of warm incoming water to reverse ice formation has important consequences for system operation. At the NERCC, Minnesota, HSSF site, shutdown due to accidental damage caused the wetland water to freeze. But upon reinstatement of flow, the wetlands regained flow capacity, and returned to unfrozen operation. This self-healing capacity is fortunate, but may not be relied upon for routine operations.

The overall ice cover for the wetlands responds to the air temperature (Figure 4.22). When the mean air temperature drops below zero, ice formation quickly covers most of the wetland, excepting the inlet zone. In most years, this cover remains intact until the mean air temperature rises above freezing, whereupon the ice disappears rapidly, over a week or two.

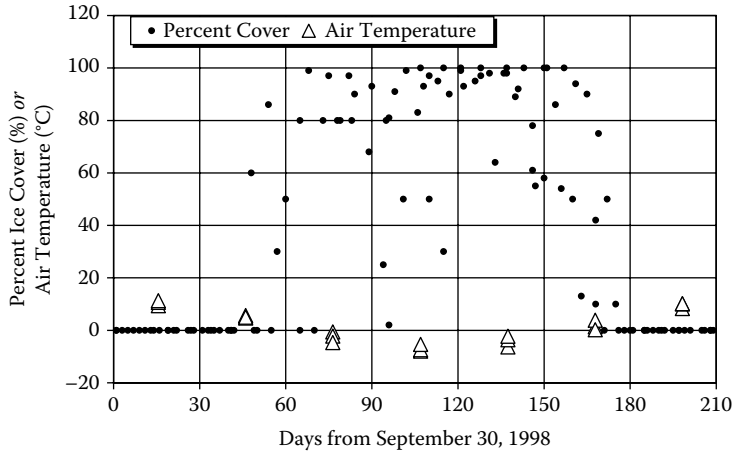



**FIGURE 4.22** Extent of the ice cover at the Hillsdale, Michigan, FWS treatment wetlands. Data span four winters, for four wetland cells. The average monthly air temperature swing was from –4 to 22°C, with an annual mean of 9°C. Note that the proverbial "January thaw" was indeed experienced, with nearly complete loss of the ice cover during January.

## Ice in Quiescent Ponds

An approximate estimating method for determination of ice thicknesses is based upon the length and intensity of sub-zero conditions. Ice formation in still water at a constant cold surface temperature proceeds according to the well-known Stefan model (Ingersoll *et al.*, 1948):

$$h_{ice} = a\sqrt{(-T_s \cdot t)} \tag{4.42}$$

where

$a$ = constant, m/(°C·d)$^{0.5}$
$h_{ice}$ = ice thickness, m
$t$ = time, d
$T_s$ = ice top surface temperature, °C

Crites *et al.* (2006) suggest that the value of $a$ = 0.027 m/(°C·d)$^{0.5}$ for open water, 0.018 for open water with snow, and 0.010 for dense vegetation and litter; however, no calibration data or source are offered.

However, it must be noted that Equation 4.42 is for still, open waters, with no incoming sensible heat, and a fixed below-zero temperature at the top ice surface, none of which prevail in treatment wetland projects. Snow insulation, heat gain from soils, and moving wind and water all combine to slow ice formation. Consequently, the predicted values of ice thicknesses for wetlands are very conservative, often by factors of two or three. For instance, winter operation has been proven feasible at FWS projects located in Ontario, where the water would be predicted to freeze to the bottom according to Equation 4.42. Consequently, a more detailed energy balance is required for treatment wetland estimates.

## The Balance Condition for Under-Ice Flow

The system chosen for energy balancing is selected to be the water under the ice, plus the wetted gravel matrix for the case of HSSF wetlands (Figure 4.23). Any storages of sensible heat in the ice layer, or in the wetland water body, are ignored. Any net heat loss is compensated by ice formation and cooling:

$$(H_a - G) = \left(\rho\lambda_f \frac{dh_{ice}}{dt}\right) \tag{4.43}$$

where

$G$ = heat gain from deep soil, MJ/m$^2$·d
$H_a$ = heat loss to air, MJ/m$^2$·d
$h_{ice}$ = ice thickness, m
$q$ = hydraulic loading, m/d
$t$ = time, d
$T$ = water temperature, °C
$y$ = distance, m
$\lambda_f$ = heat of fusion of water, MJ/kg
$\rho$ = density of water, kg/m$^3$

The heat loss to air may be represented as a heat transfer coefficient times a temperature difference:

$$H_a = U(T - T_a) \tag{4.44}$$

where

$T_a$ = air temperature, °C
$U$ = overall heat transfer coefficient, MJ/m$^2$·d·°C

The balance condition prevails after the water has lost any excess sensible heat during its initial travel distance ($\partial T/\partial y = 0$), and has achieved a fixed temperature ($T_b$):

$$\lambda_f \frac{dh_i}{dt} = U(T_b - T_a) - G \tag{4.45}$$

The overall heat transfer coefficient ($U$) includes components due to air-side and water-side boundary layers, plus the ice

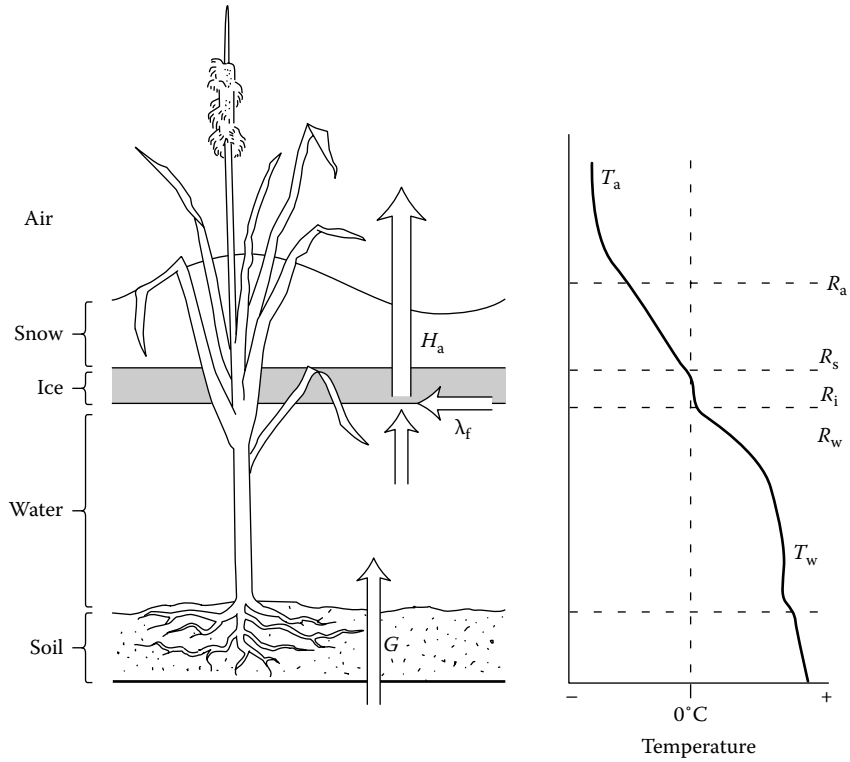



**FIGURE 4.23** Temperature gradients through the vertical profile of a FWS wetland in winter. Partially frozen, balance conditions are assumed.

and snow layers, which greatly decrease the heat loss. The overall heat transfer coefficient is comprised of the several layer components:

$$\frac{1}{U} = \frac{1}{U_{water}} + \frac{h_{ice}}{k_{ice}} + \frac{h_s}{k_s} + \frac{h_m}{k_m} + \frac{1}{U_{air}} \tag{4.46}$$

where

$h_{ice}$ = ice thickness, m
$h_m$ = mulch thickness, m
$h_s$ = snow thickness, m
$k_{ice}$ = thermal conductivity of ice, MJ/m·d·°C
$k_m$ = thermal conductivity of mulch, MJ/m·d·°C
$k_s$ = thermal conductivity of snow, MJ/m·d·°C
$U_{water}$ = water to ice heat transfer coefficient, MJ/m²·d·°C
$U_{air}$ = snow to air heat transfer coefficient, MJ/m²·d·°C

The under-ice, water-side heat transfer coefficient has not been the subject of research. It is tempting to estimate the value based upon known relationships simple geometries. There are relationships for laminar and turbulent flow near flat plates, but these do not deal with the tortuous flow path and under-water obstructions (Welty *et al.*, 1983). There are also relations for arrays of pipes and packed beds, but these do not apply to the bounding walls of the enclosure (Welty *et al.*, 1983). Based on these other geometries, the water-side heat transfer coefficient is expected to be higher than the air-side coefficient for FWS wetlands.

The purpose of energy balance calculations for the frozen season is the estimation of either the amount of ice formed (FWS), or the amount of insulation needed to prevent ice formation (HSSF) (Wallace and Knight, 2006). The conservative assumption is that there is zero resistance to heat transfer on the water side of the air–water interface ($1/U_{water} = 0$), for both FWS and HSSF wetlands.

### Example Detailed FWS Ice Calculation

Because of the time series progressions of air temperature and other variables, calculations require spreadsheet techniques for the solution of Equations 4.45 and 4.46, on a daily basis. There is an accommodation zone, in which incoming warm water cools to the winter balance point of 1–2°C. This example focuses upon energy fluxes in a downstream zone, in which that balance condition prevails. The driving force for ice formation is determined by the air temperature, which is presumed to follow a sinusoidal time progression fitted to data from 1997–1999 from Duluth, Minnesota. The annual swing in mean daily temperature is –11 to 19°C:

$$T_{air} = 4.2\left(1 + 3.57\cos\left[0.01721(t - 204)\right]\right) \tag{4.47}$$

This leads to 150 days of freezing air temperatures, with a mean of –7°C, from November 8 to April 7. The water temperature in the balance zone is assumed to be 2°C for the frozen season. The heat loss transfer coefficients for the air-side

of the ice is assumed to be 0.15 MJ/m²·d·°C. For illustration, a snow-free condition is presumed (worst case). Consequently, the heat transfer resistance to loss to air is comprised of just three components: water-side, ice, and air-side.

Heat from the underlying ground tends to counterbalance heat loss to the atmosphere. For this example, this heat addition is assumed to be that of the cyclical model described above. Importantly, energy is returned from the soil to the water during most of the freezing period. The pattern is given by Equation 4.17:

$$G = \left[\frac{0.05(8.71)}{0.93}\right] \cdot \left(\cos\left[0.01721\left(t-195\right)\right] - \sin\left[0.01721\,(t-195)\right]\right) \quad (4.48)$$

The peak heat addition of 0.66 MJ/m²·d occurs early in the freezing period, on November 30. After February 28, soil heat supply has been exhausted.

For these extreme conditions, 41 cm of ice are forecast (Figure 4.24). There were 1,058 degree-days below zero,

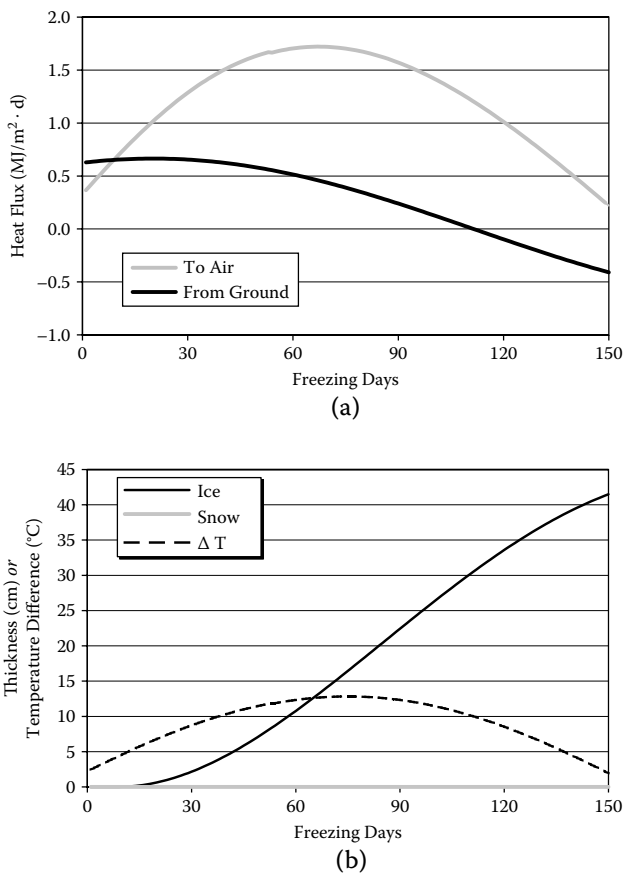



**FIGURE 4.24** Progression of heat flows (a) and ice thickness (b) for an example FWS wetland with no snow. Note that the soil heat and warmer air temperatures slow ice formation during the early portion of the freezing period, and warmer air temperatures slow it during late winter.

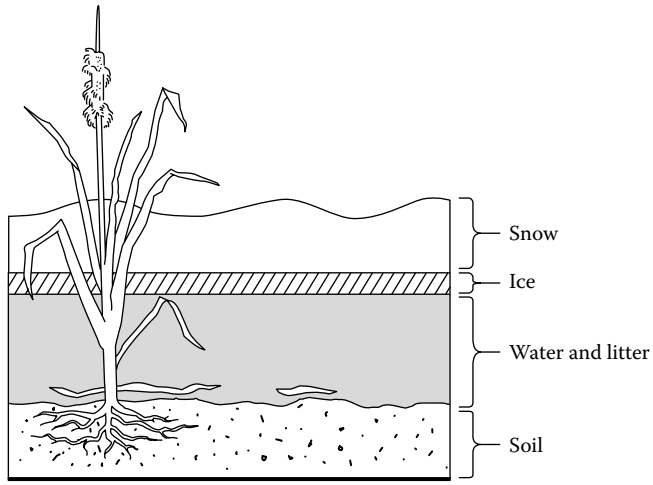



**FIGURE 4.25** Cross section of a FWS wetland in winter.

resulting in a Stefan prediction of $(0.027 \times (1{,}058)^{0.5})(100) =$ 88 cm, which is much larger.

A common occurrence is the collection of snow on top of the ice layer (Figure 4.25). If snow is considered, there is a considerably different result for the modeled ice thickness. A presumed pattern of snow accumulation over the first 20 days, persisting until March at 15 cm depth, then melting over 20 days, produces only 16 cm of ice. Further, the start of ice formation is delayed until mid-December. We note that this early winter phenomenon of snow in the canopy and unfrozen water beneath is common in cold temperate wetlands (see Figure 4.26).

## Insulation of HSSF Wetlands

Because HSSF systems can be insulated by the addition of dry gravel and mulch layers, the balance condition energy fluxes can be modified to prevent ice formation (Henneck *et al.*, 2001; Wallace *et al.*, 2001; Kadlec, 2001b; Wallace and Knight, 2006). These layers add heat flow resistance over and above that which occurs naturally in the wetland, (which include standing dead, litter, and the snow trapped in the senesced vegetation). These natural insulation effects can be very important (see Figures 4.27 and 4.28), and may in fact be one of the most important thermal functions of the vegetation during the winter months.

Equation 4.46 provides the basis for evaluating insulation requirements. It is more intuitive to deal with heat flow resistances $R$ ($R = 1/U$ or $R = h/k$), which are normally used in the insulation business. Basically, there is a need to estimate the total "$R$ factor" needed, and to then calculate the amount of this that must be supplied in the form of mulch or dry gravel layers. As indicated in Equation 4.46, contributions are provided by the air above the wetland canopy, the litter and snow layer, mulch, and dry gravel.

The only heat source is the return flux from deep soils underneath the wetland, which varies through the winter as



**FIGURE 4.26** Winter conditions at a constructed FWS wetland near Hillsdale, Michigan. Note the snow that is held up out of the water by standing dead and litter.

described by Equation 4.17. Heat loss to the atmosphere is driven by the mean daily temperature difference between the water (typically about 2°C) and the air (typically ranges from 0°C to −10°C). It is possible to allow the balance temperature to drop to zero during this period. The "bottleneck" for heat loss therefore typically occurs late in winter, when soil heat return is minimal, but the air temperatures are still subzero. The month of March brings increases in solar radiation, which raise the snow surface temperature, and thus decrease the driving force for freezing. The focus is then on the month of February, during which the mean energy flows are set by:

$$\frac{(T_b - T_a)}{R} = G - (\rho c V)\frac{dT_w}{dt} \tag{4.49}$$

where

$c$ = heat capacity of water plus gravel, MJ/kg·°C
$G$ = heat gain from deep soil, MJ/m$^2$·d
$V$ = volume of water plus gravel in bed, m$^3$/m$^2$
$R$ = overall heat transfer resistance, MJ/m$^2$·d·°C
$t$ = time, d
$T_w$ = water temperature, °C
$T_a$ = air temperature, °C
$T_b$ = water balance temperature, °C
$\rho$ = density of water plus gravel, kg/m$^3$

The ground heat flux in February was approximately 0.125 MJ/m$^2$·d at the Grand Lake and NERCC sites. The air temperature for those sites averages −9.4°C in February. Allowable cooling of the wetland releases some heat. The



**FIGURE 4.27** Winter conditions at a constructed HSSF wetland near Duluth, Minnesota.

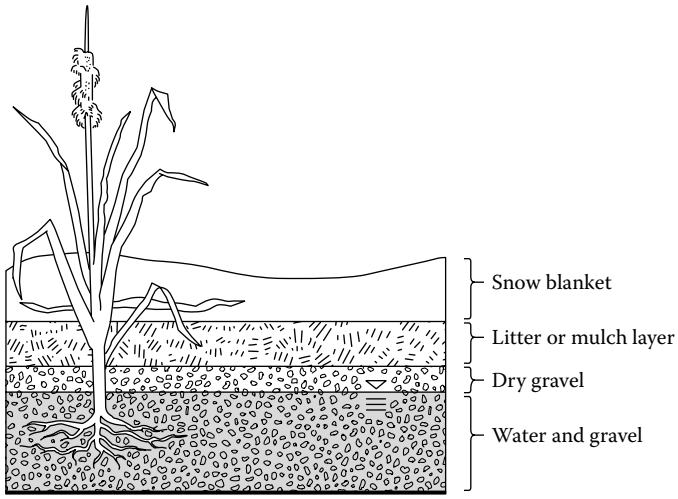



**FIGURE 4.28** Cross section of a SSF wetland in winter.

heat capacity of the HSSF wetland is estimated to be 1.32 MJ/m$^2$·°C (see Equation 4.22). An allowed temperature decrease of 2°C per 30 days can therefore absorb (2 × 1.32/30) = 0.088 MJ/m$^2$·d.

The allowable balance temperature averages 1°C (2 down to zero). Therefore:

$$\frac{(1-(-9.4))}{R} = 0.125 - (-0.088) \tag{4.50}$$

Therefore, the necessary $R$ = 49 (MJ/m$^2$·d·°C)$^{-1}$. It is comprised of several contributions, for instance (see Table 4.2 for thermal conductivities):

The expected snow depth for February for Duluth (and other locations in the United States) may be found from information at the National Climatological Data Center (NCDC), website:

http://cdo.ncdc.noaa.gov/climatenormals/clim20/

For the period 1971–2000, February had 22 days with greater than 25 cm of snow on the ground, and 27 days with greater than 12.5 cm. That much snow, of intermediate age, provides only about half the necessary insulation. The presence of a dry gravel layer is of little use. The air-side resistance is low, and also contributes little to reducing heat loss. The reader may verify that a layer of ice on top of the gravel is similarly of little or no use.

In the winter climate of Minnesota, 15 cm of mulch insulation has generally been sufficient enough to insulate the wetland bed and keep it from freezing, even during cold snaps of −45°C.

### Warm Water Influents to HSSF Wetlands

As for FWS systems, when warm water enters a HSSF wetland during freezing conditions, the incoming sensible heat can maintain an ice free zone near the inlet, for some portion of the travel distance. Data on the rate of temperature decline with distance may be used to estimate the flow length over which freezing is prevented by incoming heat. The number of HSSF wetlands for which such temperature profiles are known is small. This is because of lack of monitoring, but also because waters entering HSSF wetlands often come from cool sources such as underground transfer lines, and thus enter at temperatures comparable to the balance temperature.

The question may be posed as to whether the warm incoming water will prevent freezing, and if so, for what travel length or equivalent detention time. The empirical exponential decline is a convenient assumption:

$$T_w = T_b + (T_i - T_b)\exp\left(-\frac{t}{\tau_A}\right) \tag{4.51}$$

Calibrations of Equation 4.22 for HSSF wetlands are shown previously, in Table 4.10. An example of the temperature change with nominal detention time is shown in Figure 4.20. The mean accommodation time (63% of the change) for HSSF wetlands is 1.15 days' nominal detention. Therefore, 95% of the incoming exotherm is lost in about three days' nominal detention. Three days' detention is the 90th percentile of the distribution for HSSF wetlands in the United States (N = 65 wetland-years). The median HRT for 28 HSSF systems in New South Wales, Australia, was 8.3 days (Davison *et al.*, 2005). Czech HSSF wetlands have median nominal detention time on the order of four days. Thus, most HSSF wetlands will not be prevented from freezing by incoming warm water. The balance temperature will be controlling in the bed outlet region, which is in turn controlled by soil heat and losses to the atmosphere.

## SUMMARY

Many treatment wetlands exhibit a strong "buffer" capacity with respect to temperature due to energy flows within the wetland. Solar radiation is the driving force for evapotranspiration, which displays a strong annual cycle. In moderate temperate climates, *ET* losses are on the order of half a meter per year, but can easily be double that in hot arid climates.

Wetland exit water temperatures are approximately equal to the mean daily air temperature during unfrozen seasons, for conditions of moderate humidity and air temperature. Hot dry conditions can produce cooling, whereas humid conditions can produce heating. The equilibrium wetland water temperature represents a balance between the dominant transfers, which are incoming solar energy gains ($R_N$) and evaporative energy losses ($\rho\lambda_m ET$). The adjustment of the incoming water temperature to this balance occurs at a modest pace, with acclimation complete in about five days' detention. In the winter, the insulation provided by snow, ice, and mulch is enough to prevent water from freezing under cold climate conditions. The ice thickness is then determined by losses upward through the insulating layers as well as gains vertically upward from the earth by conduction. Energy balance equations permit calculation of wetland water temperatures.

# 5 Air, Water, and Soil Chemical Interactions

The physical and chemical environment of a wetland affects all biological processes. In turn, many wetland biological processes modify this physical/chemical environment. Four of the most widely fluctuating and important abiotic factors are dissolved oxygen (DO), oxidation-reduction potential (ORP), hydrogen ion concentration (pH), and alkalinity. Oxygen is frequently an influential factor for the growth of plants and animals in wetlands. Wetland plants have physiological adaptations that allow growth in low oxygen soils. Nitrification and oxidative consumption of organic compounds and BOD are dependent on dissolved oxygen. Wetland soils almost invariably are devoid of free oxygen, but still support a wide variety of oxidation and reduction reactions, such as ferric–ferrous iron conversion. The chemistry and biochemistry within the soil column are strongly driven by ORP. Hydrogen ion concentration, measured as pH, influences many biochemical transformations. It influences the partitioning of ionized and unionized forms of carbonates and ammonia, and controls the solubility of gases, such as ammonia, and solids, such as calcite. Hydrogen ions are active in cation exchange processes with wetland sediments and soils, and determine the extent of metal binding. Dissolved carbon dioxide, a major component of alkalinity, is the carbon source for autotrophic microbes and is the fundamental building block of wetland vegetation.

These variables may be understood by examining the normal ranges of variation in treatment wetlands. Successful design also requires that forecasts be made for intended operating conditions, which in turn implies prediction rules and equations.

It has been suggested that wetland plants are merely the substrate for microbes, which function as they would in a trickling filter. Indeed, some have suggested that the plants can be replaced by wooden or plastic dowels at the same stem density. Nothing could be further from the truth. Wetland plants are actively passing gases, both into and out of the wetland substrate. The more correct image is of a forest of chimneys, sending plumes of various gases into the atmosphere, interspersed with other plants acting as air intakes. On the diurnal cycle, the entire wetland "breathes" in and out, bringing in oxygen and discharging carbon dioxide, methane, and other gases.

## 5.1 FUNDAMENTALS OF TRANSFER

A FWS wetland provides considerable opportunity for losses of volatile compounds from the water to the atmosphere, and transfers of oxygen and carbon dioxide from the atmosphere, as does a VF system. However, HSSF wetlands have restricted ability to accomplish those transfers, because of the presence of the bed media and possibly mulch. The large areal extent, coupled with relatively long detention times and shallow water depths, are conditions that foster convective and diffusional transport to the air–water interface, upward to bulk air, and laterally off-site under the influence of winds (Figure 5.1). There is typically equilibrium between air-phase and water-phase concentrations at the interface, which separates two vertical transport zones.

Henry's law expresses the equilibrium ratio of the air-phase concentration to the water-phase concentration of a given soluble chemical. A variety of concentration measures may be used in both phases, thus generating several definitions of Henry's Law Constant ($H$). Here the water phase concentration is presumed to be given as mmol/L = mol/m$^3$, and the gas phase concentration as partial pressure in Pascals (Pa) (mole or volume fraction times total pressure). Thus:

$$P_{\text{interface}} = HC_{\text{interface}} \tag{5.1}$$

where

$C_{\text{interface}}$ = interfacial water phase concentration, mol/m$^3$
$H$ = Henry's Law Constant, atm·m$^3$/mol
$P_{\text{interface}}$ = interfacial partial pressure in air, atm

Transport in both the air and water phases may involve convective currents as well as molecular diffusion, and therefore the transport flux (flow per unit area) is commonly modeled with mass transfer coefficients (Welty *et al*., 1983):

$$J - k_w(C - C_{\text{interface}}) = k_a(P_{\text{interface}} - P) \tag{5.2}$$

where

$C$ = water phase concentration, mol/m$^3$
$J$ = loss flux, mol/m$^2$·hr
$k_a$ = air-side mass transfer coefficient, (m/hr)(mol/m$^3$)/atm = mol/(m$^2$·atm·hr)
$k_w$ = water-side mass transfer coefficient, m/hr
$P$ = partial pressure in air, atm

It is common practice to eliminate the unknown interfacial concentrations between Equations 5.1 and 5.2, yielding an

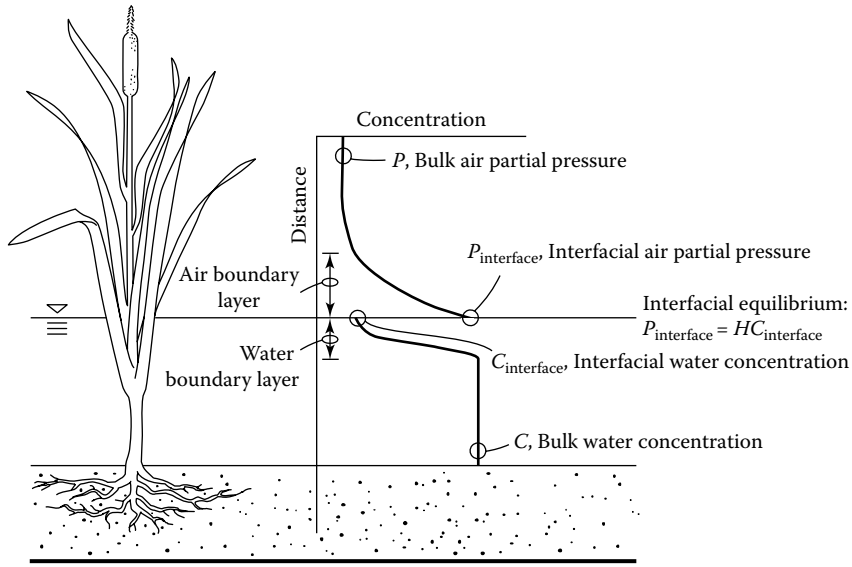



**FIGURE 5.1** A soluble volatile chemical can move from the bulk water to the air–water interface, where it equilibrates with the air-phase chemical. Movement then occurs in the air, away from the interface out to the bulk air. These routes are reversed for chemicals being taken up. Transport is typically in the turbulent range in the air, and in the laminar or transition range in the water.

expression for transfer from the bulk water to the bulk air:

$$J = K_w\left(C - \frac{P}{H}\right) \tag{5.3}$$

$$\frac{1}{K_w} = \frac{1}{k_w} + \frac{1}{Hk_a} \tag{5.4}$$

where

$K_w$ = overall water-side mass transfer coefficient, m/hr

In many instances of pollutant transfer, there is a zero bulk air concentration, and the transfer model reduces to:

$$J = K_w C \tag{5.5}$$

Air-side mass transfer coefficients are quite large, which places nearly all the mass transfer resistance on the liquid side. For instance, Mackay and Leinonen (1975) found over 80% of the transfer resistance in the water when $H > 10^{-4}$ atm·m$^3$/mol. It is again noteworthy that this theory leads to a first-order areal removal rate.

Values of $k_w$ depend upon the degree of convective mixing, as well as on the size of the molecule being transported. A large body of knowledge concerning oxygen and other gases in ponds was reviewed by Ro *et al*. (2006) and Ro and Hunt (2006). They determined a general correlation from data concerning several gases:

$$K_w = 1.706\, Sc^{-0.5} U_{10}^{1.81}\left(\frac{\rho_a}{\rho_w}\right)^{0.5} \tag{5.6}$$

where

$Sc$ = Schmidt number, = $\nu/D$, dimensionless
$D$ = diffusivity of gas, m$^2$/s
$U_{10}$ = wind speed at 10 m height, m/s
$\rho_a$ = density of air, kg/m$^3$
$\rho_w$ = density of water vapor, kg/m$^3$
$\nu$ = kinematic viscosity of gas, m$^2$/s

Experimental studies of Peng *et al*. (1995) verified the strong effect of mixing in the water phase, and established a diffusion-only value of $k_w \approx 0.03$ m/h for benzene, toluene, TCE, and PCE. In the context of treatment wetlands, these rate constants are in the range of 20–2,000 m/yr. Therefore, light molecules are very likely to be effectively stripped in wetlands that are designed to remove other constituents with equal or lower rate constants.

Plants participate in the transfer of gases to and from air, via their internal airways. For oxygen, this transfer is called the plant aeration flux, and is required to support respiration and to protect the root zone. Because any excess oxygen is available in the root zone for processes such as nitrification, further discussion of this process is to be found in Chapter 9.

## 5.2 OXYGEN DYNAMICS IN TREATMENT WETLANDS

Dissolved oxygen (DO) is of interest in treatment wetlands for two principal reasons: it is an important participant in some pollutant removal mechanisms, and it is a regulatory parameter for discharges to surface waters. In the first instance, DO is the driver for nitrification and for aerobic decomposition

of CBOD. In the second instance, DO is critical for the survival of fish and other aquatic organisms, and for the general health of receiving water bodies. In many permits in the United States, a minimum DO of 5 mg/L is specified.

Water entering the treatment wetland has carbonaceous and nitrogenous oxygen demand (NOD). After entering the wetland, several competing processes affect the concentrations of oxygen, biochemical oxygen demand (BOD), and nitrogen species. Dissolved oxygen is depleted to meet wetland oxygen requirements in four major categories: sediment/litter oxygen demand, respiration requirements, dissolved carbonaceous BOD, and dissolved NOD. The sediment oxygen demand is the result of decomposing detritus generated by carbon fixation in the wetland, as well as decomposition of accumulated organic solids which entered with the water. The NOD is exerted primarily by ammonium nitrogen; but ammonium may be supplemented by the mineralization of dissolved organic nitrogen. Decomposition processes in the wetland also contribute to NOD and BOD. Microorganisms, primarily attached to solid, emersed surfaces, mediate the reactions between DO and the oxygen consuming chemicals. Plants and animals within the wetland require oxygen for respiration. In the aquatic environment, this effect is seen as the nighttime disappearance of dissolved oxygen. Oxygen transfers from air, and generation within the wetland, supplements any residual DO that may have been present in the incoming water. Three routes have been documented for transfer from air: direct mass transfer to the water surface, convective transport down dead stems and leaves, and convective transport down live stems and leaves. The latter two combine to form the plant aeration flux, (PAF). These transfers are largely balanced by root respiration, but may contribute to other oxidative processes in the root zone.

Despite this complexity, wetlands are not particularly efficient at obtaining oxygen in sufficient quantities to deal with heavy pollutant loads. Therefore, several techniques have been employed to supplement the natural aeration processes. Compressed air bubblers, alternating fill and draw, and intermittent vertical flow have all been successfully implemented. These systems are described in more detail in Part II of this book; in this section the focus is upon passive treatment wetlands.

## Biochemical Production of Oxygen

Oxygen is the byproduct of photosynthesis (Equation 5.7). When photosynthesis takes place below the water surface, as in the case of periphyton and plankton, oxygen is added to the water internally. A large algal bloom can raise oxygen levels to 15–20 mg/L, more than double the saturation solubility, as a result of wastewater addition (Schwegler, 1978). This process requires sunlight, and algal photosynthesis is suppressed in wetlands with dense covers of emergent macrophytes.

$$6CO_2 + 12H_2O + \text{light} \rightarrow C_6H_{12}O_6 + 6O_2 + 6H_2O \quad (5.7)$$

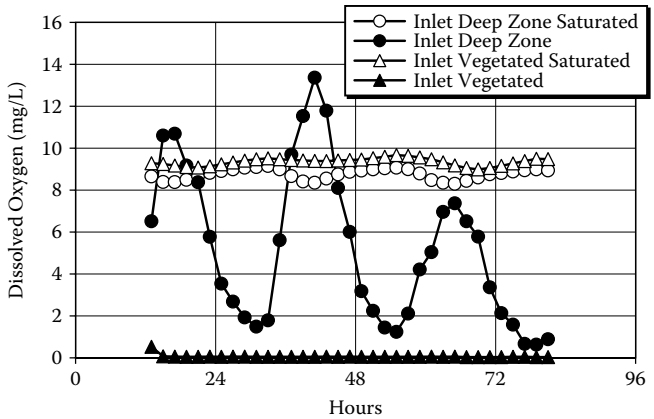



**FIGURE 5.2** Diurnal cycles in dissolved oxygen in Cell 7 of the Sacramento, California, FWS treatment wetland project, May 28–31, 1996. The inlet deep zone exceeds saturation in late afternoon. Just 46 meters downstream, in a dense community of cattails and bulrush, there is essentially no dissolved oxygen, despite a slightly higher saturation value (the water has cooled slightly). (Data from Nolte and Associates (1998a) *Sacramento Regional Wastewater Treatment Plant Demonstration Wetlands Project.* 1997 Annual Report to Sacramento Regional County Sanitation District, Nolte and Associates.)

Nonshaded aquatic microenvironments within the wetland therefore display a large diurnal swing in dissolved oxygen due to the photosynthesis–respiration cycle. Nutrients stimulate the algal community, and increase the DO mean and amplitude. When large amounts of nutrients are added to the wetland, and water depths are shallow enough for emergent rooted plants, other components of the carbon cycle are increased, such as photosynthesis by macrophytes. It is then possible for other wetland processes to become dominant in the control of dissolved oxygen. The effect is typically a depression of average DO, and a decrease in the amplitude of the diurnal cycle (Figure 5.2). This suppression of the diurnal DO cycle is a characteristic of all treatment wetlands receiving moderate to high loads of carbonaceous and nitrogenous oxygen demand.

In wetlands dominated by macrophytes, oxygen processing is more complicated. Macrophytes and periphyton contribute to respiration and photosynthesis. The decomposition of litter and microdetritus returns ammonium nitrogen and BOD to the water and to the root zone. Oxygen transfer to the root zone occurs through plants as well as from mass transfer. BOD can also degrade via anaerobic processes in the wetland litter and soil horizons.

## Physical Oxygen Transfers

The concentration of dissolved oxygen (DO) in water varies with temperature, dissolved salts, and biological activity. The effect of temperature on the equilibrium solubility of oxygen in pure water exposed to air has been widely studied, and can be calculated from regression presented in Equation 5.8

(Elmore and Hayes, 1960):

$$C_{DO}^{sat} = 14.652 - 0.41022\,T + 0.007991\,T^2 - 0.00007777\,T^3 \quad (5.8)$$

where

$C_{DO}^{sat}$ = equilibrium DO concentration at 1.0 atmosphere, mg/L
$T$ = water temperature, °C

This relation shows that at 25°C, the equilibrium DO = 8.2 mg/L, while at 5°C, the equilibrium DO = 12.8 mg/L.

There are few studies of reaeration in wetlands, and therefore the rate of oxygen supply from the atmosphere can only be estimated. Here, the methods of quantification from stream reaeration are adopted. The applicable mass transfer equation is presented in Equation 5.9:

$$J_{O_2} = K_L \left( C_{DO}^{sat} - C_{DO} \right) \quad (5.9)$$

where

$C_{DO}^{sat}$ = saturation DO concentration at water surface, mg/L = g/m$^3$
$C_{DO}$ = DO concentration in the bulk of the water, mg/L = g/m$^3$
$K_L$ = mass transfer coefficient, m/d
$J_{O_2}$ = oxygen flux from air to water, g/m$^2$·d

The parameter $K_L$ has been the subject of dozens of research studies in lakes and streams, and in shallow laboratory flume studies (U.S. EPA, 1985b). Four factors are important in determination of $K_L$: the velocity and depth of the water, the speed of the wind, and rainfall intensity.

The first two factors are typically dominant in streams and rivers, in which flow is turbulent. Accordingly, several equations in the literature are based on turbulent flow conditions, which typically do not prevail in FWS wetlands (see Chapter 2). Leu *et al*. (1997) have examined six such formulations, including the popular O'Connor and Dobbins (1958) correlation, in the context of data in laminar flow. The O'Connor and Dobbins (1958) correlation was found to greatly overpredict the mass transfer coefficient in low velocity situations (Leu *et al*., 1997).

More serious is the failure of many equations, including O'Connor and Dobbins (1958), to account for the extremely important effect of wind mixing. Chiu and Jirka (2003) present data from a large unvegetated mesocosm (1 m wide by 20 m long) that demonstrate an essentially direct proportionality between $K_L$ and the square of the wind speed. In a FWS environment, the presence of vegetation blocks wind mixing preferentially for low wind speeds. Belanger and Korzun (1990), working in sparse *Cladium* and moderately dense *Typha* wetlands, found no effect of wind up to about 3.2 m/s (as measured at ten meter height), followed by a direct proportionality to the excess of wind speed above that threshold. Thus for light winds, up to 3.2 m/s, $K_L = 0.2$ m/d, whereas $K_L$ increased dramatically to ten times that value at wind of 5.5 m/s. The presence of sparse emergent macrophytes therefore does not block physical oxygen transfer.

Low values of $K_L$ in wetlands are due in large measure to low flow rates, and the attendant low degree of water mixing. In addition to the effect of wind, rain also creates surficial mixing and increases the mass transfer coefficient. Belanger and Korzun (1990) measured a linear dependence of $K_L$ on rainfall intensity, with $K_L$ = 1.2 m/d at a rainfall rate of 5 mm/h. Thermal convection, operating on a diurnal cycle, has also been implicated in oxygen transfer in treatment wetlands (Schmid *et al*., 2005a).

## Open Water Zones

Treatment wetlands are sometimes configured with open water zones, which would seem to offer enhanced opportunity for oxygen transfer. Despite the considerable uncertainty in the mass transfer coefficient, calculations show that physical reaeration is a slow process, even under moderate windiness. For instance, in the absence of any other processes, the forecast of the detention time to bring water from zero DO to 90% of saturation is in the range of two to four days for typical wind velocities.

Bavor *et al*. (1988) operated an open water, unvegetated wetland receiving secondary effluent. This system maintained high DO, ranging from 4.3 to 14.6 mg/L over the seasons. The values of $K_L$ calculated from Bavor's open water system were 0.2–1.0 m/d under some conditions. But oxygen levels frequently exceeded saturation, indicating internal generation of oxygen, most likely by algae. Suspended solids were quite high in the effluent, 24–147 mg/L.

An open water, unvegetated wetland was monitored for DO in Commerce Township, Michigan, for a period of three years. Ammonia and BOD were very low in this "polishing" wetland, typically less than 0.2 mg/L for ammonia and less than 2.0 mg/L for $CBOD_5$. Inlet DO averaged 83% of saturation, and outlet DO was 91% of saturation after 3.3 days' detention. The corresponding mean $K_L$ value was 0.42 m/d (R.H. Kadlec, unpublished data).

The Tres Rios, Arizona, wetland H1 contained 20% deep zones (1.5 m) in seven sections, with 80% at a depth of 0.3 m. The deep zones were predominantly open water, with only occasional *Lemna* cover and sparse SAV. The incoming wastewater contained essentially no $CBOD_5$ (2.3 mg/L) and little ammonia (1.57 mg/L) during a three-year period in which DO profiles were measured. The mean detention time was 5.6 days. Wastewater entered at low DO, and was not oxygenated during transit (Figure 5.3). Thus, it appears that atmospheric reaeration of open water occurs only to a limited extent. No existing correlation for $K_L$ can be recommended, because none have been developed for wetland conditions. As a preliminary estimate for FWS wetlands, $0.1 < K_L < 0.4$ m/d (R.H. Kadlec, unpublished data).

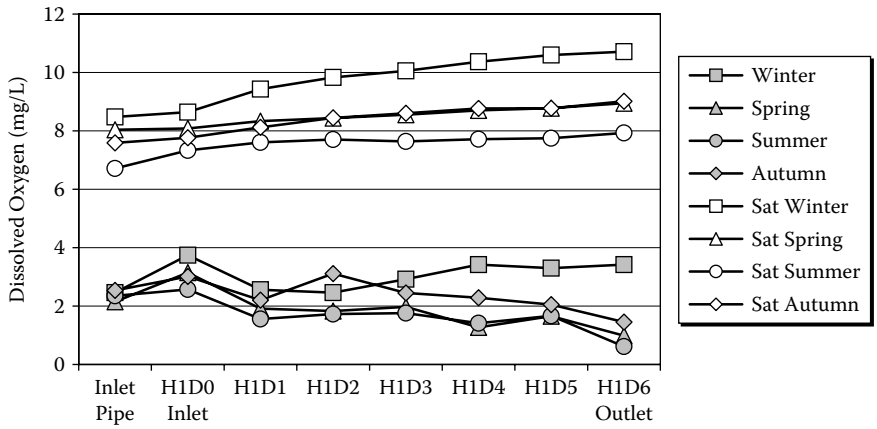



**FIGURE 5.3** Dissolved oxygen profiles along the flow path through Hayfield Cell 1 at the Tres Rios, Arizona, site. Seasonal averages of monthly data collected over three years. The sampling points were located in deep zones located at even spacing from inlet to outlet. The detention time was 5.6 days, at a depth of 30 cm in the bench areas.

## Plant Oxygen Transfer

### Emergent Plant Oxygen Transfer

Great care must be exercised in the interpretation of the literature concerning oxygen transfer by plants in wetlands. Although it is certain that oxygen transfer does occur at modest rates, the amount that is transferred in excess of plant respiration requirements is much less certain. Further, methods of measurement have been variable, and some are purely presumptive. One group of estimates relies upon measurements for individual plants or roots, commonly in hydroponic environments, and extrapolation via root dimensions and numbers. For example, Lawson (1985) calculated a possible oxygen flux from roots of *Phragmites australis* up to 4.3 g/m$^2$·d, and Armstrong *et al*. (1990) calculated 5–12 g/m$^2$·d. Gries *et al*. (1990) calculated 1–2 g/m$^2$·d. It is apparent that the oxygen demand in the root environment is an important determinant of how much oxygen is supplied to that root zone, with high demands increasing the supply, up to a limit (Sorrell, 1999). Hydroponic systems react much differently to flow through than to batch conditions (Sorrell and Armstrong, 1994). Furthermore, plants growing in anoxic conditions can modify their root structure, creating fewer small roots and more large roots, presumably as a defense against the large oxygen supplies demanded by the small roots (Sorrell *et al*., 2000). Nonetheless, such hydroponic experiments serve to elucidate the effects of variables. For example, Wu *et al*. (2000) used hydroponic experiments to estimate 0.04 g/m$^2$·d supplied by *Typha latifolia*, versus 0.60 g/m$^2$·d supplied by *Spartina pectinata*.

A second group of estimates relies upon the disappearance of CBOD and ammonia to infer an oxygen supply. Differences between side-by-side systems are then used to infer the amount of the inferred supply that came from plants. This procedure also has considerable uncertainty, because it is founded on the presumption of oxygen consumption being due to oxidative processes for ammonia and CBOD, and to specific stoichiometric relations. That presumed chemistry is in question, because of alternative loss and gain mechanisms for both ammonia and CBOD. Cooper (1999) labels the estimation of oxygen supply from ammonia and CBOD loss "a crude calculation." Consequently, such determinations are here termed "implied oxygen supply" rates. However, a number of authors have reported such implied oxygen supply (Platzer, 1999; Wu *et al*., 2000; Crites *et al*., 2006). Again, this estimate may be better used as a comparative, with reference to side-by-side studies of vegetated and unvegetated systems.

The third group of studies relies upon direct measurements of oxygen uptake. This may be done in the field (e.g., Brix, 1990), or more readily in laboratory mesocosms (e.g., Wu *et al*., 2001). Brix (1990) and Brix and Schierup (1990) cast doubts upon the importance of oxygen release from plants, and more recent studies have confirmed this lack of importance. For instance, Townley (1996) found essentially no oxygen released by *Schoenoplectus* (*Scirpus*) *validus* or *Pontederia cordata*. Wu *et al*. (2001) measured 0.023 g/m$^2$·d transferred by *Typha* in mesocosms. Bezbaruah and Zhang (2004; 2005) used direct measurement techniques to study the effects of BOD on oxygen transfer by *Scirpus validus*, and found only 1–4 mg/m$^2$·d released at BOD = 76 mg/L, and 11 mg/m$^2$·d released at BOD = 1,267 mg/L. This direct measurement evidence strongly suggests that emergent plants do not contribute "extra" oxygen transfer to any appreciable degree, although they do send oxygen to the root zone to protect themselves and conduct respiration. More information on oxygen transfer is presented in Chapter 9, in the context of nitrification.

### Floating Plants

Open water zones, in the presence of elevated nutrient supplies, may be colonized by floating plants, such as *Lemna* spp., *Hydrocotyle umbellata*, and *Azolla* spp. These form a physical cover that is a barrier to oxygen transfer. Additionally, wind can cause the formation of very thick mats by drifting and

compression. Root oxygen release rates from a number of free-floating plants in batch hydroponic laboratory studies were calculated in the range of 0.26–0.96 g/m²·d (Moorhead and Reddy, 1988; Perdomo *et al*., 1996; Soda *et al*., 2007).

As an example, the Sacramento, California, wetlands were configured with 19% of the area without emergent plants, due to design water depths of 1.5 m (Nolte and Associates, 1997). Most of the deep zones became covered with *Lemna* spp. On some occasions, DO concentrations increased in these deep zones, but on average there was little increase in DO. The ammonia loading was high, with concentrations in the range of 10–20 mg N/L. There was no discernible increase in the ammonia removal rates in the deep zones.

### Submerged Plant Oxygen Transfer

Submerged aquatic vegetation (SAV), including algae, photosynthesizes within the water column, and therefore contribute oxygen directly to the water. This activity is driven by sunlight, leading to very strong diurnal cycles in the resultant DO content of the water column. The magnitude of DO enhancement can be large, especially in lightly loaded wetlands. Root oxygen release rates from a number of submerged plants in natural environments are reported to be in the range of 0.5 to 5.2 g/m²·d (Sand-Jensen *et al*., 1982; Kemp and Murray, 1986; Caffrey and Kemp, 1991). More recent work by Laskov *et al*. (2006) shows a calculated range of 0.15–0.60 g/m²·d based on 200 plants per square meter.

Attempts to relate the effect of oxygen transfer to ammonia removal, via the presumptive enhancement of added DO, are less than clear. For instance, the data of Toet (2003) details the performance of *Phragmites* and *Typha* in the first half of a FWS wetland, followed by submerged vegetation dominated by *Elodea nuttallii*, *Potamogeton* spp., and *Ceratophyllum demersum*. Eight wetlands plus an unvegetated control were studied for a calendar year, two years after startup. Organic loadings were very low, and ammonia was typically in the range 0.4 to 0.7 mg N/L. The emergent sections of the wetlands lowered the already-low DO from the pretreatment plant. The submergent sections raised the DO to 4–18 mg/L. However, ammonia removal rates were found to be lower in the submerged sections than in the emergent sections, with mass removal efficiencies more than two times lower (33% versus 12%).

DB Environmental (DBE, 2002) operated SAV mesocosms and 0.2 ha SAV wetlands during 1999–2002. Dissolved oxygen was found to be at or above saturation during the day in the surface water layer, but was very much lower at night and in bottom water layers.

Knight *et al*. (2003) reported the performance of 13 flow through Florida water bodies dominated by SAV. Of these, seven were in the depth range (1.1–2.2 m) and the detention time range (2–20 days) of interest for treatment wetlands. Incoming ammonia levels were low (0.03–0.20 mg/L), as were TKN levels (0.1–2.8 mg/L). These large systems (147–2,452 ha) removed no ammonia, and further did not alter TKN. Therefore, the implied oxygen supply was zero, thus casting more doubts on the use of ammonia removal as an indicator of oxygen supply in the SAV environment.

U.S. EPA (1999) shows high oxygen concentrations for the surface layer of the SAV sections of FWS wetlands operating in Arcata, California. However, the vegetative cover was not stable, changing from SAV to *Lemna* on a seasonal basis (U.S. EPA, 1999). U.S. EPA (2000a) hypothesizes the necessity for including a SAV zone in FWS design for ammonia removal, based upon presumptive reoxygenation. However, they state that " … quantitative estimates of transfer are difficult to assess based on current data."

## Biological and Chemical Oxygen Consumption

### Longitudinal Gradients

When wastewater with BOD and ammonia nitrogen is discharged to rivers and streams, an oxygen sag analysis is often applied (Metcalf and Eddy Inc., 1991). This Streeter–Phelps (1925) analysis is predicated on the assumption that oxygen is increased in the flow direction by mass transfer from the air above, and by photosynthesis occurring within the water column, and decreased by consumption of BOD and ammonium nitrogen oxidation, and decreased by consumption of Sediment Oxygen Demand (SOD) and respiration. In the wetland environment, both sediments and litter consume oxygen during decomposition. Decomposition processes also release carbon and nitrogen compounds to the overlying water, which can exert an oxygen demand. It is therefore apropos to designate the sum as Decomposition Oxygen Demand (DOD). Plants transfer oxygen to their root zone to satisfy respiratory requirements, and may in some instances transfer a surplus to control the oxygen environment around the roots. The balance on DO in the wetland from the inlet (0) to a specified distance ($L$) along the flow path can be written as (Equation 5.10):

$$\begin{aligned} q\left[C_{\mathrm{DO}}(L)-C_{\mathrm{DO}}(0)\right] &= K_L\left(C_{\mathrm{DO}}^{\mathrm{sat}}-\overline{C_{\mathrm{DO}}}\right) \\ &+\left[r_{\mathrm{O,\,photo}}-r_{\mathrm{O,\,res}}-r_{\mathrm{O,\,DOD}}\right]+a_{\mathrm{N}}q\left[C_{\mathrm{N}}(L)-C_{\mathrm{N}}(0)\right] \\ &+a_{\mathrm{B}}q\left[C_{\mathrm{BOD}}(L)-C_{\mathrm{BOD}}(0)\right] \end{aligned} \quad (5.10)$$

where

$a_{\mathrm{N}}$ = stoichiometric coefficient for $NH_4$-N oxygen demand
$a_{\mathrm{B}}$ = stoichiometric coefficient for BOD oxygen demand
$\overline{C_{\mathrm{DO}}}$ = average DO concentration average over length L, g/m³ = mg/L
$C_{\mathrm{BOD}}$ = BOD concentration, g/m³ = mg/L
$C_{\mathrm{N}}$ = ammonia nitrogen concentration, g/m³ = mg/L
$q$ = hydraulic loading rate, m/d
$r_{\mathrm{O,\,photo}}$ = rate of DO generation by photosynthesis, g/m²·d
$r_{\mathrm{O,\,res}}$ = rate of DO consumption by respiration, g/m²·d
$r_{\mathrm{O,\,DOD}}$ = rate of DO consumption by decomposition, g/m²·d

There is no treatment wetland data with which to separately evaluate photosynthesis, respiration, plant aeration flux (PAF), and decomposition oxygen demand (DOD). It is necessary to lump these into Wetland Oxygen Demand (WOD) (Equation 5.11):

$$r_{O,\,WOD} = r_{O,\,DOD} + r_{O,\,res} - r_{O,\,photo} \tag{5.11}$$

where

$r_{O,\,WOD}$ = net wetland oxygen consumption rate, g/m$^2$·d

Further, there is often no data from which to estimate the reaeration coefficient $K_L$. Therefore, all transfer rates to and from the atmosphere and to and from the biomass in the wetland are lumped into a single term, the wetland net oxygen supply rate (Equation 5.12).

$$r_{NOSR} = K_L\left(C_{DO}^{sat} - \overline{C_{DO}}\right) - r_{WOD} \tag{5.12}$$

where

$r_{NOSR}$ = net oxygen supply rate, g/m$^2$ · d

## Wetland Profiles

Example profiles in dissolved oxygen are shown in Figure 5.3 for a low DO influent to a FWS system in a warm climate (Tres Rios, Arizona). There are not large increases in DO (due to reaeration), nor large decreases (due to WOD). A similar situation prevails for HSSF wetlands, as illustrated in Figure 5.4 (NERCC, Minnesota). These profiles do not resemble the "oxygen sag" profiles of streams subjected to point sources of oxygen demand.

The net oxygen supply rate can be positive (supply), negative (consumption), or zero. The data of Stengel *et al.* (1987) provide values of net oxygen consumption rates for *Phragmites* gravel bed wetlands. Fully oxygenated tap water with zero BOD and zero TKN was fed to the wetland, and the DO was found to decrease with distance in the inlet region. The SSF wetland was thus consuming oxygen in the absence of incoming BOD or NOD, with strong seasonal variations (Figure 5.5).

The interpretation of the data presented in Figure 5.5 is simply that WOD exceeded the transfer of oxygen from air; and DO was depleted. Photosynthetic production of $O_2$ was likely zero in the gravel bed, and no mass transfer would be expected at the inlet, because the water was saturated with DO. Consequently, the rates shown in Figure 5.5 correspond to $r_{O,\,WOD}$ (see Equation 5.11).

Stengel (1993) also found that after the initial drop in DO, reaeration did not occur; rather, DO reached a stable (constant) value with increasing distance along the bed. Cattails provided a stable root zone DO of about 1–2 mg/L in summer, whereas *Phragmites* stabilized at essentially zero DO. The implication is that in the downstream portions of the wetland, all oxygen uptake was consumed by respiration and SOD. It is important to note that this zero-loaded HSSF wetland was not able to sustain a high oxygen concentration in the water: the internal wetland processes consumed all transferred oxygen.

The stoichiometric coefficients in Equation 5.10 are often taken to be $a_B = 1.5$ and $a_N = 4.5$. However, wetland data sets are not consistent with that presumption (Kadlec and Knight, 1996). When Equation 5.4 was regressed for wetlands with DO, BOD, and $NH_4$-N information, the stoichiometric coefficients were very much smaller. The inference is that biomass compartments participate in dictating the oxygen level.

It is concluded that the Streeter–Phelps analysis is not suitable for wetlands, due to lack of the ability to quantify wetland oxygen demand (WOD), which is a more dominant factor in wetlands than in streams. It is therefore instructive to summarize some operational results instead. Table 5.1 lists several annual average inlet and outlet DO values for treatment wetlands, together with the associated BOD and ammonia concentrations. It is clear from these examples that HSSF

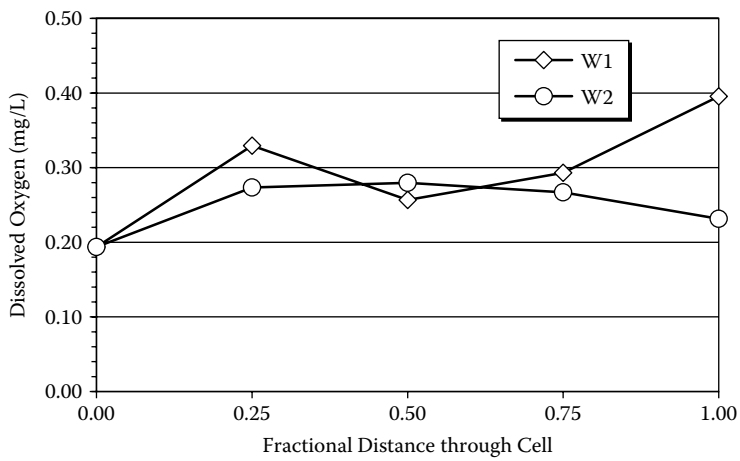



**FIGURE 5.4** Dissolved oxygen profiles for the NERCC, Minnesota, HSSF wetlands (W1 and W2). There is essentially no DO in the incoming water, and none along the flow direction including the outlet. There are 31 measurement occasions over two years.

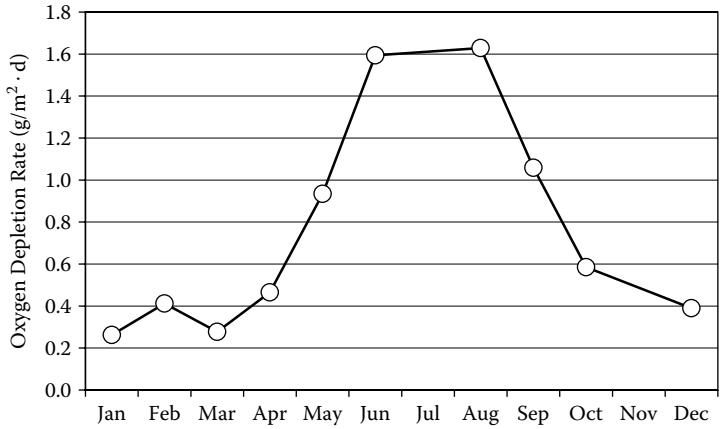



**FIGURE 5.5** Oxygen depletion rate in the inlet zone of a *Phragmites* gravel bed wetland receiving oxygenated tap water with nitrate at 30 ± 2 mg/L. (Data from Stengel *et al*. (1987) In *Aquatic Plants for Water Treatment and Resource Recovery*. Reddy and Smith (Eds.), Magnolia Publishing, Orlando, Florida, pp. 543–550.)

## TABLE 5.1
## Dissolved Oxygen Entering and Leaving Treatment Wetlands

| Wetland System | HLR (cm/d) | Inlet BOD (mg/L) | Outlet BOD (mg/L) | Inlet $NH_3$-N (mg/L) | Outlet $NH_3$-N (mg/L) | Inlet DO (mg/L) | Outlet DO (mg/L) |
|---|---|---|---|---|---|---|---|
| **Free Water Surface** | | | | | | | |
| Hillsdale, Michigan | 0.8 | ≈ 0 | ≈ 0 | 0.01 | 0.04 | 9.13 | 8.82 |
| Commerce Twp., Michigan | 18.2 | 1.18 | 2.32 | 0.064 | 0.050 | 8.32 | 9.86 |
| Orlando Easterly, Florida | 4.9 | 1.95 | 1.02 | 0.33 | 0.09 | 6.10 | 2.62 |
| Tres Rios, Arizona | 10.9 | 2.26 | 1.53 | 1.69 | 0.75 | 6.10 | 2.62 |
| Listowel 3, Ontario | 1.3 | 19.4 | 7.3 | 7.04 | 3.43 | 5.65 | 3.48 |
| Augusta, Georgia | 7.3 | 10.47 | 4.71 | 2.51 | 2.15 | 4.83 | 7.21 |
| Sacramento, California | 6.5 | 23.9 | 6.5 | 15.4 | 10.4 | 3.28 | 2.99 |
| Listowel 4, Ontario | 1.8 | 55.7 | 9.5 | 8.80 | 6.98 | 2.13 | 2.71 |
| Richmond, New South Wales Open Water | 6.4 | 51.7 | 22.9 | 35.2 | 17.5 | 1.01 | 8.50 |
| Pontotoc 2, Mississippi | 1.54 | 46.5 | 26.5 | 112 | 39 | 3.57 | 5.94 |
| Portland, New Zealand | 5.2 | 33 | 10 | 1.7 | 4.9 | 11.2 | 5.3 |
| Oregon State 2 | 3.95 | 1003 | 291 | 168 | 88 | 2.39 | 0.09 |
| **Subsurface Flow** | | | | | | | |
| Benton, Kentucky #3 | 7.1 | 25.6 | 6.2 | 4.8 | 8.6 | 8.20 | 1.00 |
| Richmond, New South Wales Bulrush | 5.1 | 51.7 | 5.8 | 35.2 | 19.5 | 1.01 | 0.00 |
| Richmond, New South Wales Cattail | 4.6 | 51.7 | 4.7 | 35.2 | 18.8 | 1.01 | 0.04 |
| Richmond, New South Wales Gravel | 3.8 | 51.7 | 4.3 | 35.2 | 19.2 | 1.01 | 0.25 |
| Hardin, Kentucky #2 | 4.9 | 32.1 | 4.6 | 3.4 | 3.2 | 3.04 | 0.60 |
| Minoa, New York | 14 | 149 | 44 | 23.2 | 20.6 | 4.21 | 0.03 |
| Grand Lake, Minnesota | 1.02 | 184 | 69 | 51.2 | 24.5 | 0.10 | 0.30 |
| NERCC, Minnesota | 1.36 | 256 | 36 | 73.5 | 50.2 | 0.17 | 0.33 |
| Břehov, Czech Republic | 2.6 | 109 | 27 | 40 | 24.7 | 1.4 | 4.0 |
| Ondřejov, Czech Republic | 7.5 | 104 | 12 | 18.3 | 25.5 | 5.5 | 4.9 |
| Čistá, Czech Republic | 17.4 | 37 | 7.3 | 14.1 | 12.8 | 4.9 | 3.7 |
| Dušníky, Czech Republic | 1.8 | 716 | 56 | 54 | 27 | 0.9 | 2.0 |
| Mořina, Czech Republic | 2.8 | 116 | 27 | 35.4 | 32.3 | 1.5 | 0.2 |
| Rector, Arkansas | 7.6 | 45 | 27 | 0.7 | 4.4 | 5.7 | 0.7 |
| Smackover, Arkansas | 19.4 | 19 | 16 | 3.5 | 2.2 | 4.1 | 0.3 |
| Waldo, Arkansas | 20.2 | 28 | 14 | 2.0 | 3.5 | 10.2 | 0.2 |
| Waipoua HQ, New Zealand | 0.4 | 63 | 11 | 47.3 | 35.7 | 1.1 | 2.9 |

*Note:* Oxygen consumption is to some extent related to the differences between inlet and outlet BOD and ammonia. Subsurface systems are more heavily loaded with BOD and NOD, and have essentially no DO in their effluents.

wetlands in North America commonly do not have any substantial amount of DO in their effluents. Additionally, the intensive studies at the Tennessee Tech site, with 14 HSSF wetlands, in Baxter, Tennessee, found DO essentially at or below the detection limit over a two-year period (George *et al*., 1998).

However, Vymazal and Kröpfelová (2006) reported substantial concentrations of DO at the outflow of many Czech HSSF systems. Out of 59 HSSF wetlands surveyed, they found 33 with outflow DO less than 3 mg/L, and 18 with DO greater than 5 mg/L. The HSSF wetlands receiving dairy wastewater in New Zealand, with high CBOD and ammonia in the inlet, produced moderate DO, in the range of 3–5 mg/L (Tanner *et al*., 1995a; Tanner *et al*., 1998b). According to the oxygen mass balance (Equation 5.12), there should be no DO in HSSF wetland discharges when treating wastewaters with high oxygen demands. Vymazal and Kröpfelová (2006) suggested that outflow DO concentration is a very poor indicator of processes occurring in the SSF wetlands, but the reverse appears to be important as well: Reduction of CBOD and ammonia are not good indicators of the outlet DO.

There are a number of potential reasons for unexpectedly high DO in some HSSF effluents. Reaeration in outlet structures may occur due to splash and exposure to air. The membrane electrode measurement is often used, and is subject to interferences from hydrogen sulfide and from dissolved salts. Preferential flow paths in the wetland, including the possibility of overland flow, can lead to effluents that are not representative of the water within the gravel matrix.

The situation for FWS wetlands is also not clear. Some lightly loaded systems have a great deal of DO (Commerce Township, Michigan), while others do not (Orlando, Florida Easterly; Tres Rios, Arizona). Some with moderate loading reaerate to a large extent (Richmond, New South Wales Open Water; Pontotoc, Mississippi).

It is of interest to compare the open water and gravel systems at Richmond, New South Wales. These had the same geometry, received the same influent water, and both were devoid of macrophytes. BOD and ammonia were reduced in both (Table 5.1). The open water system had fully aerated water at the outlet, whereas the gravel bed effluent was very low in DO. The conclusion may be drawn that the presence of gravel interfered with oxygen transfer.

## The Sediment–Water Interface

Dissolved oxygen uptake at a sediment–water interface (SOD) is controlled by mass transport and/or biochemical reactions in two adjacent boundary layers: the diffusive boundary layer in the water and the penetration in the sediment (Higashino *et al*., 2004). Those boundary layers are very thin, with dimensions measured in millimeters (Crumpton and Phipps, 1992). As a result of the slow rate of oxygen transport through interstitial water and a comparatively high oxygen demand, the surface oxidized soil or sediment horizon is thin and ranges from a few millimeters to a few centimeters in depth, depending on the oxygen consumption capacity of the material. Though this oxidized surface horizon is thin, biological and chemical processes occurring in this zone strongly influence the availability of both nutrients and toxins in flooded soils and sediment–water interface (Gambrell and Patrick, 1978).

Under FWS wetland conditions, there is a strong dependence of SOD exertion on velocity, and transport through the diffusive boundary layer is limiting.

## Vertical Stratification

Vertical dissolved oxygen profiles have not been extensively studied in treatment wetlands. However, results from three types of systems help provide insights: ponds, wetlands with submerged aquatic vegetation (SAV), and HSSF wetlands. All three of these variants of treatment wetlands exhibit vertical stratification with respect to oxygen.

Pond studies have shown some variable but strong vertical gradients over the top 25 cm of the water column (Abis, 2002). Because concentrations often exceed saturation in the top pond water layer, algal photosynthetic reaeration is present. The high values of DO at the water surface are caused by the preferential interception of photosynthetically active radiation (PAR) in the upper water layers.

Given that physical transfer occurs from the atmosphere, and biochemical generation can occur within the water column, vertical profiles of DO are anticipated in FWS wetlands, and in fact are found in the field. Extensive measurements were made in the lightly loaded treatment wetlands of the Everglades, Florida, Nutrient Removal Project (Chimney *et al*., 2006) (Figure 5.6). The highest DO values were found in the open water and submerged vegetation zones, with a strong decreasing gradient with depth. In contrast, DO values in areas of floating plants and emergent vegetation were low, only 1–2 mg/L on average. FWS wetlands with submerged aquatic vegetation display strong vertical profiles of DO (Table 5.2). This is presumably also due to photosynthetic reaeration, with the submerged macrophytes proving oxygen, rather than algae. As in algal ponds, the upper water zones are preferentially active.

Vertical profiles of DO in HSSF wetlands are also present, but with much lesser values and smaller gradients (Table 5.2). HSSF wetlands typically have very little or no dissolved oxygen anywhere in the water column (Table 5.2). Neither algae nor SAV are present to contribute to photosynthetic reaeration. Physical reaeration can and does occur, but transfer rates are lessened by the presence of the gravel media, which precludes wind enhancement and lengthens diffusion distances. As a consequence, oxidation-reduction potential ($E_h$) (see the following section of this chapter) becomes a more effective measure of conditions within the bed. Nominally, negative $E_h$ values correspond to the absence of DO, and provide conditions conducive to reduction of nitrate, iron, and sulfate (Reddy and D'Angelo, 1994). For HSSF wetlands, physical reaeration from the top represents the dominant mechanism. Comparison of planted and unplanted beds shows that there is essentially no effect of vegetation, with the vegetated systems at Minoa, New York, and Vilagrassa, Spain, showing slightly lower $E_h$ than the unvegetated systems.

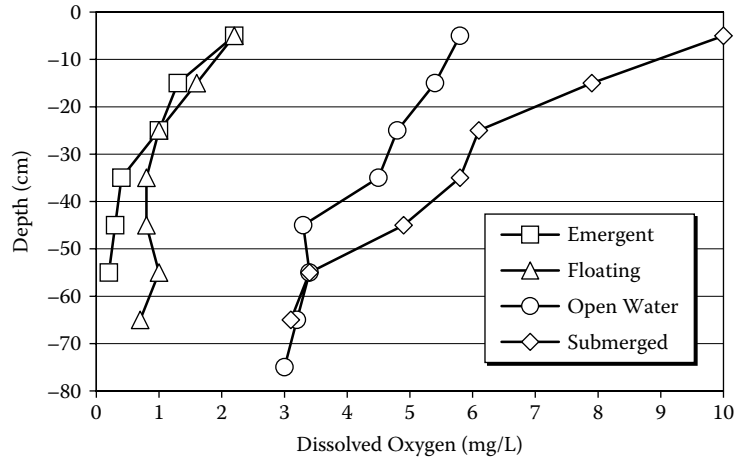



**FIGURE 5.6** Vertical profiles of dissolved oxygen in the various vegetation types in the Everglades Nutrient Removal Project FWS wetlands, Florida. Data are from 141 profiles collected over a 2.5-year period. (Data from Chimney *et al*. (2006) *Ecological Engineering* 27(4): 322–330.)

## TABLE 5.2
## Vertical $E_h$ and DO Profiles in Treatment Wetlands

| HSSF System | | Bed Depth (cm) | Bottom (cm) | Mid (cm) | Mid (cm) | Top (cm) |
|---|---|---|---|---|---|---|
| Grand Lake | | 60 | 53 | — | 23 | 8 |
| | DO | mg/L | 0.24 | — | 17.9 | 0.49 |
| NERCC 1 | | 45 | 40 | — | 23 | — |
| | DO | mg/L | 0.11 | — | 0.16 | — |
| NERCC 2 | | 45 | 40 | — | 23 | — |
| | DO | mg/L | 0.08 | — | 0.13 | — |
| Minoa Planted | | 84 | 70 | — | 40 | 10 |
| | DO | mg/L | 0.02 | — | 0.06 | 0.47 |
| | $E_h$ | mv | −243 | — | −229 | −192 |
| Minoa Unplanted | | 84 | 70 | — | 40 | 10 |
| | DO | mg/L | 0.04 | — | 0.03 | 0.20 |
| | $E_h$ | mv | −238 | — | −218 | −194 |
| Vilagrassa Planted | | 70 | 30 | 20 | 10 | 0 |
| | Inlet $E_h$ | mv | −115 | −120 | −70 | — |
| | Outlet $E_h$ | mv | −25 | −10 | 70 | 160 |
| Vilagrassa Unplanted | | 70 | 30 | 20 | 10 | 0 |
| | Inlet $E_h$ | mv | −90 | −80 | −55 | 60 |
| | Outlet $E_h$ | mv | −5 | 5 | 100 | 160 |
| **FWS SAV System** | | **Water Depth (cm)** | **Bottom (cm)** | **Mid (cm)** | **Mid (cm)** | **Top (cm)** |
| Arcata | | 100 | 90 | — | 50 | 10 |
| | DO | mg/L | 0.5 | — | 6 | 11 |
| ENR Shallow | | 40 | — | — | 30 | 3 |
| | DO | mg/L | — | — | 7.0 | 12.3 |
| ENR Medium | | 80 | — | 60 | 30 | 3 |
| | DO | mg/L | — | 3.9 | 4.2 | 13.8 |
| ENR Deep | | 120 | 90 | 60 | 30 | 3 |
| | DO | mg/L | 7.3 | 7.5 | 9.5 | 15.2 |

*Source:* For data on HSSF: for Grand Lake and NERCC, Minnesota: unpublished data; for Minoa, New York: Theis and Young (2000) *Subsurface flow wetland for wastewater treatment at Minoa*. Final Report to the New York State Energy Research and Development Authority, Albany, New York; for Vilagrassa, Spain: García *et al.* (2003a) *Ecological Engineering* 21(2–3): 129–142. For data on FWS: for Arcata, California: U.S. EPA (1999) *Free water surface wetlands for wastewater treatment: A technology assessment*. EPA 832/R-99/002, U.S. EPA Office of Water: Washington, D.C. 165 pp.; for ENR, Florida mesocosms: DBE (1999) *A demonstration of submerged aquatic vegetation/limerock treatment system technology for removal of phosphorus from Everglades agricultural area water: Final Report*. Prepared for the South Florida Water Management District (SFWMD) and the Florida Department of Environmental Protection (FDEP). Contract No. C-E10660, DB Environmental (DBE).

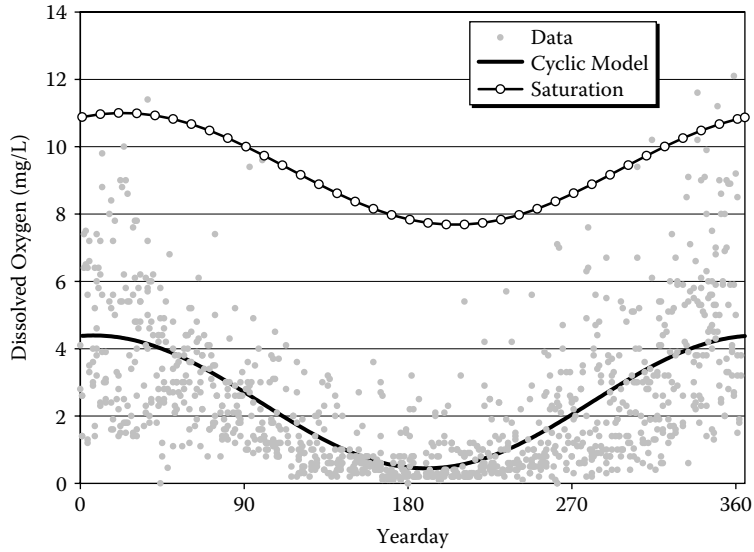



**FIGURE 5.7** Annual progression of dissolved oxygen at the Tres Rios, Arizona, FWS Hayfield wetlands. Six years' data are represented for two wetlands (H1 and H2), at an average detention time of 5.3 days.

### Trends and Variability

The annual temperature cycle in FWS systems creates a similar cycle in the saturation concentrations of dissolved oxygen, with greater solubility in the colder months. Consequently, the driving force for physical reaeration is maximum in cold months. The photosynthetic production of oxygen in the water column, by algae and/or submerged macrophytes, is driven by a seasonal cycle in solar radiation (PAR). It is therefore expected that wetland water dissolved oxygen, if any, will follow a seasonal cycle with larger values in cold months. This is indeed the case for those systems that have been studied, such as the Tres Rios, Arizona, wetlands (Figure 5.7). Equation 6.1 (see Chapter 6 for a full discussion of this equation) was fit to the DO data. The annual trend in daily values at Tres Rios had an amplitude of about 80% of the annual mean of 2.4 mg/L, with the maximum in January. Cyclic trends are similar in other FWS wetlands, with the parameters depending on location and loading (Table 5.3).

$$C = C_{\text{avg}}\left[1 + A \cdot \cos\left(\omega(t - t_{\max})\right)\right] + E \qquad (6.1)$$

The values of $E$ in Equation 6.1 follow a distribution that is nearly normal (Figure 5.8). The breadth of the scatter changes during the course of the year, with more scatter in the winter. The median amplitude of the annual cycle is 65% of the annual mean for FWS wetland outflows (Table 5.3).

**TABLE 5.3**
**Trend Multipliers for Dissolved Oxygen in FWS Wetlands**

| Wetland System | Years | Mean | Amplitude | Yearday Maximum | $R^2$ | Excursion Frequency 5% | 10% | 20% | 50% |
|---|---|---|---|---|---|---|---|---|---|
| Orlando, Florida Easterly Wetland | 10 | 2.64 | 0.41 | 21 | 0.213 | 0.44 | 0.53 | 0.74 | 1.04 |
| Hillsdale, Michigan EA | 6 | 6.92 | 0.65 | 32 | 0.370 | 0.08 | 0.17 | 0.35 | 1.00 |
| Hillsdale, Michigan ET | 6 | 6.94 | 0.65 | 32 | 0.368 | 0.08 | 0.17 | 0.35 | 1.00 |
| Hillsdale, Michigan WT | 6 | 8.65 | 0.78 | 43 | 0.509 | 1.08 | 1.17 | 1.35 | 2.00 |
| Hillsdale, Michigan WA | 6 | 8.70 | 0.61 | 44 | 0.392 | 0.05 | 0.11 | 0.59 | 0.97 |
| Tres Rios, Arizona, Hayfield 1 | 6 | 2.54 | 0.91 | 10 | 0.353 | 0.21 | 0.29 | 0.40 | 0.79 |
| Tres Rios, Arizona, Hayfield 2 | 6 | 2.29 | 0.72 | 4 | 0.356 | 0.28 | 0.35 | 0.46 | 0.81 |
| Tres Rios, Arizona, Cobble 1 | 6 | 3.29 | 0.84 | 364 | 0.280 | 0.17 | 0.20 | 0.31 | 0.65 |
| Tres Rios, Arizona, Cobble 2 | 6 | 2.77 | 0.79 | 11 | 0.278 | 0.16 | 0.23 | 0.31 | 0.65 |
| Listowel, Ontario, 3 | 4 | 3.51 | 0.63 | 360 | 0.285 | 0.06 | 0.13 | 0.26 | 0.89 |
| Musselwhite, Ontario | 4 | 5.33 | 0.65 | 42 | 0.351 | 0.50 | 0.59 | 0.71 | 0.93 |
| Titusville, Florida | 7 | 2.55 | 0.38 | 33 | 0.439 | 0.55 | 0.68 | 0.78 | 1.02 |
| ENRP, Florida | 6 | 3.7 | 0.44 | 41 | 0.418 | 0.34 | 0.52 | 0.72 | 0.99 |
| Commerce Township, Michigan | 4 | 10.23 | 0.20 | 61 | 0.602 | 0.83 | 0.87 | 0.92 | 0.98 |
| **Median** | | | **0.65** | | | **0.25** | **0.32** | **0.52** | **0.98** |

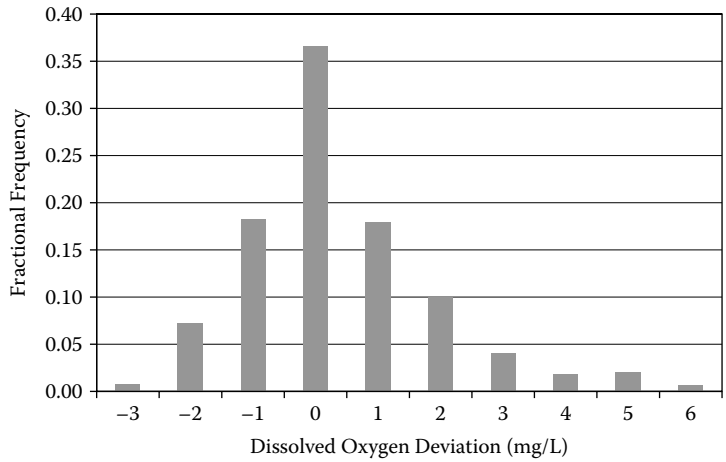



**FIGURE 5.8** Variation about the mean trend for dissolved oxygen leaving the Tres Rios, Arizona, FWS Hayfield wetlands (H1 and H2). See Figure 5.7 for the annual times series.

The median time of the maximum in outflow DO is early February (yearday = 32, Table 5.3).

Vymazal and Kröpfelová (2006) found little seasonal variation in the DO in the outflow of a number of Czech HSSF wetlands. The same is true of the various HSSF wetlands in the United States that do not display any measurable DO in the outflow.

The percentile points of the DO scatter around the annual cosine trends are given in Table 5.3. It is seen that with some frequency, the excursions from the trend DO values are lower by a considerable margin. For instance, 5% of the time, the median DO is only 25% of the trend value (Table 5.3). This means that none of the example FWS systems in Table 5.3 satisfy the United States DO requirement for discharge to receiving waters at the 95% level of confidence (greater than 5 mg/L 19 times out of 20). This means that extra design features (such as cascade aeration) must be implemented to meet the DO requirement for surface discharges. The same conclusion would be reached for HSSF wetlands, certainly in the United States, but also in the broader context of all HSSF wetlands.

## 5.3 VOLATILIZATION

Although oxygen transfer is a critical feature of treatment wetlands, there are several other gases that transfer to and from the ecosystem. Incoming volatile anthropogenic chemicals may be lost. But a treatment wetland also takes in atmospheric carbon dioxide for photosynthesis, and expels it from respiratory processes. The various treatment processes create product gases, which are also expelled from the wetland. These include ammonia, hydrogen sulfide, dinitrogen, nitrous oxide, and methane. Of these, carbon dioxide, nitrous oxide, and methane are regarded as greenhouse gases, and are of concern as atmospheric pollutants. As a result, there have been several treatment wetland studies focused on these three gases. Volatilization of ammonia is discussed in Chapter 9, and volatilization of hydrogen sulfide in Chapter 11.

Methane is produced by anaerobic processes with the wetland substrate. Carbon dioxide is produced by aerobic microbial processes, and by root respiration. Nitrous oxide is a possible product of (incomplete) denitrification. Because these greenhouse gases contribute to global warming, they have received attention in the context of treatment wetlands (Brix *et al*., 2001; Teiter and Mander, 2005).

### Nitrous Oxide

Denitrification typically proceeds through a sequence of steps, ultimately leading to formation of dinitrogen (see Chapter 9). An intermediate product is $N_2O$, which may be emitted prior to complete reduction. Partial oxidation of ammonia (partial nitrification) is another candidate mechanism for $N_2O$ formation. $^{15}N$ experiments have sometimes shown that this reaction is not dominant (Itokawa *et al*., 2001), but in other circumstances have identified partial nitrification as the primary source (Beline *et al*., 2001).

$N_2O$ is stable in the atmosphere, with a lifetime of over 100 years. It also is a major contributor to global warming, with a carbon dioxide equivalency of about 300. A number of studies have used chamber assay methods to measure $N_2O$ emission in treatment wetlands, both FWS (Freeman *et al*., 1997; Gui *et al*., 2000; Johansson *et al*., 2003; Mander *et al*., 2003; Johansson *et al*., 2004; Hernandez and Mitsch, 2005; Søvik *et al*., 2006; Liikanen *et al*., 2006); and SSF (Kløve *et al*., 2002; Mander *et al*., 2003; Teiter and Mander, 2005; Søvik *et al*., 2006). The rates of emission average about 4,000 $\mu gN/m^2 \cdot d$ for 15 wetlands, which amounts to an average of 2.2% of the nitrogen load removed in the wetlands (Table 5.4).

Denitrification is strongly seasonal, with larger rates in warm seasons, therefore it is not surprising that nitrous oxide emission is also seasonal, with maxima in summer (Teiter and Mander, 2005; Hernandez and Mitsch, 2005). However, Johansson *et al*. (2003) found no seasonality at the Nykvarn FWS treatment wetlands near Linköping, Sweden.

## TABLE 5.4
## Greenhouse Gas Emissions from Treatment Wetlands

| Wetland Type and Country | Location | Details | Reference | $CO_2$-C Emission Rate ($gC/m^2 \cdot d$) | $CH_4$-C Emission Rate ($mgC/m^2 \cdot d$) | Estimated % of Load Removed | $N_2O$-N Emission Rate ($\mu gN/m^2 \cdot d$) | Estimated % of Load Removed |
|---|---|---|---|---|---|---|---|---|
| **FWS** | | | | | | | | |
| China | Liaohe Delta | Summer, natural | Huang *et al.* (2005) | — | — | — | 41,000 | — |
| Estonia | Kodijärve | Summer | Søvik *et al.* (2006); Mander *et al.* (2003) | 0.96 | 340 | — | 7,100 | 0.66 |
| Finland | Hovi | Summer | Søvik *et al.* (2006) | 0.21 | 29 | –250 | 400 | 7.6 |
| Finland | Kompsasuo | Infilt | Søvik *et al.* (2006); Liikanen *et al.* (2006) | 0.73 | 310 | 390 | 190 | 0.29 |
| Finland | Lakeus | Summer | Søvik *et al.* (2006) | 2.00 | 350 | 57 | 350 | 0.17 |
| Finland | Ruka | Infiltration | Søvik *et al.* (2006) | 1.30 | 72 | 59 | 4,900 | 0.95 |
| Norway | Skjønhaug | Summer | Søvik *et al.* (2006) | 0.98 | — | 13 | 4,000 | 0.28 |
| China | Jiaonan | — | Gui *et al.* (2000) | — | 19 | 0.13 | 4,000 | 0.10 |
| Sweden | Ormastorp | SAV | Stadtmark and Leonardson (2005) | — | 240 | — | — | — |
| Sweden | Görarp | SAV | Stadtmark and Leonardson (2005) | — | 240 | — | — | — |
| Sweden | Genarp | SAV | Stadtmark and Leonardson (2005) | — | 240 | — | — | — |
| Sweden | Nykvarn | — | Heiberg (1999); Johansson *et al.* (2003; 2004) | — | 135 | — | 1,985 | 0.37 |
| Wales | Cerig-yr-Wyn | — | Freeman *et al.* (1997) | — | — | — | 233 | — |
| United States | Columbus, Ohio | — | Hernandez and Mitsch (2005) | — | — | — | 92 | 0.48 |
| **HSSF** | | | | | | | | |
| Estonia | Kõo | Summer | Sovik *et al.* (2006); Mander et al. (2003) | 0.38 | 160 | — | 4,200 | 0.17 |
| Norway | Ski | Summer | Sovik *et al.* (2006) | 0.26 | 130 | 24 | 6,900 | 3.3 |
| Poland | Nowa Slupia | Summer | Sovik *et al.* (2006) | 0.56 | 670 | — | — | — |
| Denmark | Kalø | — | Brix (1990) | 0.56 | 220 | 6.8 | — | — |
| New Zealand | Hamilton | High, Veg, Up | Tanner *et al.* (1997) | — | 142 | 9.0 | — | — |
| New Zealand | Hamilton | High, Veg, Down | Tanner *et al.* (1997) | — | 34 | 2.1 | — | — |
| New Zealand | Hamilton | Low, Veg, Up | Tanner *et al.* (1997) | — | 116 | 12.3 | — | — |
| New Zealand | Hamilton | Low, Veg, Down | Tanner *et al.* (1997) | — | 65 | 6.9 | — | — |
| New Zealand | Hamilton | M1 | Tanner *et al.* (2002) | 2.62 | 378 | 0.36 | — | — |
| New Zealand | Hamilton | D1 | Tanner *et al.* (2002) | 1.98 | 141 | 1.67 | — | — |
| New Zealand | Hamilton | D2 | Tanner *et al.* (2002) | 1.09 | 103 | 3.24 | — | — |
| New Zealand | Hamilton | D2A | Tanner *et al.* (2002) | 1.55 | 103 | 12.13 | — | — |
| Norway | Jordforsk | Experiment 6 | Kløve *et al.* (2002) | — | — | — | 890 | 0.06 |
| **VF** | | | | | | | | |
| Estonia | Kõo | Summer | Sovik *et al.* (2006); Mander *et al.* (2003) | 1.60 | 110 | — | 15,000 | 0.28 |
| Norway | Ski | Summer | Sovik *et al.* (2006) | 3.90 | 140 | 0.63 | 9,600 | 16 |
| **Mean** | | | | **1.29** | **187** | **20.5** | **3,989** | **2.2** |
| **Median** | | | | **1.03** | **141** | **6.9** | **4,000** | **0.3** |

**TABLE 5.5**
**Gas Emissions in Different Plant Communities in the Nykvarn, Sweden, FWS Treatment Wetland**

| Plant Community | N | $CH_4$ Flux ($mg/m^2 \cdot d$) | $N_2O$ Flux ($mg/m^2 \cdot d$) |
|---|---|---|---|
| *Typha latifolia* | 146 | 163 | 3.84 |
| *Phalaris arundinacea* | 12 | 318 | 5.95 |
| *Spirogyra* spp. | 111 | 168 | 1.53 |
| *Glyceria maxima* | 37 | 160 | 1.19 |
| *Lemna minor* | 4 | 675 | 2.27 |
| No plants | 15 | 245 | 5.95 |

*Source:* Data from Johansson *et al*. (2003) *Tellus* 55B: 737–750; Johansson *et al*. (2004) *Water Research* 38: 3960–3970.

There is also potentially an effect of the particular plant community on $N_2O$ emissions (Table 5.5). At the Nykvarn, Sweden, site, studies showed that plants generally reduced $N_2O$ emissions, but the opposite was found at the Olentangy site in Columbus, Ohio.

## Methane

Methanogenesis occurs frequently in the sediment layers of treatment wetlands, particularly HSSF systems, and particularly in wetlands receiving high loads of CBOD. Carbohydrates from various sources are broken down by fermentation, forming low molecular weight compounds which are then further broken down into methane and water by methanogenic bacteria (Equation 5.21). The methane so formed may either be oxidized, or exit the wetland via plant airways or volatilization from sediments and water (Figures 5.9 and 5.10).

Methane is stable in the atmosphere, with a lifetime of over eight years. It also is a major contributor to global warming, with a carbon dioxide equivalency of about 23. A number of studies have used chamber assay methods to measure $CH_4$ emission in treatment wetlands, both FWS (Gui *et al*., 2000; Johansson *et al*., 2003; Mander *et al*., 2003; Johansson *et al*., 2004; Søvik *et al*., 2006; Liikanen *et al*., 2006); and SSF (Brix, 1990; Tanner *et al*., 1997; Kløve *et al*., 2002; Tanner *et al*., 2002a; Mander *et al*., 2003; Teiter and Mander, 2005; Søvik *et al*., 2006). The rates of emission average about 187 $mgC/m^2 \cdot d$ for 24 wetlands, which amounts to an average of 20% of the carbon load removed in the wetlands (Table 5.4).

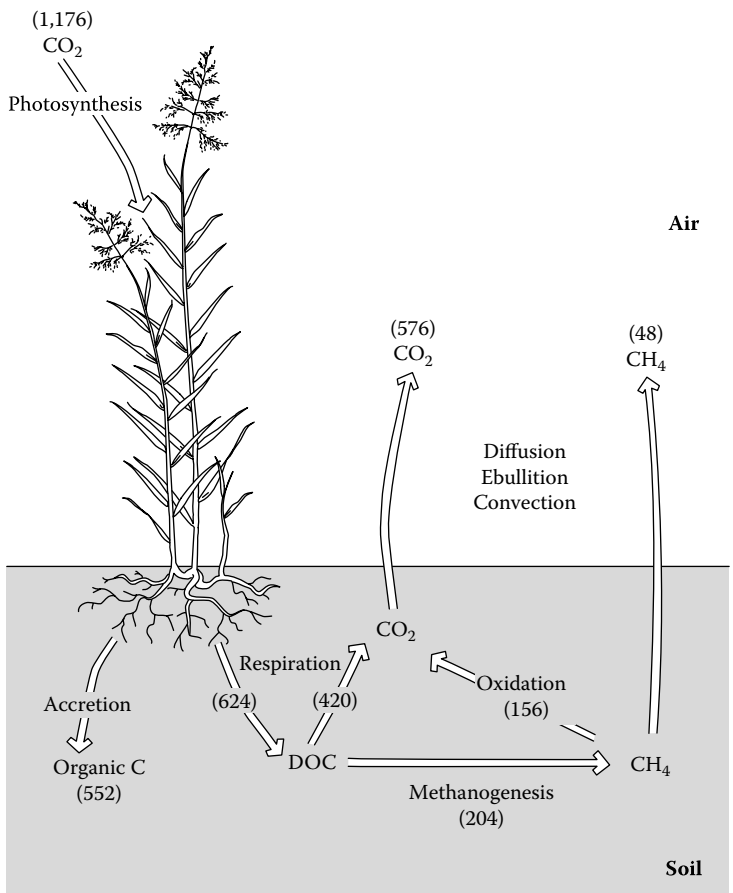



**FIGURE 5.9** Carbon processing and gas emission in treatment wetlands. The numbers are fluxes in $gC/m^2 \cdot yr$, as measured for a *Phragmites* stand at the Vejlerne Nature Preserve in Denmark. Inflows and outflows of carbon with water are minimal in this natural wetland. By comparison with values in Table 5.4, these numbers are not far different from treatment wetland values. (Redrawn from Brix *et al*. (2001) *Aquatic Botany* 69: 313–324. Reprinted with permission.)

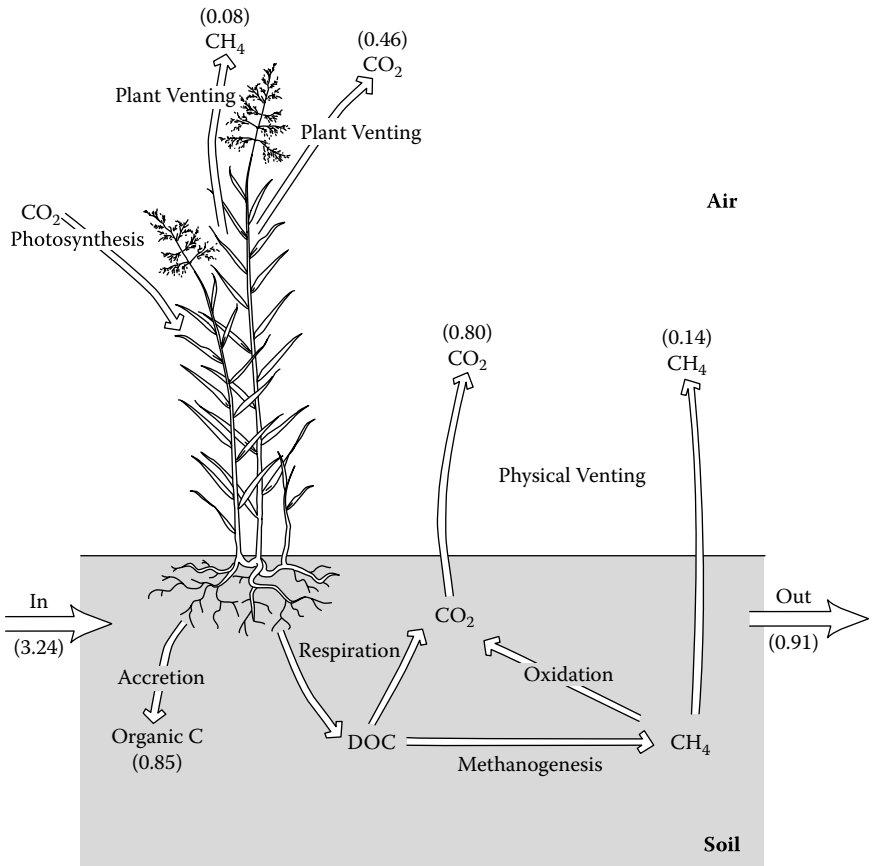



**FIGURE 5.10** Carbon processing and gas emission in the HSSF treatment wetlands at Kalø, Denmark, in April. The numbers in italics are fluxes in gC/m²·d. The *Phragmites* stand was in a senesced state. (Redrawn from Brix (1990) *Water Research* 24(2): 259–266. Reprinted with permission.)

There is also potentially an effect of the particular plant community on $CH_4$ emissions (Table 5.5). At the Nykvarn, Sweden, site, studies showed that some plants reduced $CH_4$ emissions, but others showed greater emission, compared to zones with no plants. Sorrell and Boon (1992) found that plants slightly reduced the methane emissions measured in a natural Australian wetland.

Part of the reason for differences from one plant community to another has to do with the various mechanisms of gas exchange (see Figure 5.10). The airways associated with emergent plants function as both influx and efflux conduits for gases (Sorrell and Armstrong, 1994). Live plant culms can serve either function, and indeed adjacent culms attached to the same rhizome may serve opposite functions. Standing dead (and perhaps broken) stems or culms can also transport gases. Figure 5.10 shows the losses from a *Phragmites* HSSF wetland in Denmark during April, when standing dead culms dominated the reedbed. A substantial proportion of methane loss was via the plants. The zones between plants provide for the loss of gases by volatilization from water and substrate. Obviously, the plant density affects the relative proportions of the two mechanisms.

Methane emission is strongly seasonal, with larger rates in warm seasons (Figure 5.11) (Johansson *et al*., 2004; Teiter and Mander, 2005). Reports from treatment wetland studies are reinforced by results from studies on natural wetlands, such as those of Sorrell and Boon (1992).

Wetlands exhibit strong longitudinal gradients in carbon compounds, as treatment proceeds in the flow direction. It is therefore expected that there should be gradients in methane generation, and indeed that is the case (Figure 5.12).

## Carbon Dioxide

Carbon dioxide is utilized by plants and algae in photosynthesis. It is produced by respiration in the root system of plants, and by microbial processes in soils and sediments. Oxidation of carbonaceous components of waters is largely dissipated by oxidation to $CO_2$. As a result, large fluxes of $CO_2$ are present in wetlands, some as influxes to the green plants, and some as releases. Figure 5.9 illustrates an approximate annual mass balance for carbon in a *Phragmites* wetland that is not receiving any wastewater. Approximately 50% of the net annual photosynthesis $CO_2$ fixation is ultimately respired to $CO_2$ and $CH_4$ in the sediment, but only small proportions are directly released to the atmosphere (Brix *et al*., 2001). The fixation of atmospheric carbon dioxide is synchronous with the growing season. The moderately large standing crops of biomass require on the order of 1,000–2,000 gC/m²·yr.

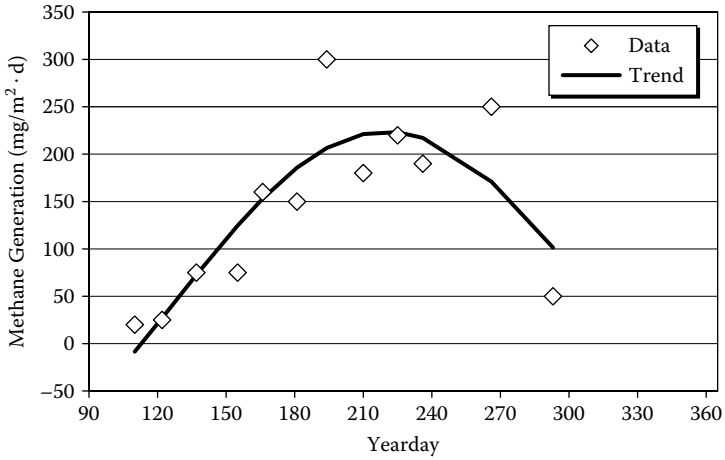



**FIGURE 5.11** Seasonal trend in methane production from the Nykvarn, Sweden, FWS treatment wetland. (From Johansson *et al.* (2004) *Water Research* 38: 3960–3970. Reprinted with permission.)

Untreated municipal wastewaters have ratios of TOC to CBOD of 0.5–0.8, settled wastewaters are 0.8–1.2, and treated effluents are 2–5 (Metcalf and Eddy, Inc., 1991; Crites and Tchobanoglous, 1998). Treatment wetlands receiving secondary, tertiary, and lagoon waters have ratios of TOC to CBOD of 5–10 (see Table 8.1). Loadings of BOD are typically in the range of 40–4,000 g/m$^2$·yr, and thus carbon loadings are roughly 100–10,000 gC/m$^2$·yr. Consequently, either atmospheric fixation or influent carbon loadings may be dominant in a treatment wetland. FWS wetlands treating secondary or tertiary effluents would fixation-dominated, whereas systems treating septic tank effluents would be influent-dominated with respect to carbon.

As for nitrous oxide and methane, part of the emitted $CO_2$ is lost through plant airways, and part via losses from the soil and water air interfaces (Figure 5.10). The rates of emission average about 1.3 gC/m$^2$·d (500 gC/m$^2$·yr) for 16 wetlands (Table 5.4). As noted by Brix (1990), it is difficult to generalize about how much of the incoming carbon load is dissipated to carbon dioxide, because of the interactions of $CO_2$ and $CH_4$ in methanogenesis (Equations 5.20 and 5.21), and because of the dual sources of incoming water and the atmosphere. Nonetheless, the amounts of $CO_2$ emitted to the atmosphere are not trivial compared to those loadings.

## Greenhouse Effects

Treatment wetlands sequester organic carbon via the accretion of new sediments and soils. However, they also emit greenhouse gases, $CO_2$, $CH_4$, and $N_2O$. The large multipliers for the radiative effect comparison (300 for nitrous oxide and 20 for methane) mean that small emissions of these gases can counteract the carbon sequestration function. Thus, although wetlands in general, including constructed wetlands, can act as carbon sinks, they still can increase the greenhouse effect because of their release of methane and nitrous oxide (Brix *et al.*, 2001). Because of the small acreage of treatment wetlands compared to natural wetlands, constructed systems are "not so remarkable" as sources of greenhouse

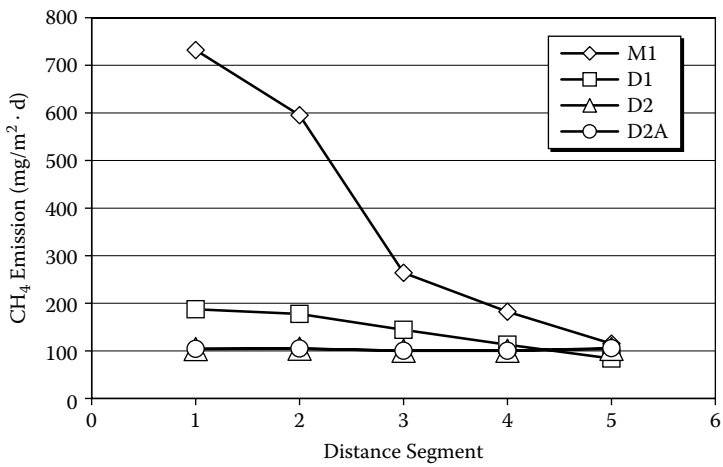



**FIGURE 5.12** Methane emissions from four SSF wetlands as a function of distance. Systems M1, D1, D2, and D2A treated different strengths of wastewater. (Data from Tanner *et al.* (2002a) *Ecological Engineering* 18(4): 499–520.)

gases (Mander *et al*., 2003). Liikanen *et al*. (2006) estimate that even if all global wastewater were treated in constructed wetlands, their share in atmospheric liability would be less than 1% of the total.

## 5.4 OXIDATION-REDUCTION POTENTIAL

Oxidation-reduction is a chemical reaction in which electrons are transferred from a donor to an acceptor. The electron donor loses electrons and increases its oxidation number or is oxidized; the acceptor gains electrons and decreases its oxidation number or is reduced. The driving force of a chemical reaction is the tendency of the free energy of the system to decrease until, at equilibrium, the sum of the free energies of the products equals that of the remaining reactants. In a reversible oxidation-reduction reaction, this driving force can be measured in Joules or in (milli)volts. Consider a reaction in which $n$ electrons are transferred:

$$\mathrm{Ox} + n\,\mathrm{e}^- \rightleftharpoons \mathrm{Red} \tag{5.13}$$

If the free energy change, represented in voltage, is measured against the standard hydrogen electrode, it is denoted by $E_h$. The equilibrium relation is then:

$$E_h = E_o + \frac{RT}{nF}\ln\left(\frac{[\mathrm{Ox}]}{[\mathrm{Red}]}\right) \tag{5.14}$$

where

$E_o$ = reference potential, mV
(zero for the standard hydrogen electrode)
$E_h$ = oxidation reduction potential, mV
$F$ = Faraday's constant, 96.4 J/mol·mV
$n$ = number of electrons transferred
$R$ = gas constant, 8.314 J/mol·°K
$T$ = temperature, °K

and in which the brackets denote concentrations. The interested reader may find more details in chemistry references, such as Ponnamperuma (1972), Pankow (1991), or Morel and Hering (1993).

$E_h$ is a quantitative measure of the tendency of a given system to oxidize or reduce susceptible substances. $E_h$ is positive and high in strongly oxidizing systems; it is negative and low in strongly reducing systems.

Oxidation-reduction conditions affect chemical and microbial processes, and have a very large effect on the biological availability of major and trace nutrients in soils in general (Patrick *et al*., 1985; Gambrell *et al*., 1987).

In submerged sediments and soils, redox potential ranges from around −400 mV (strongly reduced) to +700 mV (well oxidized). The oxidation of organic matter yields energy; the amount of energy depends on the nature of oxidant, or electron acceptor. Energetically, the most favorable oxidant is oxygen; after oxygen is depleted there follows a succession of organisms capable of reducing $NO_3^-$, $MnO_2$, FeOOH, $SO_4^{2-}$, and $CO_2$, with each oxidant yielding successively less energy for the organism mediating the reaction (Westall and Stumm, 1980). This succession leads to zonation, either in the vertical direction with depth into sediments in FWS wetlands, or in the radial direction around roots. The former case is illustrated in Figure 5.13, in which upper layers of the wetland bed display the more energetic reaction zones (Reddy and D'Angelo, 1994). It should be noted that the intermediate zones of Figure 5.13, in which the transition from oxic to anaerobic conditions occurs, are thin in FWS wetlands, typically comprising no more than one or two centimeters. Depending on the magnitude of the vertical transpiration flow, this zone thickness is controlled by downward advection of surface water and its redox potential, with much lesser contributions from diffusion. However, the zonation around wetland plant roots is much smaller still, with typical zone thicknesses of a millimeter or two (Figure 5.14). In HSSF wetlands, the dominant flow is through and under the rhizosphere, and therefore one or more zones may occupy most of the bed thickness (Table 5.2). The direction of supply of oxidants is transverse to the flow direction.

The chemistry of these thin transition layers maybe summarized in a number of equivalent ways (Reddy and D'Angelo, 1994; Mitsch and Gosselink, 2000b); here, a simple version is chosen with organic matter represented by $CH_2O$. Oxygen is the terminal electron acceptor in aerobic zones, and is reduced while electron donors are being oxidized, notably organic substances and ammonia. This reduction of $O_2$ to $H_2O$ is carried out by true aerobic microorganisms, and $CO_2$ is evolved as a waste product:

$$CH_2O + O_2 \rightarrow CO_2 + H_2O \tag{5.15}$$

As $O_2$ is depleted, nitrate will be used as electron acceptor followed by oxidized manganese compounds and then followed by ferric iron compounds. The order of these reductions is the same as that indicated by thermodynamic considerations (Reddy *et al*., 1986).

Nitrate is the next oxidant to be reduced following oxygen depletion. Many microorganisms can utilize $NO_3^-$ as terminal hydrogen acceptor instead of $O_2$, which is the denitrification process (see Chapter 9):

$$5CH_2O + 4NO_3^- \rightarrow 2N_2 + 4HCO_3^- + CO_2 + 3H_2O \tag{5.16}$$

As the redox potential continues to decrease, manganese is transformed from manganic to manganous compounds at about +200 mV (Laanbroek, 1990):

$$CH_2O + 3CO_2 + H_2O + 2MnO_2 \rightarrow 2Mn^{2+} + 4HCO_3^- \tag{5.17}$$

When the reduction of nitrate stops by depletion of this electron acceptor, the reduction of ferric oxide starts. A wide range of anaerobic bacteria are able to conserve energy through the reduction of $Fe^{3+}$ to $Fe^{2+}$ (Laanbroek, 1990; Younger *et al*., 2002). Many of these microorganisms also have the ability to grow through the reduction of $Mn^{4+}$ to $Mn^{2+}$.

$$CH_2O + 7CO_2 + 4Fe(OH)_3 \rightarrow 4Fe^{2+} + 8HCO_3^- + 3H_2O \tag{5.18}$$

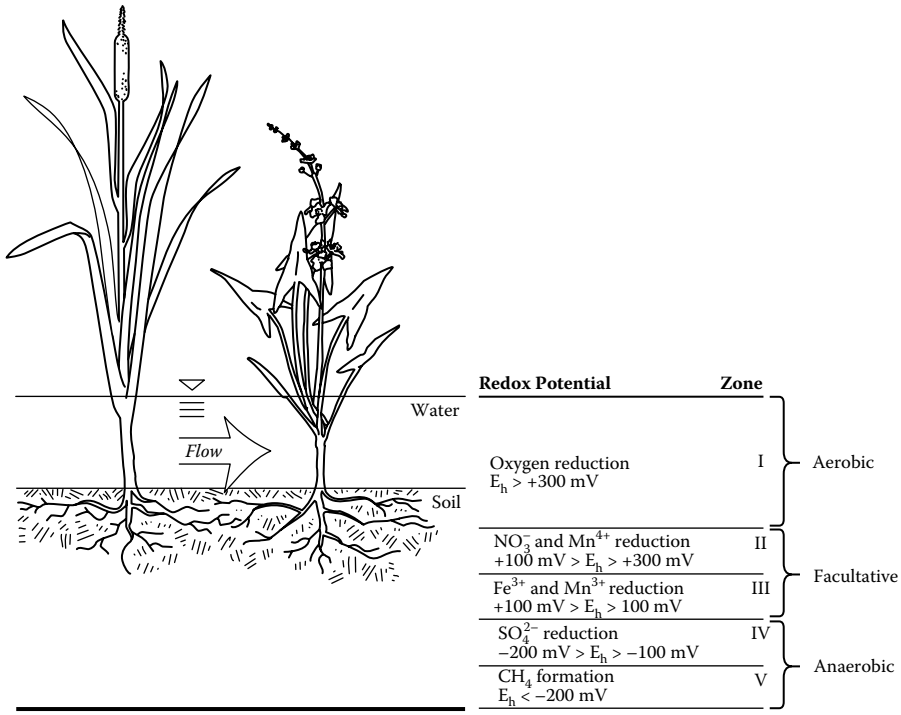



**FIGURE 5.13** Hypothetical vertical redox zonation in the soils under a FWS wetland.

Sulfate reduction occurs when the redox potential drops below –100 mV. Only a small amount of reduced sulfur is assimilated by the organisms, and virtually all is released into the external environment as sulfide (Wake *et al*., 1977).

$$CH_2O + SO_4^{2-} \rightarrow H_2S + 2HCO_3^- \qquad (5.19)$$

Sulfate reduction is promoted by design in wetlands built to remove metals with insoluble sulfides (Younger, 2000).

Methane production requires extremely reduced conditions, with a redox potential below –200 mV, after other terminal electron acceptors have been reduced.

$$4H_2 + CO_2 \rightarrow CH_4 + 2H_2O \qquad (5.20)$$

$$4H_2 + CH_3COOH \rightarrow 2CH_4 + 2H_2O \qquad (5.21)$$

Methanogenic bacteria utilize hydrogen as an electron source, but can also use formate ($HCOO^-$) or acetate ($CH_3COO^-$) (Equation 5.21). Methane is either released to the atmosphere or is oxidized to $CO_2$ by methanotrophic bacteria as soon as it enters the oxic zone.

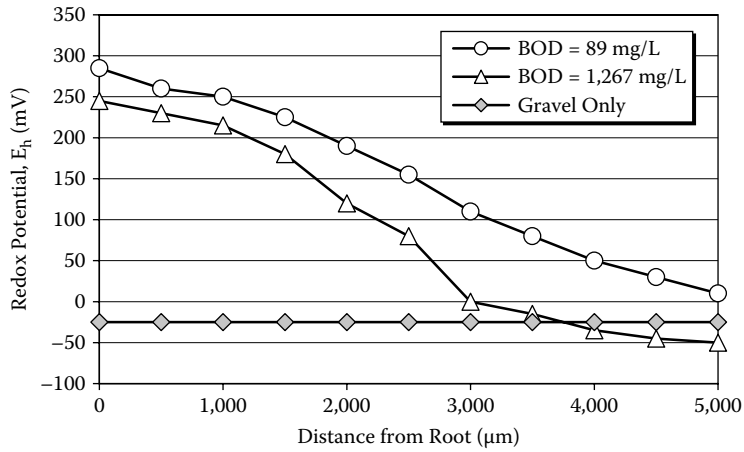



**FIGURE 5.14** Profiles of redox in the vicinity of main roots of *Schoenoplectus* (*Scirpus*) *validus* in a HSSF gravel bed wetland, along with an unvegetated control. These profiles were determined via micro-electrodes. Dissolved oxygen at the root surface was 1.0 mg/L decreasing to zero at 800 μm for BOD = 89 mg/L, and 2.0 mg/L decreasing to zero at 1,100 μm for BOD = 1,267 mg/L. (From Bezbaruah and Zhang (2004) *Biotechnology and Bioengineering* 88(1): 60–70. Reprinted with permission.)

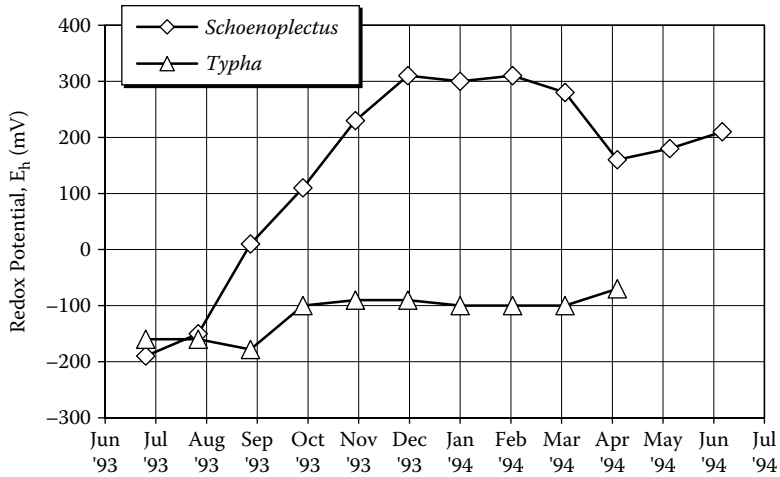



**FIGURE 5.15** Annual progression of redox potential present in FWS wetland soils located in North Carolina. Data points are averages of three depths into soil (2, 5, 10 cm), three longitudinal positions (25, 50 and 75%), and two wetlands. (From Szögi *et al.* (2004) *Applied Engineering in Agriculture* 20(2): 189–200. Reprinted with permission.)

### Redox Potentials in Treatment Wetlands

Szögi *et al.* (2004) studied the redox profiles in FWS wetlands receiving swine wastewater in Duplin County, North Carolina. The wetlands were of shallow depth (10 cm), and received light loadings (HLR = 2.1–2.8 cm/d, ammonia loadings 175–200 $gN/m^2 \cdot yr$). In general, there were slightly higher values near the soil surface, by 20–80 mV. The *Typha* wetlands were more anoxic than the *Schoenoplectus* wetlands (Figure 5.15).

Table 5.2 summarizes results from HSSF wetlands at Minoa, New York, and Vilagrassa, Spain. Typically, redox potentials are higher in the top layers of the HSSF beds than in the bottom. The Minoa beds were very anaerobic; the Vilagrassa beds were mildly anoxic, in terms of ORP values.

## 5.5 WETLAND HYDROGEN ION CONCENTRATIONS

Healthy aquatic systems can function only within a limited pH range. As a consequence, surface water discharge permits frequently require $6.5 < pH < 9.0$. Wetland water chemistry and biology are likewise affected by pH. Many treatment bacteria are not able to exist outside the range $4.0 < pH < 9.5$ (Metcalf and Eddy Inc., 1991). Denitrifiers operate best in the range $6.5 < pH < 7.5$, and nitrifiers prefer pH = 7.2 and higher. The same principles apply to other wetland biota; the acid bog vegetation is adapted to low pH, and differs greatly from the vegetation of an alkaline fen. In addition to controlling various biological processes, pH is also a determinant of several important chemical reactions. Ammonium changes to free ammonia at pH above neutral and at higher temperatures (see Chapter 9). The protonation of phosphorus changes with pH (see Chapter 10), and the hydroxide and oxyhydroxide precipitates of iron, manganese, and aluminum are pH sensitive (see Chapter 11). The pH value profoundly influences hydroxide, carbonate, sulfide, phosphate, and silicate equilibria in submerged soils. These equilibria regulate the precipitation and dissolution of solids, carbon equilibria (see last section of this Chapter), the sorption and desorption of ions, and the concentrations of nutritionally significant ions or substrates (Ponnamperuma, 1972).

Natural wetlands exhibit pH values ranging from slightly basic in alkaline fens (pH = 7–8) to quite acidic in sphagnum bogs (pH = 3–4) (Mitsch and Gosselink, 2000b). Natural freshwater marsh pH values are generally slightly acidic, (pH = 6–7). The organic substances generated within a wetland via growth, death, and decomposition cycles are the source of natural acidity. The resulting humic substances are large complex molecules with multiple carboxylate and phenolate groups. The protonated forms have a tendency to be less soluble in water, and precipitate under acidic conditions. As a consequence, wetland soil/water systems are buffered against incoming basic substances. They may be less well buffered against incoming acidic substances, since the water column contains a limited amount of soluble humics.

Treatment wetland effluent hydrogen ion concentrations are typically circumneutral. The notable exceptions are those wetlands receiving acid mine drainage, which reflect the low pH of the incoming waters. This special type of treatment wetland is not considered here; the reader is referred to Weider (1989) and Davis (1995). Furthermore, there is an important distinction between FWS and SSF systems in the ability of algae to conduct photosynthetic modulation of pH.

### Surface Flow Wetlands

In aquatic systems, algal photosynthetic processes peak during the daytime hours, creating a diurnal cycle in pH. Photosynthesis utilizes carbon dioxide and produces oxygen, thereby shifting the carbonate–bicarbonate–carbon dioxide equilibria to higher pH. During nighttime hours, photosynthesis

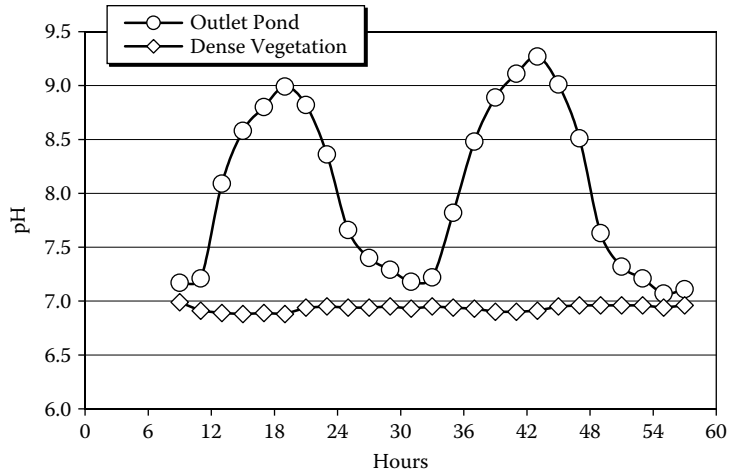



**FIGURE 5.16** Diurnal variation in pH a near the exit of Cell 7 at Sacramento, California. In dense vegetation just prior to the outlet deep zone, pH does not vary. In the outlet deep zone, there is a large diurnal swing in pH, presumably driven by algal activity in the open water. (Data from Nolte and Associates (1997) *Sacramento Regional Wastewater Treatment Plant Demonstration Wetlands Project*. 1996 Annual Report to Sacramento Regional County Sanitation District, Nolte and Associates: Sacramento, California.)

is absent, and algal respiration dominates, producing carbon dioxide and using oxygen. Open water zones within wetlands can develop high levels of algal activity, which in turn creates a high pH environment. Open water areas in wetlands also exhibit these phenomena. Diurnal pH fluctuations are not evident in areas with dense emergent vegetation. Data collected at the Sacramento, California, wetlands illustrate these phenomena (Figure 5.16). In a densely vegetated zone near the outlet, there is no diurnal cycle in pH. However, there is a large diurnal cycle in the outlet deep zone, in which the detention time is about one day. Large exports of TSS occurred episodically, indicating high algal activity, which is in turn consistent with the large pH swing.

Vegetated FWS wetlands produce effluent waters with pH just above neutrality. This occurs whether the incoming water is acidic (Figure 5.17) or basic (Figure 5.18). The Connell, Washington, wetlands treat food processing wastewater which is acidic, and which contains a large amount of nitrogen (TN of about 150 mg/L). The process of nitrification reduces alkalinity, and would be expected to drive pH downward. However, other wetland processes are involved, such as solids and COD removal, and the wetland causes a pH increase (Figure 5.17). In contrast, the Estevan, Saskatchewan, FWS wetlands treat municipal wastewater from lagoon pretreatment, which produces a high pH influent to the wetlands. The combination of wetland processes drives the pH downward (Figure 5.18).

The annual trends in FWS pH are typically quite weak (Figure 5.19). The residuals account for about one third of the variability, are normally distributed, and are independent of the time of the year. Because of these weak annual trends, FWS behavior can be adequately described by an annual mean and the associated standard deviation (Table 5.6). The pH produced in FWS treatment wetlands is within a surprisingly narrow band. Constructed systems treating municipal effluents produce an intersystem annual average of pH = 7.18 ± 0.35 (N = 20, total years data = 56). Nine of these twenty constructed wetlands exhibited a weak annual cycle, with a

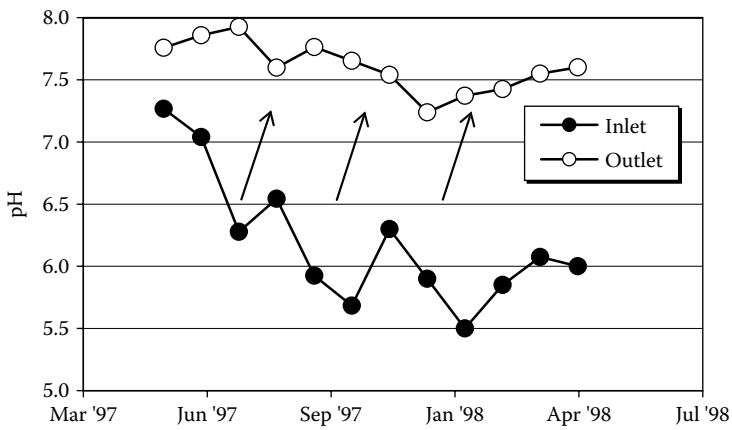



**FIGURE 5.17** Water enters the Connell, Washington, FWS wetlands at low pH, and is modified to values just above neutral.

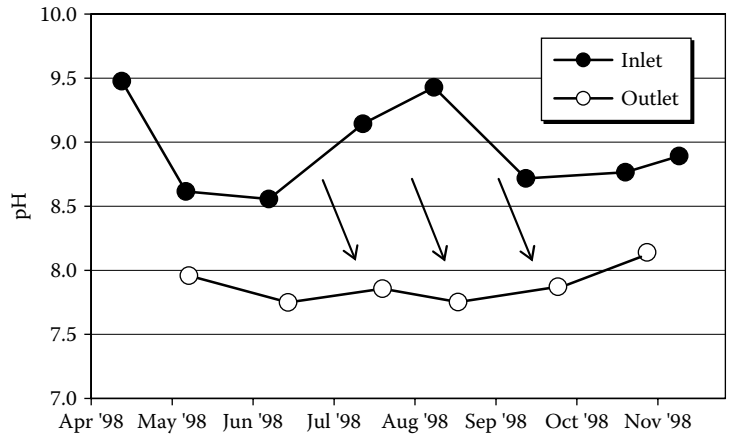



**FIGURE 5.18** Water enters the Estevan, Saskatchewan, FWS wetlands at high pH, and is modified to values just above neutral.

mean amplitude of 0.25. Most of these nine contained significant amounts of open water, including terminal deep zones. Industrial and groundwater sources may cause wetlands to produce pH about a half unit higher (Table 5.6).

Natural treatment wetlands produce slightly lower pH, by about 0.5 units. This is possibly due to the antecedent peat soils that occupied most of these. Continued application of circumneutral wastewater to a naturally acidic wetland can eventually alter the pH of the surface waters in the wetland. This was the case for an acid sphagnum–black spruce bog, which received circumneutral wastewater for approximately 25 years (Kadlec and Bevis, 1990), as well as for a slightly acid peatland at Houghton Lake receiving slightly basic lagoon water. The effect on the peatland in both cases was the partial solubilization of the solid humic substances that formed under more acidic natural conditions. In addition to the chemical effect of humic solubilization, those decomposition processes that were acid-inhibited can resume under the less acidic conditions.

Treatment wetland information thus allows prediction of FWS wetland water effluent pH to within about ±0.3 units, based upon the character of the influent and the open water fraction and location in the wetland.

### Subsurface Flow Wetlands

Subsurface flow wetlands also moderate and buffer the pH variations and levels of incoming basic waters (Table 5.7). There are typically weak or nonexistent annual cycles, and pH is driven to values just above neutral. For example, for the Holtby, United Kingdom, HSSF system (Figure 5.20), residuals comprise a large portion (90%) of the variability, are normally distributed, and are independent of the time of the year. Vertical, transverse and longitudinal pH profiles have been monitored at Minoa, NERCC, and Grand Lake. These data show essentially no spatial variability within the beds. As a consequence, system performance is adequately described by input/output information (Table 5.7). Twenty-four United Kingdom

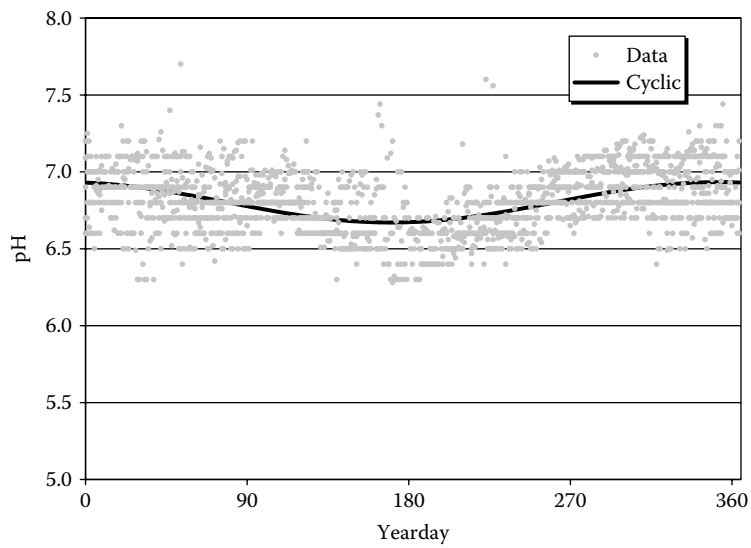



**FIGURE 5.19** The annual cyclic trend in daily effluent pH from the Titusville, Florida, FWS wetland. There is a midsummer minimum, and the amplitude of the cycle is only $A = 0.13$ pH units. Trend line is a least-squares fit to an equation of the form of Equation 6.1.

## TABLE 5.6
## Effluent pH for Several Classes of FWS Treatment Wetlands

| Site | Wetland | Location | Source Water | Data Years/ Operational Years | pH | Standard Deviation | Percent Open Water | Annual Cycle Mplitude | Summer pH | Peak Time |
|---|---|---|---|---|---|---|---|---|---|---|
| **Constructed Municipal** | | | | | | | | | | |
| Columbia | All | Missouri | Secondary | 3/10 | 7.41 | 0.12 | — | None | 7.41 | — |
| Orlando Easterly | Stratum 1 | Florida | Tertiary | 8/13 | 6.91 | 0.19 | — | None | 6.91 | — |
| Orlando Easterly | Stratum 2 | Florida | Tertiary | 8/13 | 6.87 | 0.20 | — | None | 6.87 | — |
| Orlando Easterly | Stratum 3 | Florida | Tertiary | 8/13 | 6.99 | 0.21 | — | None | 6.99 | — |
| Tres Rios | H1 | Arizona | Secondary, partial nit-denit | 2/6 | 7.04 | 0.13 | 25 | None | 7.04 | — |
| Tres Rios | H2 | Arizona | Secondary, partial nit-denit | 2/6 | 7.06 | 0.10 | 25 | None | 7.06 | — |
| Tres Rios | C1 | Arizona | Secondary, partial nit-denit | 2/6 | 7.09 | 0.11 | 15 | None | 7.09 | — |
| Tres Rios | C2 | Arizona | Secondary, partial nit-denit | 2/6 | 7.12 | 0.10 | 10 | None | 7.12 | — |
| Sacramento | 1 | California | Secondary | 1/5 | 6.92 | 0.17 | 25 | 0.15 | 7.07 | Summer peak |
| Sacramento | 5 | California | Secondary | 1/5 | 7.06 | 0.15 | 36 | 0.20 | 7.26 | Summer peak |
| Sacramento | 7 | California | Secondary | 1/5 | 6.89 | 0.09 | 33 | 0.10 | 6.99 | Summer peak |
| Listowel | System 3 | Ontario | Lagoon | 3/4 | 7.06 | 0.21 | 0 | None | 7.06 | — |
| Listowel | System 4 | Ontario | Lagoon | 3/4 | 7.02 | 0.21 | 0 | None | 7.02 | — |
| Richmond | Emergent | New South Wales | Secondary | 2/3 | 6.78 | 0.20 | 0 | 0.10 | 6.68 | Winter peak |
| Richmond | Open water | New South Wales | Secondary | 2/3 | 7.83 | 0.61 | 100 | 0.55 | 8.38 | Summer peak |
| Warangal | All | India | Screened raw | 1/1 | 7.32 | 0.03 | 0 | None | 7.32 | — |
| Byron Bay | All | New South Wales | Advanced secondary | 3/3 | 6.91 | 0.31 | 50 | 0.27 | 7.18 | Summer peak |
| Minot | All | North Dakota | Lagoon | 1/10 | 7.91 | 0.26 | 59 | 0.29 | 8.20 | Summer peak |
| Brighton | All | Ontario | Lagoon | 2/3 | 7.57 | 0.21 | 10 | 0.40 | 7.97 | Double peak |
| Estevan | All | Saskatchewan | Lagoon | 1/6 | 7.84 | 0.22 | 10 | | | — |
| **Mean** | | | | | **7.18** | | | | **7.24** | |
| **Standard Deviation** | | | | | **0.35** | | | | **0.45** | |
| **Natural Municipal** | | | | | | | | | | |
| Drummond | All | Wisconsin | Lagoon | 6/6 | 4.61 | 0.72 | 0 | None | 4.61 | — |
| Houghton Lake | All | Michigan | Lagoon | 14/25 | 6.47 | 0.54 | 0 | None | 6.47 | — |
| Cannon Beach | All | Oregon | Lagoon | 16/16 | 6.71 | 0.24 | 0 | 0.10 | 6.81 | Summer peak |
| Genoa–Oceola | All | Michigan | RIB | 11/11 | 6.90 | 0.35 | 5 | None | 6.90 | — |
| Portage-base | All | Michigan | RIB | 11/11 | 7.03 | 0.39 | 0 | None | 7.03 | — |
| **Mean (excluding Drummond)** | | | | | **6.78** | | | | **6.80** | |
| **Standard Deviation** | | | | | **0.24** | | | | **0.24** | |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| **Constructed Other Sources** | | | | | | | | | | |
| Des Plaines | EW3 | Illinois | River | 1/9 | 7.96 | 0.34 | 75 | 0.49 | 8.45 | Spring peak |
| Des Plaines | EW4 | Illinois | River | 1/9 | 7.97 | 0.22 | 60 | 0.30 | 8.27 | Spring peak |
| Des Plaines | EW5 | Illinois | River | 1/9 | 8.44 | 0.40 | 50 | 0.18 | 8.62 | Spring peak |
| Schilling | EA | Michigan | Groundwater | 3/4 | 7.31 | 0.36 | 20 | None | 7.31 | — |
| Schilling | ET | Michigan | Groundwater | 3/4 | 7.66 | 0.32 | 60 | None | 7.66 | — |
| Schilling | WT | Michigan | Groundwater | 3/4 | 7.54 | 0.38 | 30 | None | 7.54 | — |
| Schilling | WA | Michigan | Groundwater | 3/4 | 7.39 | 0.24 | 15 | None | 7.39 | — |
| Everglades Nutrient Removal Project | All | Florida | Agricultural runoff | 5/9 | 7.39 | 0.27 | 50 | None | 7.39 | — |
| New Hanover | Raw | North Carolina | Leachate | 2/4 | 7.70 | 0.38 | 10 | 0.25 | 7.95 | Summer peak |
| New Hanover | Treated | North Carolina | Leachate | 2/4 | 7.55 | 0.26 | 10 | 0.16 | 7.71 | Summer peak |
| Connell | W1/2 | Washington | Food processing | 1/7 | 7.61 | 0.19 | 0 | 0.3 | 7.72 | Summer peak |
| **Natural Other Sources** | | | | | | | | | | |
| Northern mine | All | Ontario | Minewater lagoon | 4/4 | 7.35 | 0.31 | 10 | None | 7.35 | — |
| **Mean** | | | | | **7.66** | | | | **7.78** | |
| **Standard Deviation** | | | | | **0.33** | | | | **0.45** | |

**TABLE 5.7**
**Examples of pH in HSSF Treatment Wetlands**

| Site | Wetland | Location | Source Water | Data Years | Inlet pH | Standard Deviation | Outlet pH | Standard Deviation |
|---|---|---|---|---|---|---|---|---|
| **United States** | | | | | | | | |
| Grand Lake | — | Minnesota | STE | 4 | 7.33 | 0.28 | 7.16 | 0.19 |
| NERCC | 1 | Minnesota | STE | 3 | 7.19 | 0.13 | 7.06 | 0.15 |
| NERCC | 2 | Minnesota | STE | 3 | 7.19 | 0.13 | 7.06 | 0.16 |
| Minoa | Planted | New York | Primary | 2 | 7.15 | 0.23 | 7.05 | 0.23 |
| Minoa | Unplanted | New York | Primary | 2 | 7.15 | 0.23 | 7.08 | 0.21 |
| Carville | — | Louisiana | Lagoon | 4 | — | — | 7.3 | 0.3 |
| Benton | — | Louisiana | Lagoon | 3 | 8.5 | 0.7 | 7.3 | 0.3 |
| Mandeville | — | Louisiana | Lagoon | 1 | — | — | 7.2 | 0.2 |
| Haughton | — | Louisiana | Lagoon | 4 | 7.5 | 0.6 | 7.2 | 0.2 |
| Benton | 3 | Kentucky | Lagoon | 1 | 7.46 | 0.55 | 7.05 | 0.23 |
| **Australia, New Zealand** | | | | | | | | |
| Richmond | Cattail | NSW | Secondary | 2 | 7.23 | 0.15 | 6.73 | 0.23 |
| Richmond | Bulrush | NSW | Secondary | 2 | 7.23 | 0.15 | 6.78 | 0.20 |
| Richmond | Unplanted | NSW | Secondary | 2 | 7.23 | 0.15 | 6.90 | 0.19 |
| Portland | — | New Zealand | — | — | 9.15 | 1.00 | 7.18 | 0.51 |
| Waipoua | — | New Zealand | — | — | 7.32 | 0.27 | 6.96 | 0.24 |
| **Scandinavia** | | | | | | | | |
| Esval | — | Norway | Leachate | 5 | 7.5 | — | 7.5 | — |
| Haugstein | — | Norway | STE | 5 | 7.3 | — | 7.3 | — |
| Tveter | — | Norway | STE | 5 | 8.5 | — | 7.4 | — |
| **Mean** | | | | | **7.56** | | **7.12** | |
| **Standard Deviation** | | | | | **0.60** | | **0.20** | |

| Site | Wetland | Location | Source Water | Data Years | Inlet pH | Standard Deviation | Outlet pH | Standard Deviation |
|---|---|---|---|---|---|---|---|---|
| **United Kingdom** | | | | | | | | |
| Cheshire, England | 1 | U.K. | STE | 2 | 7.53 | 0.16 | 7.16 | 0.17 |
| Cheshire, England | 2 | U.K. | STE | 2 | 7.53 | 0.16 | 7.21 | 0.15 |
| Cheshire, England | 3 | U.K. | STE | 2 | 7.53 | 0.16 | 7.22 | 0.21 |
| Cheshire, England | 4 | U.K. | STE | 2 | 7.53 | 0.16 | 7.15 | 0.13 |
| Cheshire, England | 5 | U.K. | STE | 2 | 7.53 | 0.16 | 7.24 | 0.13 |
| Cheshire, England | 6 | U.K. | STE | 2 | 7.53 | 0.16 | 7.31 | 0.37 |
| Cheshire, England | 7 | U.K. | STE | 2 | 7.53 | 0.16 | 7.23 | 0.19 |
| Cheshire, England | 8 | U.K. | STE | 2 | 7.53 | 0.16 | 8.13 | 0.20 |
| Cheshire, England | 9 | U.K. | STE | 2 | 7.53 | 0.16 | 7.18 | 0.13 |
| Cheshire, England | 10 | U.K. | STE | 2 | 7.53 | 0.16 | 7.35 | 0.15 |
| Essex, England | Lower | U.K. | STE | 1 | 8.02 | 0.25 | 7.70 | 0.27 |
| Essex, England | Upper | U.K. | STE | 1 | 8.02 | 0.25 | 7.91 | 0.30 |
| Londonderry, Northern Ireland | 1 | U.K. | STE | 7 | 7.02 | 0.25 | 7.09 | 0.56 |
| Londonderry, Northern Ireland | 2 | U.K. | STE | 7 | 7.02 | 0.25 | 7.10 | 0.31 |
| Londonderry, Northern Ireland | 3 | U.K. | STE | 7 | 7.01 | 0.26 | 6.98 | 0.27 |
| Londonderry, Northern Ireland | 4 | U.K. | STE | 7 | 6.99 | 0.30 | 6.95 | 0.23 |
| Yorkshire, England | — | U.K. | STE | 3 | 8.21 | 0.34 | 7.40 | 0.21 |
| Leicestershire, England | — | U.K. | STE | 2 | 7.54 | 0.35 | 7.50 | 0.19 |
| North Yorkshire, England | — | U.K. | STE | 9 | 7.64 | 0.35 | 7.50 | 0.41 |
| Fife, Scotland | 1 | U.K. | STE | 2 | 7.65 | 0.35 | 7.56 | 0.14 |
| Fife, Scotland | 2 | U.K. | STE | 2 | 7.65 | 0.35 | 7.88 | 0.11 |
| Fife, Scotland | 3 | U.K. | STE | 2 | 7.65 | 0.41 | 7.18 | 0.20 |
| Fife, Scotland | 4 | U.K. | STE | 2 | 7.65 | 0.41 | 7.12 | 0.07 |
| **Mean** | | | | | **7.54** | | **7.35** | |
| **Standard Deviation** | | | | | **0.31** | | **0.31** | |

(*Continued*)

**TABLE 5.7 (CONTINUED)**
**Examples of pH in HSSF Treatment Wetlands**

| Site | Wetland | Location | Source Water | Data Years | Inlet pH | Standard Deviation | Outlet pH | Standard Deviation |
|---|---|---|---|---|---|---|---|---|
| **Czech Republic** | | | | | | | | |
| Mořina | — | CR | Primary | 2 | 7.76 | 0.21 | 7.69 | 0.10 |
| Chlístovice | — | CR | Secondary | 2 | 7.40 | 0.16 | 7.29 | 0.16 |
| Cistá | — | CR | Primary | 11 | 7.37 | 0.26 | 7.33 | 0.26 |
| Dolní Mesto | — | CR | Primary | 2 | 7.14 | 0.26 | 7.10 | 0.26 |
| Krátká | — | CR | Primary | 1 | 7.13 | 0.34 | 7.46 | 0.44 |
| Krucemburk | — | CR | Primary | 1 | 7.90 | 0.11 | 8.00 | 0.09 |
| Ondřejov | — | CR | Primary | 5 | 6.98 | 0.26 | 6.95 | 0.67 |
| Příbraz | — | CR | Primary | 7 | 7.04 | 0.47 | 7.17 | 0.46 |
| Rudíkov | — | CR | Primary | 5 | 7.86 | 0.27 | 7.69 | 0.25 |
| Spálené Pořící | — | CR | Primary | 13 | 7.45 | 0.74 | 7.39 | 0.75 |
| Zahrádky | — | CR | Primary | 2 | 7.30 | 0.20 | 7.70 | 0.34 |
| Žitenice | — | CR | Primary | 7 | 7.59 | 0.31 | 7.43 | 0.29 |
| **Mean** | | | | | **7.41** | | **7.43** | |
| **Standard Deviation** | | | | | **0.31** | | **0.30** | |

*Note:* U.K. site names are approximate. STE = septic tank effluent.

reed beds had outlet pH = 7.33 ± 0.32, measured over time periods of one to nine years. However, 18 other HSSF systems located in Norway, Australia, New Zealand, and the United States had similar outlet pH = 7.12 ± 0.20, measured over time periods of one to five years. Thus, it is possible to generalize, and to expect SSF effluent pH to be just above neutrality. Also, results from the Czech Republic (Table 5.7) indicated literally no change of pH after passage through the HSSF wetlands. The average inflow and outflow pH values from the 12 systems were 7.41 ± 0.31 and 7.43 ± 0.30, respectively.

When HSSF wetlands follow a lagoon in a treatment train, algal activity in the pond often creates elevated pH entering the wetland. This may be seen for the Benton, Louisiana, system in Table 5.7. The pH modification in the wetland most likely was due to the interactions between the substrate and its biofilms, rather than to the macrophytes. Data from Richmond, New South Wales, Australia (Bavor *et al.*, 1988), and from Minoa, New York (Theis and Young, 2000), support this idea, since unplanted gravel beds produced the same pH as planted systems.

Much the same conclusion may be reached for VF wetlands, which also display circumneutral pH and little or no pH change throughout the wetland (Table 5.8).

## Wetlands Treating Acid Mine Drainage

There are a number of variants of constructed wetlands that target acid mine drainage, with the purpose of reducing

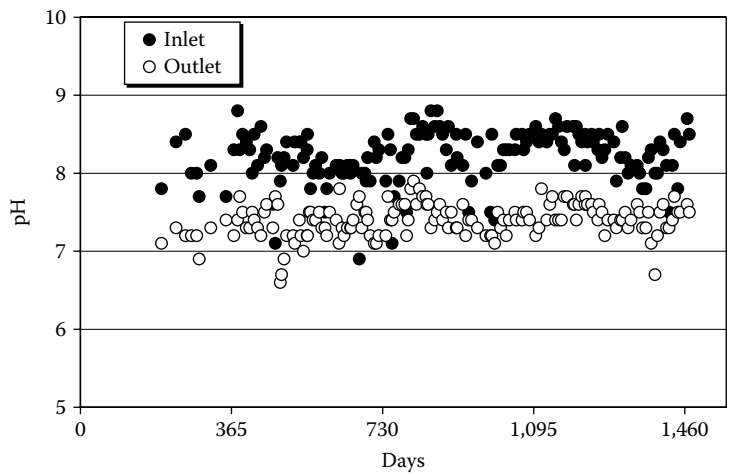



**FIGURE 5.20** Inlet and outlet pH for a SSF wetland in Yorkshire, England. (Data from CWA database (2006) *Constructed Wetlands Interactive Database*, *Version 9.02*. Compiled by G.D. Job and P.F. Cooper. United Kingdom Constructed Wetland Association (CWA): Gloucestershire, United Kingdom.)

**TABLE 5.8**
**Hydrogen Ion in Waters Entering and Leaving Example VF Wetlands**

| Country | Site Name | pH In | pH Out |
|---|---|---|---|
| United Kingdom | Londonderry, Northern Ireland (gravel bed) | 7.05 | 6.95 |
| United Kingdom | Londonderry, Northern Ireland (peat bed) | 7.03 | 6.82 |
| United Kingdom | Staffordshire 1, England (1st stage vertical flow) | 7.59 | 7.93 |
| United Kingdom | Staffordshire 1, England (2nd stage vertical flow) | 7.93 | 7.97 |
| United Kingdom | Staffordshire 2, England (1st stage vertical flow) | 7.39 | 7.81 |
| United Kingdom | Staffordshire 2, England (2nd stage vertical flow) | 7.83 | 7.81 |
| United Kingdom | Somerset, England | 7.70 | 7.70 |
| Netherlands | Hobbitstee (Wapserveen, Netherlands) | 7.47 | 6.88 |
| Netherlands | Spijkerman (Wapserveen, Netherlands) | 6.86 | 6.83 |
| Netherlands | van Ravenhorst (Woudenberg, Netherlands) | 6.59 | 6.91 |
| Netherlands | van Oirschot (Boxtel, Netherlands) | 7.99 | 7.23 |
| Netherlands | Adema (Lemmer, Netherlands) | 7.31 | 7.15 |
| Netherlands | Klein Profijt (Oud Biejerland, Netherlands) | 7.76 | 7.45 |
| Netherlands | Nooyen (Deurne, Netherlands) | 6.79 | 6.80 |
| **Mean** | | **7.30** | **7.19** |
| **Standard Deviation** | | **0.45** | **0.46** |

*Note:* U.K. site names are approximate.

*Source:* Data from CWA database (2006) *Constructed Wetlands Interactive Database*, *Version 9.02*. Compiled by G.D. Job and P.F. Cooper. United Kingdom Constructed Wetland Association (CWA): Gloucestershire, United Kingdom; and ECOFYT (F. van Dien).

metal content and improving (raising) the pH. However, such wetland systems typically do not change pH very much. For instance, Wieder (1989) surveyed 128 constructed wetlands, and found a mean inlet pH = 3.61 and a mean outlet pH = 3.72. However, it must be remembered that pH is the negative logarithm of concentration, and thus this small increase in pH corresponded to a median reduction in hydrogen ion concentration of 68% (Wieder, 1989). More intensive individual studies corroborate this finding. For instance, Mitsch and Wise (1998) found an inlet pH of 2.82 was increased to only pH 3.34 by passage through a FWS treatment wetland complex. Likewise, Wieder (1992) found an average increase from 2.89 to 3.08 in five organic-substrate wetlands. The frequency distribution of results of a number of other acid mine wetland studies are shown in Figure 5.21. It is clear from this body of knowledge that constructed wetlands do not provide a mechanism for adjustment of strongly acidic water conditions.

## Substrate Effects

The selected substrate for both FWS and SSF wetlands can have an effect on the pH of the water, at least for a period of

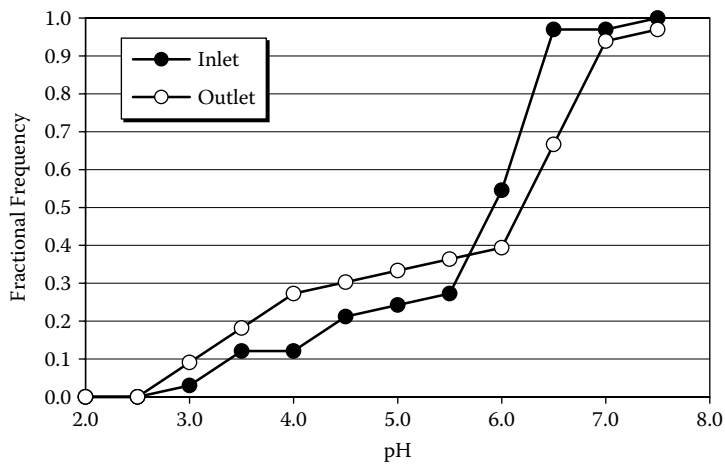



**FIGURE 5.21** Distribution of pH in the inlet and outlet of acid mine drainage wetlands. (Data from 33 wetlands are represented from Brodie (1990) *Constructed wetlands for treating acid drainage at Tennessee Valley Authority Coal Facilities.* Cooper and Findlater (Eds.). Proceedings of the International Conference on the Use of Constructed Wetlands for Water Pollution Control, 24–28 September 1990, Pergamon Press: Cambridge, United Kingdom, pp. 461–470; and from Younger *et al.* (2002) *Mine Water: Hydrology, Pollution, Remediation.* Kluwer Academic Publishers, London, United Kingdom.)

time prior to alteration of the substrate. The use of alkaline ash as the substrate for a FWS wetland was investigated by Ahn *et al*. (2001). The parent ash material had pH = 10.6, and was used as a liner. First-year water pH was somewhat elevated, but after two years little or no pH effect of the liner could be found.

Mesocosm studies on the use of shale as a medium to enhance phosphorus removal in SSF wetlands were conducted by Drizo *et al*. (1997). The pH of the incoming water was approximately 7.0, but outlet waters began at pH = 4.0, and slowly rose to 6.0 over the span of 250 days of operation.

The use of zeolites or expanded clays as a medium in SSF systems poses the opposite problem: the media may be very alkaline, and create high pH in the waters being treated. For instance, Zhu *et al*. (1997) tested a variety of light weight aggregates, again for purposes of phosphorus removal. Ten different varieties of expanded clays had pH = 9.78 ± 0.53. Similarly, a number of sands tested by Brix *et al*. (2001) showed pH = 8.39 ± 0.18 (N = 13). These substrate effects may be transitory, and of small consequence in passive systems, but could be important in wetlands in which media replacement is a design intent.

## 5.6 ALKALINITY AND ACIDITY

Hydrogen ion content is but one possible measure of the ionic condition of a particular water. A broader concept is that of alkalinity, defined as the net concentration of strong base in excess of strong acid (Morel and Hering, 1993). Operationally, a sample is titrated with strong acid (hydrochloric or sulfuric) to an endpoint of about pH = 4.5 (APHA, 1992). In "pure" waters, the base requiring neutralization is present because of dissolved carbon dioxide and its equilibrium dissociation products:

$$\begin{aligned} CO_{2(air)} &\rightleftharpoons CO_{2(dissolved)} + H_2O \rightleftharpoons H_2CO_3 \\ &\rightleftharpoons H^+ + H_2CO_3^- \rightleftharpoons H^+ + CO_3^{2-} \end{aligned} \quad (5.22)$$

The equilibria associated with these chemical conversions are discussed in more detail in Chapter 8. The distribution of chemical species at 25°C is shown as a function of pH in Figure 8.1. The sum of all carbonate species is dissolved inorganic carbon (DIC), which is the source of energy for autotrophic microorganisms. In this simplistic pure water context, alkalinity is defined as (Pankow, 1991):

$$\text{Alkalinity} = [HCO_3^-] + 2\left[CO_3^{2-}\right] + [OH^-] - [H^+] \quad (5.23)$$

Conversion factors for the computation of DIC from pH, temperature, and alkalinity are given in Wetzel and Likens (1991). The resulting alkalinity of pure water at 25°C and pH = 8 is 50 mg/L as calcium carbonate.

Treatment wetlands are not pure waters, and the other dissolved constituents can contribute to the amount of titrating acid needed. The definition of alkalinity must then be expanded, for example to include other common components of waters to be treated (Morel and Hering, 1993):

$$\begin{aligned} \text{Alkalinity} = {} & [HCO_3^-] + 2\left[CO_3^{2-}\right] + [OH^-] \\ & - [H^+] + [NH_3] - [H_2S] + [S^{2-}] \end{aligned} \quad (5.24)$$

Phosphates, borates, and silicates may also contribute.

As seen in the preceding section, the water may be acidic rather than basic, thus requiring titration with a strong base such as sodium hydroxide. The same concepts still apply, but alkalinity is replaced by acidity (Morel and Hering, 1993):

$$\text{Acidity} = -\text{Alkalinity} \quad (5.25)$$

The concept of acidity applies particularly to wetlands treating acid mine drainage (Younger *et al*., 2002).

### Alkalinity in Treatment Wetlands

Examples of alkalinity entering and leaving treatment wetlands are given in Table 5.9. In general, total alkalinity ranges upward from the values expected for pure water (approximately 100 mg/L), to much higher values for landfill leachates (>400 mg/L). FWS wetlands typically reduce alkalinity by a small margin. Conversely, HSSF systems cause a slight increase.

There are no seasonal trends of consequence. For example, regressions of total alkalinity against yearday produced essentially flat lines, with $R^2 = 0.000$ at Musselwhite, Ontario, 0.097 at Estevan Saskatchewan, and 0.020 and 0.037 at Tres Rios, Arizona H1 and H2.

### Carbonates in Treatment Wetlands

Solid calcium carbonate, in the form of the minerals calcite and aragonite (both $CaCO_3$), may be important in the function of some treatment wetlands. In lakes, macrophytic vegetation of the littoral zone may become encrusted with massive deposits of $CaCO_3$, formed by the photosynthetic utilization of $CO_2$ (Wetzel, 1983). Blue-green algae growing attached to substrates also produce large deposits of carbonates (Wetzel, 1983). The precipitation of $CaCO_3$ is extremely sensitive to pH, because of the dependence of dissolved carbonate on pH. For instance, in the pure water situation, with $CO_2$ controlled by atmospheric equilibrium to pH = 8.3, the solubility of $CaCO_3$ is 20 mg/L calcium (Morel and Hering, 1993).

The extraction of $CO_2$ from an algal growth system through assimilation into algal biomass at a rate faster than it can be replaced through atmospheric $CO_2$ diffusion, respiration, fermentation processes, and readjustment of carbonate equilibria leads to an increase in pH level (Figure 5.17), and perhaps resulting in the precipitation of carbonates in lake environments (Wetzel, 1983).

In the wetland environment, the utilization of carbon dioxide by plants and algae also may drive the pH to high levels. The elevation in pH that results from intense SAV and periphyton photosynthesis can lead to $CaCO_3$ supersaturation, which in turn may facilitate precipitation of calcitic material. In a SAV community, submerged leaves may provide nucleating sites for $CaCO_3$ crystallization due to the very high pH levels that can occur at the leaf surface–water interface, and therefore may be important locations for encrustation with calcite.

**TABLE 5.9**
**Total Alkalinity Entering and Leaving Example Treatment Wetlands**

| | | | | | Total Alkalinity | | pH | |
|---|---|---|---|---|---|---|---|---|
| **System** | **Location** | **Wetland** | **WW Type** | **Years** | **Inlet** | **Outlet** | **Inlet** | **Outlet** |
| **FWS** | | | | | | | | |
| Byron Bay | Australia | — | Activated Sludge | 1990–1993 | 70 | 53 | 6.95 | 6.92 |
| Orlando Easterly | Florida | WP1-MM7 | Tertiary | 1993–2002 | 94 | 101 | 7.11 | 6.97 |
| Musselwhite | Ontario | — | Mine | 1997–2002 | 133 | 87 | 7.90 | 6.97 |
| Tucush | Peru | — | Mine | 2006 | 158 | 148 | 7.51 | 7.95 |
| Tres Rios | Arizona | H1 | Nitrified 2° | 1995–2002 | 174 | 174 | 6.94 | 7.14 |
| Tres Rios | Arizona | H2 | Nitrified 2° | 1995–2002 | 174 | 181 | 6.95 | 7.22 |
| Tres Rios | Arizona | C1 | Nitrified 2° | 1995–2002 | 176 | 185 | 6.92 | 7.36 |
| Tres Rios | Arizona | C2 | Nitrified 2° | 1995–2002 | 177 | 184 | 6.92 | 7.23 |
| Imperial | California | — | Ag Runoff | 2001–2005 | 242 | 210 | 7.57 | 7.63 |
| Brawley | California | — | Ag Runoff | 2001–2005 | 278 | 247 | 7.35 | 7.76 |
| Estevan | Saskatchewan | — | Lagoon | 1994–2003 | 346 | 372 | 8.15 | 8.70 |
| Champion | Florida | A & B | Lagoon | 1991–1993 | 418 | 373 | 7.80 | 7.90 |
| Champion | Florida | C & D | Lagoon | 1991–1993 | 418 | 390 | 7.80 | 7.80 |
| Isanti-Chisago | Minnesota | — | Leachate | 1996–2000 | 480 | 415 | 7.91 | 7.67 |
| New Hanover | North Carolina | — | Leachate | 1997 | 889 | 536 | 7.85 | 7.84 |
| **Mean (FWS)** | | | | | **282** | **244** | **7.44** | **7.54** |
| **HSSF** | | | | | | | | |
| Benton | Kentucky | 3 | Lagoon | 1988–1989 | 100 | 183 | 7.20 | 7.00 |
| Hardin | Kentucky | 1 | Package | 1988–1989 | 126 | 179 | 7.00 | 7.10 |
| Hardin | Kentucky | 2 | Package | 1988–1989 | 126 | 164 | 7.00 | 7.20 |
| Grand Lake | Minnesota | — | Septic Tank Effluent | 1995–1998 | 383 | 393 | 7.23 | 7.12 |
| NERCC | Minnesota | 1 | Septic Tank Effluent | 1996–1999 | 466 | 479 | 7.23 | 7.07 |
| NERCC | Minnesota | 2 | Septic Tank Effluent | 1996–1999 | 466 | 488 | 7.23 | 7.08 |
| **Mean (HSSF)** | | | | | **278** | **314** | **7.15** | **7.10** |

This carbonate chemistry has extremely important ramifications in the south Florida environment. Carbonates dominate the substrates of some natural Everglades systems, notably the marl prairies (Gleason and Stone, 1994) (Figure 5.22). Phosphorus coprecipitates with the calcitic solids, and therefore there have been attempts to emulate the natural system with constructed wetlands designed to remove phosphorus. There are two principal variants on the theme: systems that maximize algal components and the availability of calcium substrates (periphyton systems), and those that maximize submerged plant surfaces and their photosynthesis (SAV systems). Periphyton system data has been



**FIGURE 5.22** Calcium carbonate deposits supported by the canopy of sawgrass (*Cladium jamaicense*) in the marl prairie of the Florida Everglades. These form as the result of calcitic periphyton mats that desiccate during dry periods.

reviewed and summarized in Kadlec and Walker (2004). The SAV systems use the structure of the underwater plants to support algal encrustations (Dierberg *et al*., 2002). Limerock components may be added to the SAV system to augment the calcium supply (DeBusk and Dierberg, 1999). At the time of this writing, large-scale (40 ha each) demonstration projects are underway at two locations. One site was prepared by scraping all soil from a limerock base. The second was prepared by incorporating calcareous material in the top layer of a sand bed material. No results are available at the time of this writing.

The purpose of treatment in acid mine wetlands is the removal of acidity, or equivalently, the production of alkalinity. The terms *reducing and alkalinity producing systems* (RAPS) and *successive alkalinity producing systems (*SAPS) have been coined to describe wetland systems designed for that purpose (Younger *et al*., 2002). These are beds of organic material overlying limestone rock beds, through which the water flows downward. The depth of standing water on top of these beds determines the type and vitality of the wetland plants. This concept is further discussed in Chapter 11 (see Figure 11.15).

## SUMMARY

Some natural wetlands operate with high levels of dissolved oxygen, others at low levels. Most treatment wetlands receive enough BOD and NOD loading to drive the DO level down to about 1–2 mg/L. Exceptions are oversized wetlands receiving very clean effluents. The soils, sediments, and biota in the wetland exert a strong influence on the DO concentrations in the water. Therefore, it is not accurate to assume that BOD and NOD disappearance is a measure of oxygen transfer.

In the case of FWS wetlands, most $O_2$ transfer is probably due to interfacial aeration and underwater photosynthetic production. In HSSF wetlands, interfacial aeration is significantly reduced because the water surface is not exposed above the bed media, and also because the bed media increases diffusional distances. Therefore it is not surprising that HSSF wetlands exhibit extremely low oxygen transfer rates, which can be exceeded by oxygen demands exerted by internal wetland processes.

Based on the current body of knowledge, there is little or no evidence that FWS and HSSF wetlands are inherently "aerobic" and will automatically produce high effluent DO levels. VF wetlands that rely on principles of unsaturated flow (see Chapter 2) and design variants of SSF wetlands such as fill-and-drain (tidal flow) and aerated systems may achieve this standard, at the cost of additional mechanical input (see Part II of this book).

There is a growing body of evidence that plants provide no significant aeration flux to the water or soil, in excess of their respiratory demands.

In sediments and submerged soils, redox potential ranges from around −400 mV (strongly reduced) to +700 mV (well oxidized) and is better poised and fairly reproducible at the more reduced levels. Redox potentials are strongly influenced by the influent carbonaceous and nitrogenous oxygen loadings, internal oxygen demands within the wetland, and the rate of oxygen transfer into the wetland.

Treatment wetlands typically operate at circumneutral pH for influents that are not strong acids or bases. This is true for both FWS and SSF constructed wetlands. One exception is for wetlands designed to treat acid mine drainage, where low influent pH levels are the norm. Some HSSF wetlands are designed with reactive medias in the bed material (zeolite, LECA, or blast furnace slag; discussed in further detail in Chapter 10) that can produce high pH effluents. Unless reactive medias are employed (or the influent is highly acidic), effluent DO, pH, redox, and alkalinity levels from treatment wetlands are typically driven by carbonate equilibrium chemistry dictated by the air–water interface.

# 6 Representing Treatment Performance

This chapter examines the available means of collecting and analyzing the large amount of performance data that now exists for treatment wetlands. Wetlands are "open" systems heavily influenced by environmental factors. This makes them more complex than other types of biological treatment reactors (activated sludge, trickling filters) described in the environmental engineering literature. Nevertheless, attempts have been made to adapt models from these other technologies to treatment wetlands (Burgoon *et al.*, 1999; McBride and Tanner, 2000; Langergraber, 2001; Rousseau *et al.*, 2005b; Wu and Huang, 2006). Wetlands are dominated by biomass storage compartments that are very large relative to pollutant mass in the water column (again, different than other biological reactors). These biomass storage compartments are affected by seasonal cycles that are different than temperature cycles.

Treatment performance is represented by two components: the central treatment tendency for a wetland (or group of wetlands) and the anticipated variability away from that central tendency. Central tendencies are driven by flows and concentrations, in concert with environmental factors. Random events within the wetland will produce stochastic variations in effluent performance. Both must be assessed to describe treatment performance in constructed wetlands.

Different types of wetlands (e.g., wetland configuration, vegetative community) function differently. Therefore, a set of "universal" parameters for describing treatment performance in wetlands is not to be expected.

## 6.1 VARIABILITY IN TREATMENT WETLANDS

Two types of variability are of interest for understanding and design of treatment wetlands. First, it is necessary to understand the scatter of performances for an individual wetland, around either the central tendency of data or the model characterization of that central tendency. This is the intrasystem, or internal variability, and it is needed to understand the excursions that may be expected, and to design to meet permit requirements that involve allowable maximums. Internal variability includes seasonal, stochastic, and year-to-year changes. Wetland performance can also change from year to year due to changes in vegetative communities, hydraulic or organic loadings, or weather conditions. Second, it is useful to understand how comparable wetlands vary, which is the intersystem variability. Causes of this variation will include factors such as vegetation species, system geometry, and climatic conditions. Both types of variability are best explored by graphical methods.

### Intrasystem Variability

Data frequency influences the degree of scatter in data. Variability decreases daily–weekly–monthly–annual, but the central tendency is the same. For example, the coefficients of variation for total phosphorus over four years at Brighton, Ontario, were weekly = 89%, monthly = 83%, and annual = 19%.

Many factors contribute to random variability in the outlet concentrations from a single treatment wetland. This variability is typically not small, with coefficients of variation of 20%–60% being common. Deterministic models reproduce the central tendency of performance, but not the random variability. Whether there is microbial or vegetative control, seasonal patterns of wetland variables are the rule, accompanied by a random variable term (Kadlec, 1999a).

### Data Folding

A choice may be made to either deal with "raw" data or detrend a concentration time series using either a mechanistic model or a cyclic annual trend. Most of the existing treatment wetland literature considers the probability distributions of the raw data for concentration time series. The typical method is to present the cumulative probability distributions for concentrations entering and leaving the wetland (see, e.g., Kadlec and Knight, 1996; U.S. EPA, 1999). Typical probability distributions are shown for weekly average for data Columbia, Missouri (Figure 6.1). The median inlet BOD = 26 mg/L in 1995, while the median outlet BOD = 9 mg/L. However, inlet concentrations ranged from 8 to 60 mg/L, and outlets from 4 to 24 mg/L. At the weekly time scale, the maximum BOD exiting the wetland was 2.7 times the median. The data in this BOD example are not detrended.

Seasonal changes in treatment performance can often be represented by cosine trends (Kadlec, 1999a).

Stochastic variability will report as a "cloud" around the seasonal trend line:

$$C = C_{avg}\left[1 + A \cdot \cos\left(\omega(t - t_{max})\right)\right] + E \qquad (6.1)$$

where

$A$ = fractional amplitude of the seasonal cycle, dimensionless
$C$ = instantaneous outlet concentration, mg/L
$C_{avg}$ = average (trend) outlet concentration, mg/L
$E$ = random portion of the outlet concentration, mg/L
$t$ = time of the year, Julian day
$t_{max}$ = time of the year for the maximum outlet concentration, Julian day

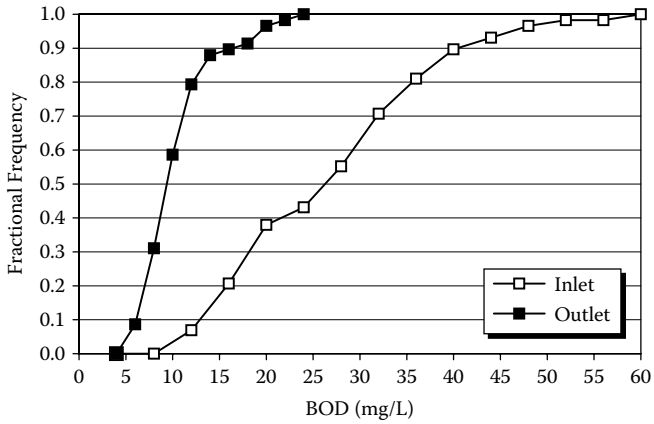



**FIGURE 6.1** Distribution of BOD concentrations measured at the Columbia, Missouri, FWS wetlands in 1995. (Unpublished data from city of Columbia.)

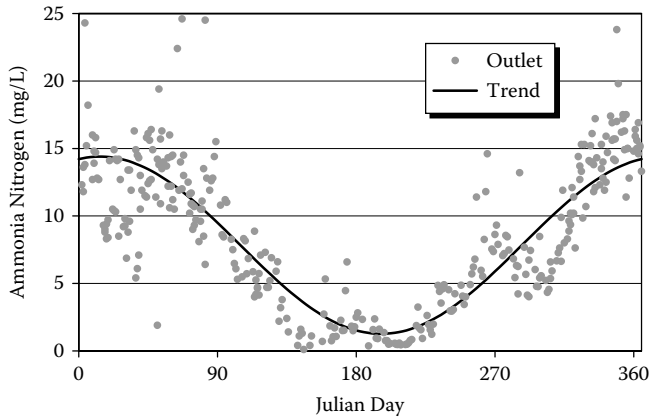



**FIGURE 6.2** Ammonia nitrogen concentrations leaving the Columbia, Missouri, FWS wetlands in 1995, together with the annual trend. Data were acquired daily on weekdays. (Unpublished data from city of Columbia.)

The deterministic portion of this representation may in turn be modeled by the *k*-rate technique with appropriate rate constants and background concentrations, both of which may respond to temperature and season, as will further be discussed.

The existence of the error term (*E*) means that sampling must either be at high frequency or cover many annual cycles before meaningful trend averages can be determined. Data from several years may be "folded" to create an annualized grouping, distributed across the year according to Julian day. This use of many annual cycles has the advantage of including year-to-year variations in climate, flow, and ecosystem condition.

The stochastic portion (*E*) will have a probability distribution, which will be different depending upon sampling frequency and sample averaging period. The ammonia concentration data for Columbia, Missouri, serve to illustrate that stochastic variability may be considered separately from annual trends. At that site and most other treatment wetlands, there is a strong annual cycle in ammonia, occasioned by the slow-down in treatment during the winter months, as well as by trends in the ammonia levels leaving pretreatment (Figure 6.2). For that FWS system, Equation 6.1 was calibrated to the data from 1994 to 1995 as follows:

$$\text{Inlet:} \quad C_{avg} = 10.0 \qquad A = 0.61 \qquad t_{max} = 19$$

$$\text{Outlet:} \quad C_{avg} = 7.8 \qquad A = 0.84 \qquad t_{max} = 14$$

The variability in the inlet and outlet concentrations may then be expressed as fractional departures from the trend values, which is the random variable denoted by *E*/*C* from Equation 6.1 The cumulative probability distributions for both inlet and outlet time series are similar (Figure 6.3).

### Intersystem Variability

Apart from the concept of how one wetland may vary in its performance, there is the issue of how the parameters of the deterministic portion of the wetland performance model change from system to system. Typically, the difference in treatment performance between wetland systems is much greater than the difference in performance within a particular wetland system. There are several ways to express this variability, including:

- Side-by-side comparisons of wetlands with different attributes, such as type, or presence, or absence of vegetation
- Distributions of model parameter values, such as *k*-values, across a large number of comparable wetlands
- Graphical performance comparisons for sets of wetlands, based upon some period of performance such as annual or entire period of data record

The key to assigning differences to "variability" is the process of accounting for the principal factors affecting performance

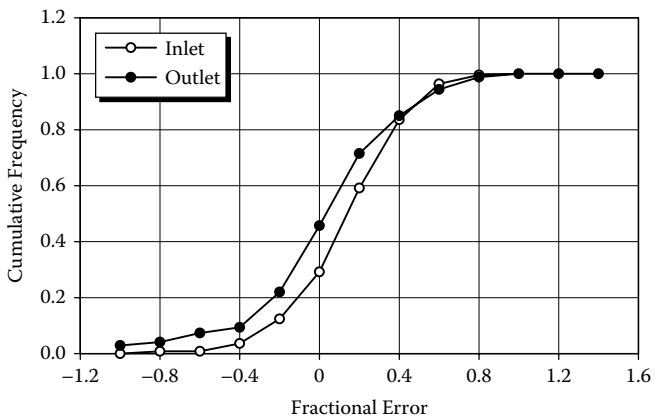



**FIGURE 6.3** Distribution of ammonia concentration fractional departures from annual trends measured at the Columbia, Missouri, FWS wetlands in 1995. Derived from the data in Figure 6.2.

separately and in advance of comparison. For example, the methods for describing effects of detention time or hydraulic loading, inlet concentration, temperature, and season will be discussed in the following text. It is clear that it is not useful to compare the summer behavior of one wetland to the winter behavior of another, because we have already identified the potential for seasonal and temperature differences. A choice that minimizes seasonal effects is the annual averaging period, which retains climatological effects, such as mean annual temperature and rainfall.

### Replication

Two wetlands of the same size and type should be expected to perform similarly if they receive identical water flows and concentrations. This has generally been observed to be the case in the few side-by-side studies that have involved such replication (see, e.g., Moore *et al.*, 1994; CH2M Hill and Payne Engineering, 1997; CH2M Hill, 1998; Mitsch *et al.*, 2004). Typical effluent concentration patterns follow similar time series, with occasional differences of unknown causes (Figure 6.4). Because of the expense of building and monitoring replicated wetlands, most of the comparative studies of treatment wetlands have not involved replication; this is apparently a justifiable step.

### Side-by-Side Studies

There have been numerous side-by-side studies conducted to elucidate possible effects of vegetation type, media size, aspect ratio, and other factors. In general, such studies have not involved replication, as noted in the previous text. In these studies, the incoming water chemistry and often the inlet flow rates are the same. Climatological effects, such as rainfall and air temperature, are identical for the comparison systems. The results of side-by-side testing determine the effect of the tested variable, but only for the specific circumstances of test wetland systems. For instance, Wolverton *et al.* (1983) bench-tested *Phragmites* and bulrushes (*Schoenoplectus* (*Scirpus*) spp.) in HSSF wetland microcosms and determined a significantly better performance for *Phragmites.* On the other hand, Gersberg *et al.* (1986) tested *Phragmites* and bulrushes (*Schoenoplectus* (*Scirpus*) spp.) in outdoor pilot HSSF wetland environments at Santee, California, and determined a significantly better performance for *Schoenoplectus.* However, when the same plants were tested in a full-scale HSSF facility at Minoa, New York (Liebowitz *et al.*, 2000), essentially no difference was found for COD and other parameters (Figure 6.5). These analyses emphasize the need for great care in detailing the circumstances of side-by-side studies. Further extrapolations to other situations may be very misleading, however similar the circumstances may be.

### Aggregated Data Sets

Combining performance data from different wetland systems to create an aggregated data set results in data clouds that have considerably more variability than the individual wetland data sets they were created from. These aggregated data sets are useful for exploring the bounds of treatment performance in a particular application, but may not accurately predict the performance of an individual treatment wetland.

Aggregated data sets can be used to define the central tendency in treatment performance for a given type of wetland reactor and application (e.g., BOD removal in HSSF wetlands). However, use of the central tendency to create a "rule of thumb" is only one piece of the description of treatment performance. Because of the loss of specificity and high variance in these aggregated data sets, statistics such as confidence intervals and effluent multipliers have to be developed to assess short-term variances that may be important for risk assessment.

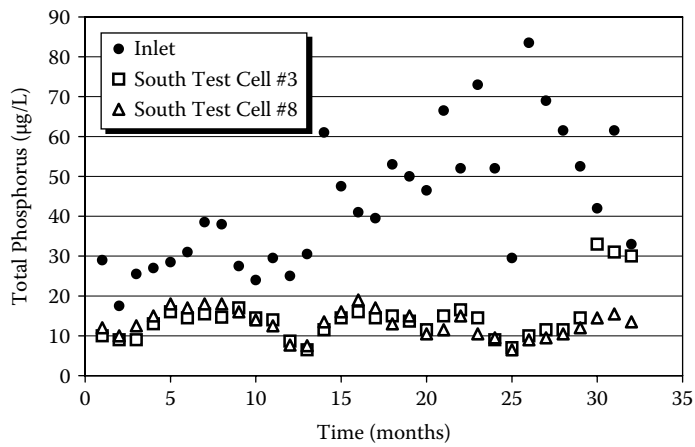



**FIGURE 6.4** Performance of two FWS wetland replicates for phosphorus reduction at low concentrations. These behave similarly over most of the period of record. The reason for departure during the last three months of the record is not known. (Unpublished data from South Florida Water Management District.)

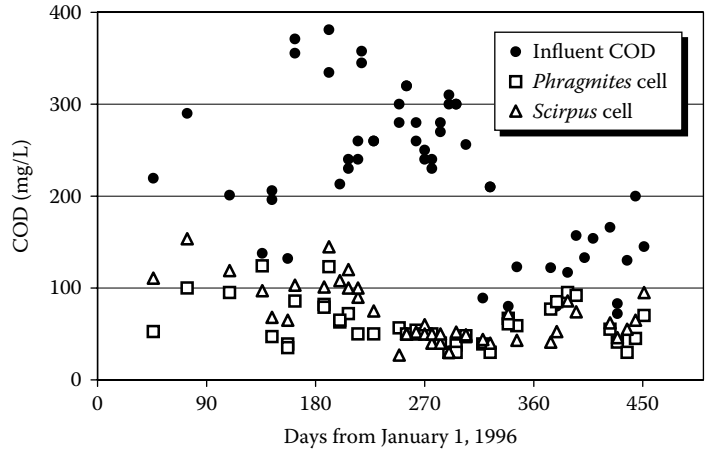



**FIGURE 6.5** Performance of side-by-side wetlands at Minoa, New York, vegetated with *Phragmites* spp. and *Scirpus* (*Schoenoplectus*) spp. (Data from Theis and Young (2000) *Subsurface flow wetland for wastewater treatment at Minoa*. Final Report to the New York State Energy Research and Development Authority, Albany, New York.)

## 6.2 GRAPHICAL REPRESENTATIONS OF TREATMENT PERFORMANCE

There exist a large number of data sets for some of the more common pollutants, such as TSS, BOD, phosphorus, and nitrogen species. Several types of graphs may be used to compare performances across systems, and these have been used in prior treatment wetland literature:

1. Output concentration versus input concentration
2. Output concentration versus input areal loading
3. Output loading versus input loading
4. Load removed versus input areal loading
5. Rate constant versus input areal loading

The first two of these are useful representations, but the last three very often lead to spurious relationships that serve no useful purpose. Many important variables are lost in these plots, because of their restrictive 2-D nature.

### Outputs Versus Inputs

The input–output concentration graph essentially extends the idea of percent removal to a group of wetlands. That is useful in obtaining first estimates of the potential of a class of treatment wetlands to reduce a particular contaminant. But, that plot is of no value in sizing the wetland, because it does not contain any information on the detention time or hydraulic loading.

The phosphorus concentration produced in treatment wetlands depends upon three primary variables (area, water flow, and inlet concentration), as well as numerous secondary variables (vegetation type, internal hydraulics, depth, event patterns, and others). It is presumed that the area effect may be combined with flow as the hydraulic loading rate ($q$ = HLR) since two side-by-side wetlands with double the flow should produce the same result as one wetland. Therefore, two primary variables are often considered: HLR and inlet concentration ($C_i$). Older kinetic removal models (e.g., the $k$-$C^*$ model) and performance regressions are based upon these two variables (Kadlec and Knight, 1996).

Later in this chapter, it will be shown that wetland outlet concentrations are often well represented by:

$$\frac{C_o - C^*}{C_i - C^*} = \frac{1}{(1 + k/Pq)^P} \tag{6.2}$$

where

$C_o$ = outlet concentration, mg/L
$C_i$ = inlet concentration, mg/L
$C^*$ = background concentration, mg/L
$k$ = modified first order areal constant, m/d
$P$ = apparent number of TIS
$q$ = hydraulic loading rate, m/d

Here this model is used to explore the expected corresponding appearance of intersystem performance graphs.

An equivalent approach is to rearrange the primary variables, without loss of generality, by using inlet loading rate (LRI = $q \cdot C_i$) and concentration ($C_i$). Thus, it is expected that the effluent concentration produced ($C_o$) will depend upon LRI and $C_i$. A graphical display has often been adopted in the literature (Kadlec and Knight, 1996; U.S. EPA, 2000a). In the broad context, multiple data sets are represented by trends that show decreasing $C_o$ with decreasing LRI, with a different trend line associated with each inlet concentration (Figure 6.6). For low inlet concentration or for higher hydraulic loadings, the log–log slope of the data cloud is approximately 0.33 (Figure 6.6), but the resultant outlet concentration range moves upward to higher values. The right-hand asymptote of each data group, at very high pollutant loading, is an outlet concentration equal to the inlet concentration—or in other words, no removal. The left-hand asymptote, reached only for low inlet concentrations, is the background concentration, $C^*$. The fact that there exist data clusters for each inlet range indicates that there are at least two major factors influencing outlet concentration: inlet concentration and inlet loading.

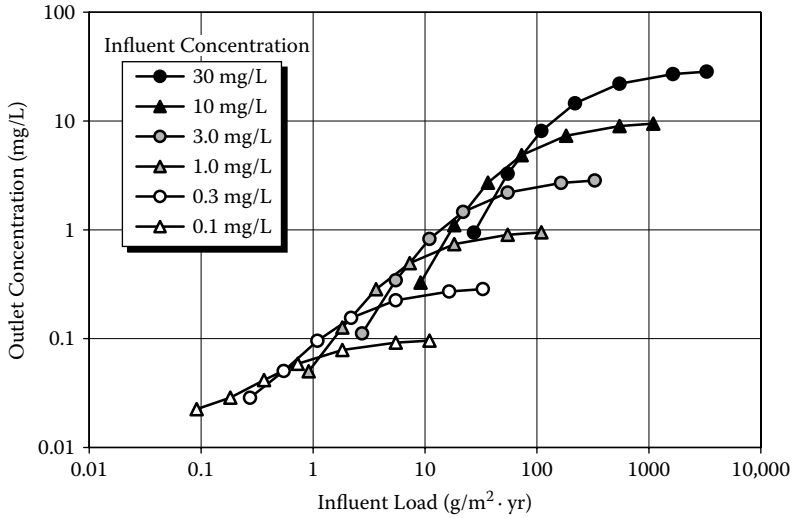



**FIGURE 6.6** Hypothetical concentration load response for the *P-k-C** model, with $P = 3$, $k = 6$ m/yr, and $C^* = 0.02$ mg/L. The lines are for different values of influent concentration, as indicated in the legend. On each line, the hydraulic loadings are from left to right: 0.25, 0.50, 1.0, 2.0, 5.0, 15.0, and 30.0 cm/d.

If the entire set of points in Figure 6.6 is considered, ignoring the effect of inlet concentration, the general trend line has a log–log slope of about 1.0. However, such a single variable plot is nonunique, because of the effect of inlet concentration, and may be misleading. For instance, use of a small intersystem data set might result in use of left data points for high $C_i$, as well as right data points for low $C_i$, thus exaggerating the slope. Consequently, the $C_o$– LRI graph advocated in some literature (U.S. EPA, 2000a) is inadequate. The *P-k-C** model typically spans the entire cloud of intersystem results when exercised for various choices of $C_i$, $k$, and $C^*$ (Kadlec, 1999c). It is expected that real data would display behavior like that in Figure 6.6, and that expectation is found to be realized in later chapters concerning individual contaminants.

The outlet concentration load graph is a useful addition to the design sizing toolkit for treatment wetlands. However, it cannot be used in isolation as a design sizing basis, because it does not separate the individual effects of inlet concentration and hydraulic loading. Inspection of Figure 6.6 shows that the inlet loading is not a unique design variable, and that the hydraulic loading and inlet concentration that define it are not interchangeable. Part II of this book discusses the use of a concentration-loading graph as an important component of the design process.

## Perspectives Derived from the Loading Graph

The principal tool or examination of intersystem variability in this book will be the outlet concentration versus inlet loading graph. The period of data averaging involved for comparison purposes should be long enough to encompass as much as possible of the intrasystem or internal variability, so as to focus on system differences. The operations of the systems being compared should be past start-up, so that sustainable performance can be analyzed. A subtle paradox occurs due to the fact that periods of record will not typically be equivalent among comparison wetlands, except in side-by-side studies. Suppose Wetland *A* has two years and Wetland *B* has ten years, respectively, of data past start-up. Neither Wetland *A* nor Wetland *B* will necessarily operate or perform in the same way from year to year, so the choice of annual averaging will produce two distinct data points for Wetland *A* and ten for Wetland *B*. There will be interannual variability represented for each, which will, to some extent, obfuscate the comparison between Wetlands *A* and *B*. Thus there are two logical choices: the use of interannual, intersystem information, involving one point for each year for each wetland; and the use of period of record (POR), intersystem data, involving one point for each wetland.

These concepts are illustrated in Figure 6.7 for phosphorus reduction for two similar wetlands treating facultative lagoon effluents. Brighton provides some phosphorus removal via alum pretreatment, with a long-term mean influent of 0.45 mg/L. In contrast, the inlet to the Estevan (Saskatchewan) wetlands was 2.26 mg/L. Removal was 24% at Estevan, at an average hydraulic loading of 2.6 cm/d over a nine-year period of record past start-up. Removal was 40% at Brighton, at an average hydraulic loading of 5.1 cm/d over a 4.25-year period of record (POR) past start-up. Data are shown as monthly, annual, and period of record averages of weekly measurements. The monthly data scatter is in part due to seasonal differences, which spanned May through November for Estevan, and all 12 months for Brighton. This seasonal effect is removed by annual averaging, which causes only interannual and intersystem effects to remain. Finally, interannual effects are removed by constructing the period of record averages, involving four years for Brighton and nine years for Estevan.

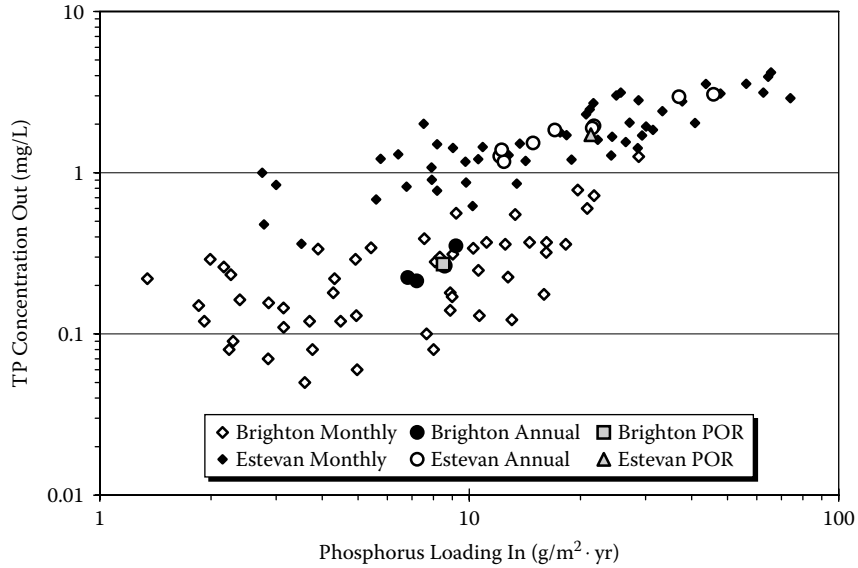



**FIGURE 6.7** Outlet TP concentration versus inlet TP loading for Estevan, Saskatchewan, and Brighton, Ontario, treatment wetland systems. The period of record past start-up was 4.25 years for Brighton, and nine years for Estevan. (Unpublished data from city of Estevan and city of Brighton.)

The reasons for the differences between these two systems cannot be determined from the graphical representation. However, as shall be seen in Chapter 10, much of the difference is attributable to the nonuniqueness of the phosphorus-loading variable, meaning that the difference in inlet concentrations places the two systems in different groupings.

It is also possible to look further via the *P-k-C** model. There are no tracer tests of either wetland, so it will be presumed that $N = P = 4$. It is known that $C^*$ is quite low for phosphorus, and it will be presumed that $C^* = 0.01$ mg/L. The POR data then indicate an annual $k = 11$ m/yr for Brighton, and $k = 3$ m/yr for Estevan.

## Pitfalls of Graphical Representations

The purpose here is to illustrate the fallacy of graphical data representations and associated regressions between variables that contain the same multiplier and the errors that accompany an incorrect model choice. This subject has been elucidated for natural treatment systems by Von Sperling (1999). As a hypothetical example, consider concentrations entering the wetland vary randomly between 0.2 and 1.2 g/m$^3$. Likewise, the concentrations leaving are also random between 0.1 and 0.3 g/m$^3$. Therefore, the mean inlet concentration is 0.7 g/m$^3$, the mean outlet concentration is 0.2 g/m$^3$, and the resulting average concentration reduction is 71%.

A set of 50 experiments is run, in which the hydraulic loading is varied linearly between 1 and 50 m/yr. For any experiment, the inlet and outlet concentrations are independently random within the ranges selected (Figure 6.8). Not surprisingly, linear regression of the input/output concentrations explains virtually none of the variability There is a 72 ± 18% (mean ± SD) concentration reduction, and that is all that may be determined.

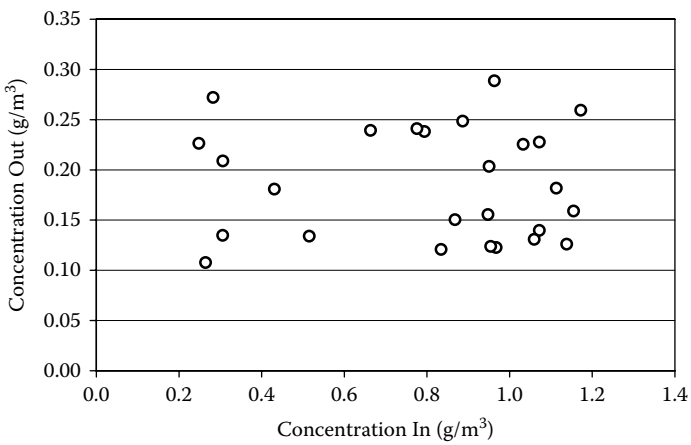



**FIGURE 6.8** Scatter plot of input and output concentrations for a hypothetical data set for 71% reduction.

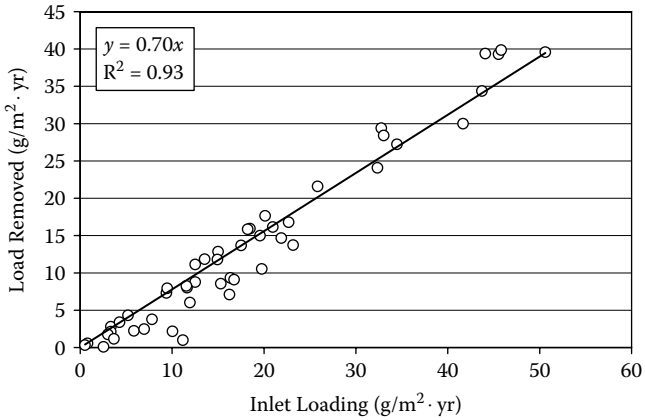



**FIGURE 6.9** Load reduction versus incoming load for the hypothetical, random data set of Figure 6.8.

Next, the correlation between pollutant load reduction and inlet pollutant loading is examined. Pollutant loading is defined as hydraulic loading multiplied by concentration, for both the inlet and outlet. Pollutant load reduction is the difference between inlet and outlet pollutant loadings. A wonderful correlation is obtained with an $R^2 = 0.93$, which makes the data look great and makes us feel that we can use this for design (Figure 6.9). Unfortunately, there is no connection of performance to inlet loading, no matter how much this load graph appeals to us. The hydraulic loading appears in both the ordinate and the abcissa, thus causing a stretching of a random 2-D cloud along a diagonal axis. The only useful feature of the graph is the slope of the line, 0.70, which is the correct result for the percent reduction. Many examples of this representation and analysis are to be found in the treatment wetland literature (Knight *et al.*, 1993; Hammer and Knight, 1994; Vymazal, 2001), but they are of virtually no value in design.

The formerly popular first-order plug-flow model is then examined. The same hypothetical random data set is easily manipulated to calculate a *k*-value for each pair of input–output concentrations, or to provide a least-squares estimate that best fits the entire data set, according to:

$$k = q \cdot \ln\left(\frac{C_i}{C_o}\right) \tag{6.3}$$

The *k*-values so calculated average 32 m/yr. The important question is whether this model fits the data, so that it may be used for predictions at specified hydraulic loading rates. The answer is that the first-order model fails and predicted concentrations scatter randomly with respect to observed concentrations.

The subtle trap that has created trouble, in this example and in some of the existing treatment wetland literature, is the failure to check whether or not the model has any validity. That can be done in a number of ways, but the easiest method is the direct examination of the data trends expected from the model. For the simple first-order case, the fraction of pollutant remaining is expected to decline exponentially with detention time, or equivalently with the inverse of hydraulic loading, as indicated by Equation 6.3. In the present hypothetical example, log-linear regression of data in this manner has an $R^2 = 0.000$.

## 6.3 MASS BALANCES

There are many measures and models for pollutant reductions in treatment wetlands. In this chapter, various definitions and options for system description are explored as a necessary precursor to the discussions of individual pollutants that follow in ensuing chapters.

### Concentrations

Individual concentration measurements are very often averaged to eliminate some of the variability inherent in wetlands. The time average concentration, denoted by an overbar ($\bar{C}$), is defined as

$$\bar{C} = \frac{1}{t_m}\int_0^{t_m} C\,dt \tag{6.4}$$

where

$C$ = chemical concentration, mg/L
$t$ = time, d
$t_m$ = averaging period, d

Such average concentrations may be acquired from time-proportional autosamplers, or computed from a time series.

An average mass flow of a chemical ($\overline{QC}$) is the product of the average flow and the flow-weighted (or mass average) concentration, defined by:

$$\hat{C} = \frac{\frac{1}{t_m}\int_0^{t_m} QC\,dt}{\frac{1}{t_m}\int_0^{t_m} Q\,dt} = \frac{1}{\bar{Q}t_m}\int_0^{t_m} QC\,dt \tag{6.5}$$

where the "hat" notation indicates a flow-weighted average.

$$\overline{QC} = \bar{Q}\,\hat{C} \tag{6.6}$$

Percent concentration reduction is often used in the literature:

$$\%\ \text{Concentration reduction} = 100 \times \left(\frac{C_i - C_o}{C_i}\right) \tag{6.7}$$

This term is quite ambiguous, because it usually refers to the average of one or more synchronous samples for selected stream flows. Such contemporaneous measures do not properly reflect the internal chemical dynamics of the wetland, such as production of the chemical. Further, dilution or

concentration due to rain (*ET*) or other unaccounted flows renders this an imperfect measure of true removal. Nevertheless, this terminology is frequently used in the literature.

## Chemical Terminology

It is important to distinguish among the various measures of global wetland chemical removal. Some further definitions used in this book are specified in the following text.

### Inlet Mass Loading Rates

$$M_i = Q_i C_i \tag{6.8}$$

$$m_i = \frac{M_i}{A} = \frac{Q_i C_i}{A} = q_i C_i \tag{6.9}$$

where
$M_i$ = inlet mass loading, g/d
$m_i$ = (specific) inlet mass loading, g/m$^2$·d

Acronyms are also often used for designating the chemical; for example, PLR denotes *phosphorus loading rate*. A chemical loading rate is a measure of the distributed "rainfall" equivalent of a chemical mass flow. It does not imply the physical distribution of water uniformly over the wetland.

### Mass Removal Rate

$$\tilde{J}A = (Q_i C_i - Q_o C_o) \tag{6.10}$$

This represents the areal average amount of a chemical that gets stored, destroyed, or transformed. This single-number measure of wetland performance can be misleading in the common event of strong concentration gradients and removal gradients.

### Percent Mass Removal

This quantity links water losses and gains to chemical losses and gains.

$$\%\ \text{Mass removal} = 100 \times \left( \frac{Q_i C_i - Q_o C_o}{Q_i C_i} \right) = 100 \times \left( \frac{m_i - m_o}{m_i} \right) \tag{6.11}$$

$$\left(1 - \frac{\%M}{100}\right) = \left(1 - \frac{\%Q}{100}\right)\left(1 - \frac{\%C}{100}\right) \tag{6.12}$$

where
$\%C$ = percent concentration reduction
$\%M$ = percent mass removal
$\%Q$ = percent flow reduction

The term is less ambiguous than concentration reduction, because it traces the chemical of interest, and accounts for the effect of the quantity of water in which that chemical is located. However, the difficulties of contemporaneous measurement remain.

### The Utility of Reduction Numbers

It is very easy to compare the amounts of a pollutant in the inlet and outlet streams of a wetland, and to compute the percentage difference. Unfortunately, this information is of very limited use in design or in performance predictions, because it reflects none of the features of the ecosystem, which are the target of design. By implication, it would be necessary to replicate the wetland that produced the percentage data, as well as to replicate the operating and environmental conditions that prevailed during data acquisition. The second is clearly impossible, and past experience has given strong indications that the first is also difficult.

The literature is replete with review papers that tabulate removals for a selected spectrum of wetlands (e.g., Strecker *et al.*, 1992; Cueto, 1993; Johnston, 1993).

The implication is that wetlands of a similar type will achieve a similar reduction. Whereas such groups of data begin to elucidate the bounds of performance, the effects of size, loading, flow patterns, depth, and other design variables cannot be deduced from efficiency values alone.

In some instances, the incoming concentration of a particular chemical may be small for some period of time. Then, due to measurement errors or small transfers from wetland, storages and productions may give outflow concentrations that are greater than the incoming values. A one-time calculation of a "reduction efficiency" will properly reflect that condition as a (large) negative percent reduction. At other times, a larger inflow concentration may be reduced by the wetland, leading to a positive percentage removal. If the removal percentages are then averaged, the large negative value improperly dominates the calculation.

As a result of these considerations, great care must be exercised in interpretation of percentage reduction values.

## Chemical Mass Balances

Measurements of chemical composition of wetland inflows and outflows are the most obvious method of characterizing water quality functions. However, such measurements by themselves can be very misleading. A much better characterization is achieved by computing the mass balance or budget for an individual chemical constituent.

A proper mass balance must satisfy the following conditions:

1. The *system* for the mass balance must be defined carefully. A system in this context means a defined volume in space; this is often taken to be the surface water in the wetland in the case of a free water surface (FWS) wetland or the water in the media for a subsurface flow (SSF) wetland. A precise definition is needed to compute the change in storage. The mass balance is termed *global* when

the entire wetland water body is chosen as the system. In later chapters, it will be useful to compute the *internal* mass balance, which is based on an internal element or subdivision of the water body.

2. The *time period* for totaling the inputs and outputs must be specified. It may be desirable to express inflows and outflows in terms of rates, but these must then be averaged over the time period chosen.
3. All *inputs* and *outputs* to the chosen system must be included. The concept of mass conservation may be invoked to calculate one or a group of material flows. A partial listing of some of the inflows and outflows does not constitute a mass balance.
4. Compounds undergo chemical reactions within a wetland ecosystem. Any *production* or *destruction reactions* that occur within the boundaries of the chosen system are to be included in the mass balance. Reactions outside the boundary are not counted, because an outflow must occur to transport the chemical to the external reaction site, and that is accounted as an outflow.
5. Waterborne chemical flows are determined by separate measurements of water flows and concentrations within those waters. Therefore, an accurate *water mass balance is a prerequisite* to an accurate chemical mass balance.
6. If at all possible, it is desirable to *demonstrate closure* of the mass balance. This is achieved by independently measuring every component of the mass balance. The degree of closure is often expressed as a percentage of total inflows. Unfortunately, closure has rarely been demonstrated for any chemical in any wetland.

The foundation for chemical mass balances is the wetland water mass balance (see Chapter 2). Transfers of water to and from the wetland follow the same pattern for both surface and subsurface flow wetlands. In treatment wetlands, wastewater additions are normally the dominant flow; but under some circumstances, other transfers of water are also important. The dynamic overall water budget for a wetland is:

$$Q_i - Q_o + Q_c - Q_b - Q_{gw} + Q_{sm} + A(P - ET) = \frac{dV}{dt} \qquad (2.13)$$

where

$A$ = wetland top surface area, $m^2$
$ET$ = evapotranspiration rate, m/d
$P$ = precipitation rate, m/d
$Q_b$ = bank loss rate, $m^3/d$
$Q_c$ = catchment runoff rate, $m^3/d$
$Q_{gw}$ = infiltration to groundwater, $m^3/d$
$Q_i$ = input wastewater flowrate, $m^3/d$
$Q_o$ = output wastewater flowrate, $m^3/d$
$Q_{sm}$ = snowmelt rate, $m^3/d$
$t$ = time, d
$V$ = water storage in wetland, $m^3$

It is difficult to establish detailed chemical mass balances over the wetland surface water because of the number and complexity of the possible transfers to and from the water, and their nonsteady character. It is common practice to measure only the principal inflows and outflows, and to ascribe the difference to "removal," which may be positive or negative. This lumping of all transfers to and from the water body is often unavoidable due to economic constraints. It is possible to write a general mass balance equation for a generic chemical species:

$$Q_i C_i - Q_o C_o + Q_c C_c \pm Q_{gw} C_{gw} + A(PC_p - \tilde{J}) = \frac{d(VC_s)}{dt} \qquad (6.13)$$

where

$C_c$ = concentration in catchment runoff, $g/m^3$
$C_{gw}$ = concentration in groundwater recharge or discharge, $g/m^3$
$C_i$ = concentration in inflow, $g/m^3$
$C_o$ = concentration in outflow, $g/m^3$
$C_p$ = concentration in precipitation, $g/m^3$
$C_s$ = concentration in wetland surface water, $g/m^3$
$\tilde{J}$ = spatially averaged removal rate, $g/m^2 \cdot d$

In Equation 6.13, bank losses and snowmelt have been omitted for the sake of simplicity. All the transfers have been lumped into one removal rate. Flow rates are instantaneous. The removal rate is the average over the entire wetland area, and the system concentration is averaged over the entire water volume.

The time period for the global mass balance is of critical importance because of the time scale of interior phenomena. Many wetlands, whether treatment or pristine natural, have long nominal detention times, which usually reflect long actual detention times. A two-week detention is not uncommon. If the wetland were in plug flow, an entering cohort of water would exit two weeks later. Clearly, same-day samples taken from inlet and outlet should not be used to compute "removals." In fact, wetland flow patterns are more complex than plug flow; the entering cohort breaks up, and pieces depart at various times after entry, some earlier and some later than the implied two-week detention. This difficulty of synchronous sampling may be alleviated in the mass balancing process by selecting a mass balance period that spans several detention times.

The removal term is the result of transfers to and from the soils and biomass compartments in the wetland, as well as of transfers to and from the atmosphere, and chemical conversions. Those biomass and soils compartments dominate the overall wetland storage and transformations for most chemicals. Therefore, the water body mass balance is very sensitive to small changes in transfers, reactions, and storages in biomass and soils. The removal rate depends very strongly on events in these solids compartments, and hence is determined in major part by the changing ecological state of the wetland. Because wetland biological processes are more or less repetitive on an annual cycle, the long-term performance of the wetland is best characterized by global mass balances

that span an integer number of years. Seasonal effects require a time period of three months, which is usually long enough to avoid storage errors and detention time offset.

Removal in Equation 6.14 is an areal average. However, in most flow through wetlands, there is a strong gradient in the unaveraged removal in the direction of flow. As the downstream wetland system "boundary" is moved successively further from the inlet, the areal average removal rate decreases. The average removal rate depends on the size of the portion of the overall wetland that is chosen for the global mass balance. This weakness of the global mass balance can be corrected by using the internal mass balance that reflects distance effects.

## 6.4 PROCESSES THAT CONTRIBUTE TO POLLUTANT REMOVALS

A large number of wetland processes may contribute to the removal or reduction of any given pollutant. Here, some of the most important are described and the commonly used rules for quantification are presented. More details are presented in the following chapters for the most common chemicals of interest. The discussion here relates to localized phenomena. Removal processes must also be quantitatively placed in the context of internal wetland hydraulics as well as the topography and vegetative structure of the wetland.

### Microbially Mediated Processes

Many wetland reactions are microbially mediated, which means that they are the result of the activity of bacteria or other microorganisms. Very few such organisms are found free-floating; rather, the great majority are attached to solid surfaces. Often, the numbers are sufficient to form relatively thick coatings on immersed surfaces.

Transfer of a chemical from water to immersed solid surfaces is the first step in the overall microbial removal mechanism. Those surfaces contain the biofilms responsible for microbial processing, as well as the binding sites for sorption processes. The following discussion analyzes the transport of dissolved constituents to reaction sites located in the biofilms that coat all wetland surfaces. Mass transfer takes place both in the biofilm and in the bulk water phase. Roots are the locus for nutrient and chemical uptake by the macrophytes, and these are accessed by diffusion and transpiration flows. The sediment–water interface is but one such active surface; the litter and stems within the water column comprise the dominant wetted area in FWS wetlands, and the media surface is the dominant area in SSF wetlands.

Dissolved materials must move from the bulk of the water to the vicinity of the solid surface, then diffuse through a stagnant water layer to the surface, and penetrate the biofilm while undergoing chemical transformation (Figure 6.10). This sequence of events has been described and modeled in the text of Bailey and Ollis (1986), and is outlined here. The case of zero wetland background concentration will be described here; but extension to the case of nonzero background is possible.

The rate of transfer across the two films is:

$$J_{mt} = \frac{D_w}{\delta_w}(C - C_{interface}) \tag{6.14}$$

$$J_{mt} = E\,k_b\,\delta_b\,C_{interface} \tag{6.15}$$

where

$C$ = concentration in the bulk water, mg/L = g/m$^3$
$C_{interface}$ = concentration at the biofilm surface, mg/L = g/m$^3$
$D_w$ = diffusion coefficient in water, m$^2$/d
$D_b$ = diffusion coefficient in biofilm, m$^2$/d
$\delta_b$ = thickness of the biofilm, m
$\delta_w$ = thickness of the stagnant boundary layer, m
$E$ = tanh($\phi$)/$\phi$, biofilm effectiveness factor, dimensionless
$J_{mt}$ = mass transfer rate, g/m$^2$·d
$k_b$ = reaction rate constant inside biofilm, d$^{-1}$

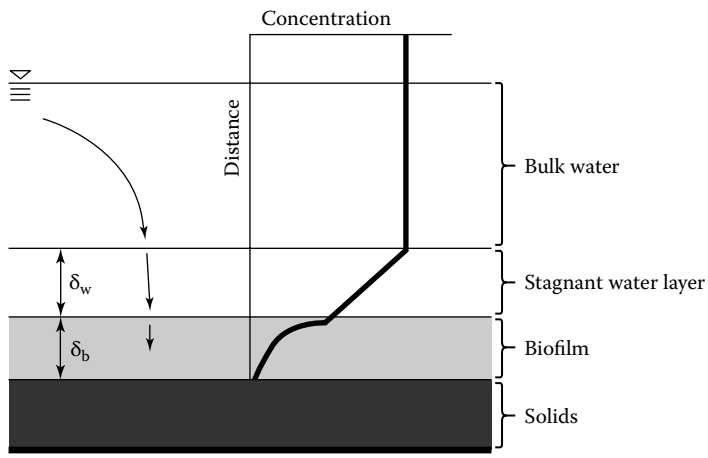



**FIGURE 6.10** Pathway for movement of a pollutant from the water across a diffusion layer and into a reactive biofilm. The solid may be sediment, a litter fragment, or a submerged portion of a live plant. (From Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

and where

$$\phi = \delta_b \sqrt{\frac{k_b}{D_b}} \tag{6.16}$$

Combining equations (the rate of transport of the pollutant from the bulk water to the biofilm) is then:

$$J_{mt} = \left[ \frac{E\, k_b\, \delta_b}{1+M} \right] C = k_i C \tag{6.17}$$

where

$k_i$ = intrinsic first order areal reaction rate constant, m/d.

$$M = \frac{E k_b \delta_b \delta_w}{D_w} \tag{6.18}$$

It is seen that this theory produces a local first-order rate of overall reaction, which depends upon biofilm properties and diffusion coefficients.

In a field situation, it is also necessary to know the area of biofilms that occupy a given area of wetland (Figure 6.11). The overall removal rate from a wetland area $A_w$ occurs from a biofilm area of $A_b$, and hence the rate of removal is

$$\begin{aligned} &\text{(a)} \qquad J\, A_w = k_i\, A_b\, C \\ &\text{(b)} \qquad J = k_i \left( \frac{A_b}{A_w} \right) C = k_i\, a_s\, C = k\, C \end{aligned} \tag{6.19}$$

where

$a_s$ = biofilm area per unit wetland area, $m^2/m^2$

$A_w$ = wetland area, $m^2$

$A_b$ = biofilm area, $m^2$

$k$ = first-order areal reaction rate constant, m/d

Data have been obtained for only a few FWS wetland systems, giving only a rough estimate of the magnitude of $a_s$. If there is no vegetation, and only the wetland bottom serves as the potential location of biofilms, the value of $a_s \leq 1.00$. If the emergent vegetation is considered an additional biofilm area, a dense stand of plants can yield $a_s \approx 5$. Inclusion of the litter can further increase the value to $a_s \approx 10$.

This theory has been calibrated for treatment wetlands by Polprasert and coworkers (Polprasert *et al.*, 1998; Khatiwada and Polprasert, 1999a), who determined $a_s$ in the range 2.2–2.9. Measurements of immersed vegetation surface area were made at Arcata, California, and Houghton Lake, Michigan, and produced $a_s$ in the range 1.0–9.0 (see Chapter 3).

Microbially mediated reactions are affected by temperature. Response is typically much greater to changes at the lower end of the temperature scale (<15°C) than the warmer range (20–35°C) (Kadlec and Reddy, 2001). Processes regulating organic matter decomposition are affected by temperature. Similarly, all nitrogen cycling reactions (mineralization, nitrification, and denitrification) are affected by temperature. The temperature coefficient (θ) varies from 1.05–1.37 for carbon and nitrogen cycling processes under isolated conditions. Phosphorus sorption reactions are least affected by temperature with θ-values of 1.03–1.12. However, treatment wetlands

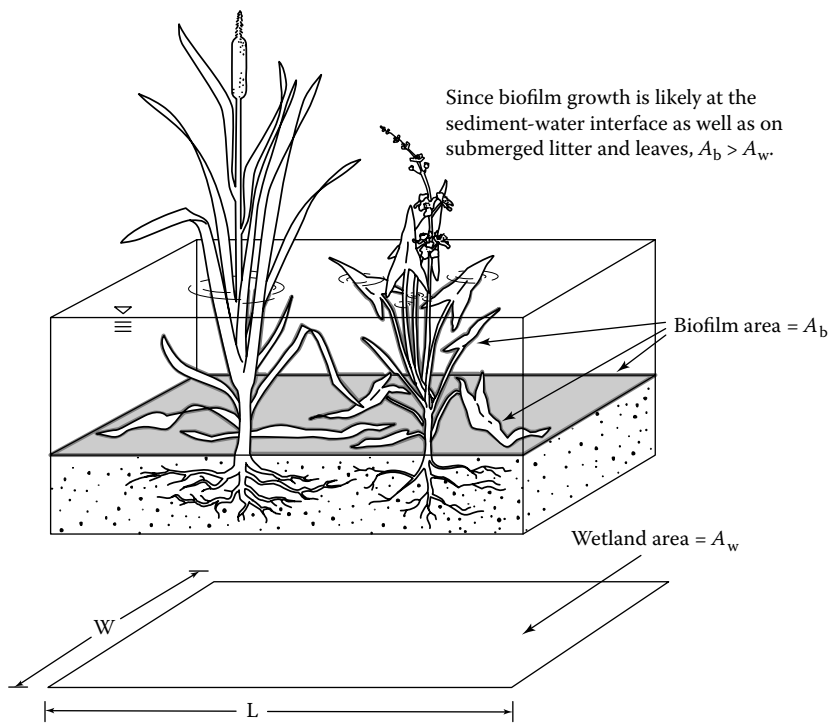



**FIGURE 6.11** Biofilms dominate the sediment–water interface, as well as the surfaces of the litter and standing dead material. (Adapted from Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

display lesser temperature effects because of their complexity (Kadlec and Reddy, 2001).

### Chemical Networks

Several wetland chemical removal processes involve more than one reaction and more than one chemical species. Many removal reactions create products that are themselves contaminants of interest. An important example is the (microbial) sequential conversion network for nitrogenous species:

$$\begin{aligned}\text{organic N} &\rightarrow \text{ammonia N} \rightarrow \text{oxidized N} \\ &\rightarrow \text{gaseous } N_2\end{aligned} \tag{6.20}$$

Each of the first three species is important in its own right as a potential contaminant. Both consumption and production can occur; it is, therefore, misleading to isolate one species and compute its "removal."

Another example is the reductive dechlorination of a chlorinated organic compound. Trichloroethylene has daughter products that are sequentially formed in a wetland environment:

$$\begin{aligned}\text{trichloroethylene} &\rightarrow \text{dichloroethylene} \rightarrow \text{vinyl chloride} \\ &\rightarrow CO_2 + H_2O + Cl^-\end{aligned} \tag{6.21}$$

In such cases, it is essential to utilize reaction models that account for both production and destruction. Each step may individually have a simple model, such as first order; but in combination, removal is quantitatively more complex.

### Volatilization

Various processes in wetland create product gases that are released from the wetland environment to the atmosphere, such as ammonia, hydrogen sulfide, dinitrogen, nitrous oxide, and methane. Wetlands also take in atmospheric carbon dioxide for photosynthesis and expel it from respiratory processes. The mechanism of volatilization is further discussed in Chapter 5 (for nitrous oxide, methane, and carbon dioxide), Chapter 9 (for ammonia), and Chapter 11 (for hydrogen sulfide).

### Sedimentation

Chapter 7 deals with the removal of suspended solids. Here, it is noted that a first-order areal removal model is the outcome of theory and practice. Many pollutants partition to suspended solids, and thus removal of those sorbed substances also is expected to follow that model:

$$J = k_{TSS} \cdot C_{TSS} \cdot K_p \cdot C \tag{6.22}$$

where

$C_{TSS}$ = suspended matter concentration, mg/L = g/m$^3$
$k_{TSS}$ = TSS removal rate constant, m/d
$K_p$ = partition coefficient, m$^3$/g

The necessary connection to solids removal is the quantification of sorption.

### Sorption

Partition coefficients relate the amount of sorbed pollutant to the concentration in the water under equilibrium conditions. Three types of sorption isotherms are in common use in wetland technology:

$$\begin{aligned}&\text{(a)}\quad \text{Linear:} && C_s = K_p C \\ &\text{(b)}\quad \text{Fruendlich:} && C_s = K_f C^n \\ &\text{(c)}\quad \text{Langmuir:} && C_s = K_L\left(\frac{C}{C+b}\right)\end{aligned} \tag{6.23}$$

The sorption potential for the principal contaminants of interest is discussed in the chapters pertaining to those contaminants. Here, a few generalities are noted:

- Sorption is important for phosphorus during the start-up period for a treatment wetland. If initially absent in the sediments, phosphorus will be stored until the existing soils and sediments reach equilibrium with the overlying water. If initially present, phosphorus may be released.
- Sorption is important for ammonia nitrogen in intermittently dosed or operated wetlands. Short-term storage may be oxidized during drawdown periods.
- Sorption is important for hydrophilic organic chemicals, which partition strongly to the carbonaceous content of wetland sediments.
- The water-phase concentration that is experienced by wetland sediments and soils is pore water, which can have very different concentrations than the bulk water overlying those sediments and soils.
- Sorption sites are a partially renewable resource, because they may be added from the accumulation of newly formed sediments.
- Sorption may be partially irreversible, due to mineralization of sorbed materials, or to the formation of very strong chemical bonds.
- Linear sorption (Equation 6.23(a)) results in a theoretical first-order removal process at the local level.

### Photodegradation

Sunlight can degrade or convert many waterborne substances. Many microorganisms, including pathogenic bacteria and viruses, can be killed by ultraviolet radiation. The effectiveness is presumptively determined by the radiation dose rate as well as the concentration of organisms. Although this is theoretically a second-order process, the sunlight dose in the wetland is relatively constant in the long run, and the elimination rate

is therefore pseudo first order in the organism concentration. A wide variety of chemicals are also susceptible to removal, in one or both of two ways. Direct photolysis involves the breakdown of the molecule, usually by the ultraviolet component of the sunlight. The nitrotoluenes are examples of readily photolyzable substances. Photooxidation occurs via reactions with free radicals formed by the incident radiation, such as alkylperoxy, hydroroxyl, and singlet oxygen radicals (Howard *et al.*, 1991). Photodegradation has received essentially no attention in treatment wetland research and development.

### Plant Uptake

Plants take up nutrients to sustain their metabolism. They may also take up trace chemicals found in the root zone, which may then be stored, or in some cases, expelled as gases. Uptake is by the roots, which are most often located in the wetland soils, although adventitious roots may sometimes be found in the water column. Submerged plants may absorb nutrients and metals from the water column into stems and leaves.

### Vertical Diffusion in Soils and Sediments

If there is no infiltration, driven either by hydraulic head or plant transpiration, to carry dissolved contaminants to sorption and reaction sites and roots located below ground, then diffusion is the dominant mechanism for vertical downward movement of pollutants. The presence of the soil matrix prevents convection currents; therefore, the diffusive process is further restricted to molecular diffusion. The model for this process is the diffusion equation:

$$J_D = -D\frac{dC_{pw}}{dz} \tag{6.24}$$

where

$C_{pw}$ = porewater concentration, g/m$^3$
$D$ = diffusion coefficient, m$^2$/d
$J_D$ = vertical diffusion flux, g/m$^2$·d
$z$ = vertical distance, m

The values of diffusion coefficients in pure water are of the order of $2 - 10 \times 10^{-5}$ m$^2$/d at 25°C (i.e., $2.9 \times 10^{-5}$ for COD, and $7.6 \times 10^{-5}$ for $H_2PO_4^-$). Values in the soil pore water are likely to be lower, by about a factor of 4, because of tortuosity and porosity effects.

Some idea of the importance of the diffusive process may be gained by examining the situation of mildly eutrophic surface waters overlying a fully saturated peatland. Reddy *et al.* (1991) report soluble reactive phosphorus pore water gradients as large as 3.0 gP/m$^3$·m in the top 20 cm of an Everglades cattail-dominated peatland. Under these circumstances, the diffusion flux predicted by Equation 6.24 is:

$$J_D = -\left(\frac{7.6 \times 10^{-5}}{4}\right)(3)(365) = 0.02 \text{ g/m}^2 \cdot \text{yr}$$

Uptake rates in that Everglades environment were independently measured, and found to be more than an order of magnitude higher than this predicted diffusive flux (Reddy *et al.*, 1991; Richardson *et al.*, 1992). Consequently, other mechanisms were operative. An important additional mechanism is the flow of water and phosphorus pulled into the root zone of emergent macrophytes to support transpiration.

### Transpiration Flux

Vertical flows of water in the upper soil horizon are also driven by plant water uptake to support transpiration. In aquatic and wetland systems with fully saturated soils or free surface water, the meteorological energy budget requires the vaporization of an amount of water sufficient to balance solar radiation and convective losses. Some of this vaporization is from the water surface (evaporation); some is from the emergent plants (transpiration). Emergent plants "pump" water from the root zone to the leaves from which water evaporates through stomata, which constitutes the transpiration loss (see Figure 4.6). In a densely vegetated wetland, transpiration dominates the combined process (evapotranspiration, which is abbreviated as *ET*) (see Chapter 4). Water for transpiration must move through the soil to the roots. That movement is vertically downward from overlying waters in FWS wetland situations, but directly from the flowing water in SSF wetlands.

Thus, transpiration has the potential to move on the order of 1 m/yr of water vertically downward to the root zone in an FWS system. That water carries with it the contaminant concentrations associated with the bottom layer of overlying water, which is the litter–benthic zone of the wetland. This flow is termed the transpiration stream (*TS*), and it draws from pore water that is typically at a concentration different from that of the bulk surface water. In turn, the plant may block a portion of the dissolved pollutant, and take up a concentration less than that of pore water. These factors combine to determine the amount of plant uptake (Trapp and Matthies, 1995; Gomez and Pardue, 2002):

$$J_U = TS \times TSCF \times C_{pw} \tag{6.25}$$

where

$C_{pw}$ = porewater concentration, g/m$^3$
$J_U$ = uptake flux, g/m$^2$·d
$TS$ = transpiration stream, m/d
$TSCF$ = transpiration stream concentration factor, dimensionless

In a moderately dense emergent FWS wetland, the transpiration flux is far greater than the estimated diffusion flux.

### Vertical Root Profiles

Plant roots are typically located in the top 30 cm of the soil, and most are in the top 20 cm (see Figure 2.29). However, rooting depths have been reported over a wide range. For example, for *Phragmites*, Moore *et al.* (1994) reported 10 cm,

while Börner *et al.* (1998) reported 150 cm. U.S. EPA (2000a) recommends rooting media depths for FWS constructed wetlands in the range 15–40 cm. For other species, rooting depth in FWS wetlands is typically 20–30 cm. For instance, Murkin *et al.* (2000) report that roots were found entirely within the top 20 cm for *Phragmites, Typha* spp., and *Scirpus* spp. in a natural prairie marsh. Similarly, Wentz (1976) reported decreasing root biomass down to 45 cm for *Carex* spp. in the Houghton Lake wetland. Given the vertical profile of root density, there is presumptively a corresponding vertical profile in the uptake of water and chemicals by the plant. However, such differential uptake is very difficult to measure; consequently, plant uptake is usually assigned to the vertically integrated root zone.

### Seasonal Cycles

Nutrient removal displays considerable seasonality for ammonia at low loadings. Accordingly, temperature is not always an acceptable surrogate for seasonality for nitrogen removal. Vegetative uptake in temperate climates is maximum during spring, at moderate temperatures, but release via decomposition is maximum during fall, also at moderate temperatures. Plants utilize phosphorus, nitrate, and ammonium, and decomposition processes release nitrogen and phosphorus back to the water. On an instantaneous basis, plant uptake can be important for many wetland systems (Kadlec, 2005d).

### Accretion

One of the least studied aspects of pollutant transfer in wetlands is in the creation of new soils and sediments, with their attendant chemical content. Not all the dead plant material undergoes decomposition. Some small portions of both aboveground and belowground necromass resist decay, and form stable new accretions. Such new stores of chemicals are presumed to be resistant to decomposition. The origins of new sediments may be from remnant macrophyte stem and leaf debris, remnants of dead roots and rhizomes, and from undecomposable fractions of dead microflora and microfauna (algae, fungi, invertebrates, bacteria).

The amount of such accretion has been quantified in only a few instances for FWS wetlands (Reddy *et al.*, 1991; Craft and Richardson, 1993; Rybczyk *et al.*, 2002), although anecdotal reports also exist (Kadlec, 1997a). Quantitative studies have relied upon either atmospheric deposition markers (radioactive cesium or radioactive lead), or introduced horizon markers, such as feldspar or plaster. Either technique requires several years of continued deposition for accuracy.

## 6.5 CHARACTERIZATION OF INTERNAL HYDRAULICS

The removal of pollutants within a constructed wetland occurs through the diverse range of interactions between the sediments, substrate, microorganisms, litter, plants, the atmosphere, and the wastewater as it moves through the system. The dynamics of water movement through the wetland has a significant influence on the efficiency and extent of these interactions. Many of the important biogeochemical reactions rely on contact time between wastewater constituents and microorganisms and the associated substrate, whereas wastewater velocity can be an important determining factor for other pollutant removal processes, such as mass transfer. Any short-circuiting or dead zones that occur within a wetland will, consequently, have an effect on contact time as well as flow velocities and, therefore, impact on treatment efficiency. Nonideal flow patterns can have very large effects upon the removal of pollutants in wetland treatment systems (Kadlec *et al.*, 1993; Carleton, 2002). It is, therefore, necessary to consider flow pattern effects and the related mixing in the design of wetland treatment systems.

Three types of hydraulic inefficiencies may occur in treatment wetlands:

1. Internal islands and other topographical features
2. Preferential flow channels at a large distance scale
3. Mixing effects, such as water delays in litter layers and transverse mixing

The first mechanism is characterized by a gross areal efficiency, which relates to the volumetric efficiency ($e_V$) of the wetland, as discussed in Chapter 2. The second and third types are characterized by an equivalent set of well-mixed units in series, or other "mixing" model. All three influence a wetland's ability to improve water quality.

The main method by which wetland scientists and engineers have gained information about internal hydraulic processes is through the use of inert tracers, which provide a means of tracking the movement of water through a wetland. The theory and practice behind hydraulic investigations have predominantly evolved out of the field of chemical reaction engineering (Fogler, 1992; Levenspiel, 1995). The details of tracer testing are covered in Appendix B. Here, a brief summary is presented.

### Tracer Tests

A tracer test is conducted by introducing an impulse of an inert substance into the wetland inlet at time zero. If water moved through the wetland in lock-step, such a tracer impulse would also exit as an impulse (a sharp spike of concentration). This result has never been observed in a wetland tracer test; the exit tracer is always a blurred, skewed bell-shaped curve. In the FWS wetland environment, there are mixing processes on a number of different distance scales. Expanses of open water permit development of surface wind-driven currents, which are matched by return flows in lower water layers. Deeper parallel zones in the FWS wetland carry more flow because of the depth effect on hydraulic resistance. These preferential channels may also be due to a lower vegetation density along some flow paths. A tracer impulse added to

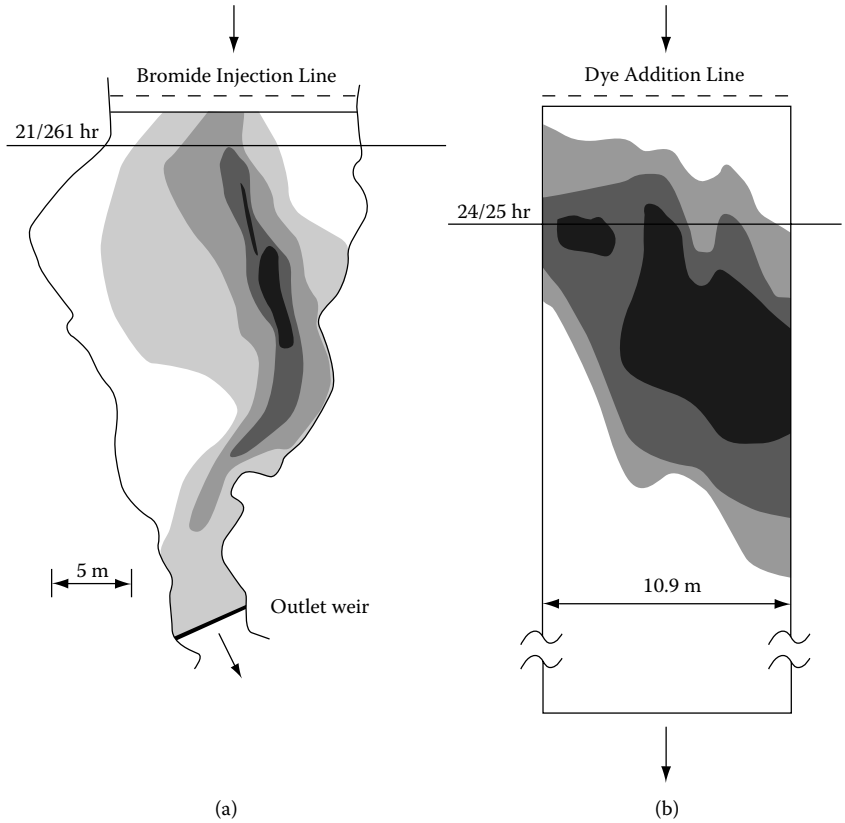



**FIGURE 6.12** Tracer isopleths in a natural (a) and constructed (b) wetland. In both cases, tracer was added uniformly across the inlet width. The theoretical location of the pulse centroid is shown by the horizontal line, labeled with elapsed time and theoretical detention time. (a) *Typha orientalis* natural wetland in New Zealand. (Data from A.B. Cooper (1992) *Coupling wetland treatment to land treatment: An innovative method for nitrogen stripping.* Proceedings of the 3rd International Conference on Wetland Systems for Water Pollution Control, Australian Water and Wastewater Association and IWA, Sydney, pp. 37.1–37.9.) (b) *Typha latifolia* FWS constructed wetland in Ontario. (Data from Herskowitz (1986) *Listowel Artificial Marsh Project Report.* Ontario Ministry of the Environment, Water Resources Branch, Toronto, Ontario.) (From Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

the incoming water provides a way to find such preferential paths as the tracer will later be found preferentially in those wetland zones. Both natural and constructed FWS wetlands display such flow variability (Figure 6.12). In particular, the results for constructed wetlands indicate that it is not possible to avoid such flow irregularities even with extreme care in construction.

There are also mixing effects in the vertical direction in FWS wetlands. Water may be moving more slowly near the bottom because of the increased drag of the dense litter layer. Those slow-moving zones exchange chemical constituents with adjacent faster-moving layers, and thus create vertical mixing. Dense plant clumps can effectively block flow even though these are of very high void fraction. Water in these clumps can exchange constituents with the adjacent microchannels by diffusive processes. All these effects combine to form a complicated overall mixing pattern. The result of such mixing is evidenced in the blurring of a tracer impulse added to the incoming water (Figure 6.12).

In a subsurface flow wetland, large-scale eddies and wind mixing are absent. However, preferential flow channels can occur on a large scale. Lateral inhomogeneities may contribute to nonuniform flow distribution across the width of the wetland (Marsteiner, 1997). Evidence of this was found for a HSSF wetland at Benton, Kentucky, by internal sampling of tracer responses (Figure 6.14). An impulse of tracer (Rhodamine WT) was added to the inlet flow to this HSSF wetland. Water was distributed across the entire width of the rectangular wetland. The observed responses were considerably different at equidistant sampling points, indicating subsurface preferential paths. Further, there is abundant evidence that vertical stratification occurs in gravel beds, with larger flows occurring at lower elevations (Fisher, 1990; Marsteiner, 1997; Drizo *et al*., 2000). The tracer concentrations that reach the HSSF wetland effluent are there blended to form an average outlet concentration. The response of a typical HSSF wetland to an impulse tracer input is a time-delayed bell-shaped curve (Figure 6.15).

## Interpretation of Data

The results of an impulse tracer test provide the volumetric efficiency ($e_V$) of the wetland, together with information on the distribution of detention times in the system. The first requirement of a valid tracer test is that the tracer be recovered nearly in its entirety at the wetland outlet. To that end, a simple check is made by summing the tracer at the

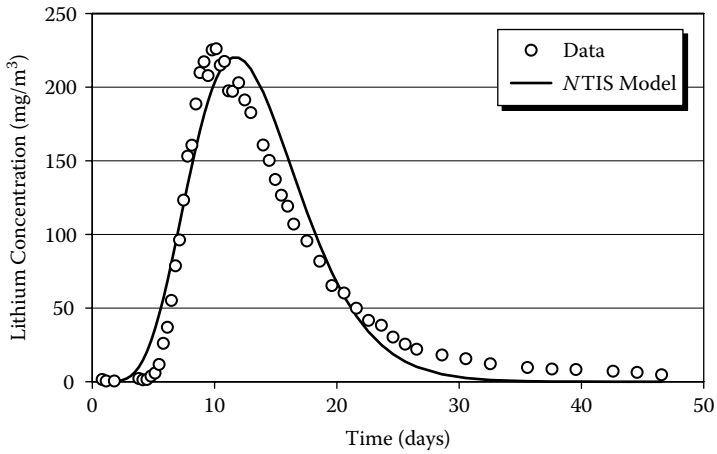



**FIGURE 6.13** Results of a lithium tracer test of a 91-ha FWS wetland receiving 193,000 m³/d. Approximately 500 kg of lithium were added. The TIS model is calibrated by about 8 TIS. (Data from Dierberg and DeBusk (2005) *Wetlands* 25(1): 8–25.)

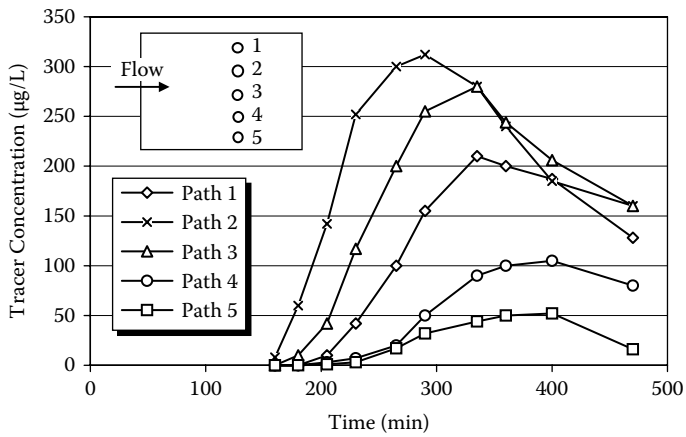



**FIGURE 6.14** Tracer concentrations at five stations normal for the flow direction in Gravel Bed Wetland #3 at Benton, Kentucky. Although these traces are not complete, it is clear that more tracer arrives sooner at Station 2 than at other stations. (Data from TVA unpublished data.) (From Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

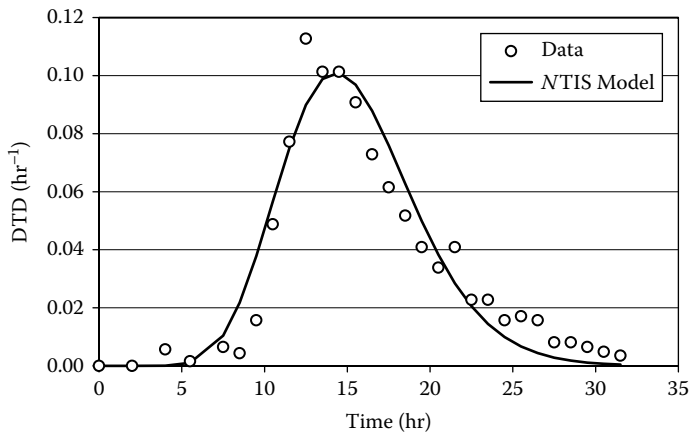



**FIGURE 6.15** Response of Cell 1 at Minoa, New York, to a tracer impulse. The TIS model is calibrated to 14 TIS, and the volumetric efficiency is 75%. (Data from Marsteiner (1997) *Subsurface Flow Constructed Wetland Hydraulics*. M.S. Thesis, Clarkson University (Potsdam, New York) 130 pp.)

wetland outfall:

$$M_o = \int_0^\infty Q_o C \, dt = ? = M_i \tag{6.26}$$

where

$C$ = outlet tracer concentration, mg/L
$M_i$ = mass of tracer in, g
$M_o$ = mass of tracer out, g
$Q_o$ = outflow, $m^3/d$
$t$ = time, days

The detention time distribution (DTD) represents the time various fractions of fluid (water in the case of the wetland) spend in the reactor, and hence is the contact time distribution for the system. In a broader context, the DTD is the probability density function for residence times in the wetland. This time function is defined by

$$f(t)\Delta t = \text{fraction of the incoming water which stays in the wetland for a length of time between } t \text{ and } t + \Delta t \tag{6.27}$$

where

$f$ = DTD function, $d^{-1}$

For an impulse input of tracer into a steadily flowing system, the function $f(t)$ is

$$f(t) = \frac{Q_o C}{\int_0^\infty Q_o C \, dt} = \frac{C}{\int_0^\infty C \, dt} \tag{6.28}$$

The first numerator is the mass flow of tracer in the wetland effluent at any time, *t,* after the time of the impulse addition. The first denominator is the sum of all the tracer collected and thus should equal the total mass of tracer injected.

The mean *tracer detention time* ($\tau$) is presumed to be the actual mean detention time, and is calculated from

$$\tau = \frac{1}{M_i} \int_0^\infty t Q C \, dt \tag{6.29}$$

where

$\tau$ = tracer detention time, days

A wetland may have internal excluded zones that do not interact with flow, such as the volume occupied by plant materials. In a steady-state system without excluded zones, the tracer detention time ($\tau$) equals the nominal residence time ($\tau_n$). This is true whether the flow patterns are ideal (plug flow or well mixed) or nonideal (intermediate degree of mixing). An adsorbing tracer will produce an artificially short detention time, which may then be erroneously presumed to result from a large excluded zone. An incorrect topography may be due to either positive or negative differences between $\tau$ and $\tau_n$. The ratio of tracer to nominal detention time is the volumetric efficiency:

$$e_V = \frac{\tau}{\tau_n} \tag{6.30}$$

where

$\tau_n = V/Q$ = nominal detention time, days
$V$ = wetland water volume, $m^3$

There are a variety of reasons why the value of $e_V$ is different from unity, as discussed previously in this chapter and in Chapter 2.

A second parameter which can be determined directly from the residence time distribution is the variance ($\sigma^2$), which characterizes the spread of the tracer response curve about the mean of the distribution, which is $\tau$:

$$\int_0^\infty (t - \tau^2) f(t) \, dt = \sigma^2 \tag{6.31}$$

where

$\sigma^2$ = DTD variance, $d^2$

The variance of the DTD is created by mixing of water during passage or, equivalently, by a distribution of velocities of passage. This can be lateral, longitudinal, or vertical mixing, or parallel flows of different velocities. An adsorbing tracer will lead to a narrowed response pulse, and hence to an erroneously low degree of mixing. This measure of dispersive processes may be rendered dimensionless by dividing by the square of the tracer detention time:

$$\sigma_\theta^2 = \frac{\sigma^2}{\tau^2} \tag{6.32}$$

The new parameter is $\sigma_\theta^2$, the *dimensionless variance* of the tracer pulse.

## Models for Internal Hydraulics

Tracer testing is not an end in itself; rather, it is conducted to support the modeling and calculation of contaminant removals in the wetland system. Accordingly, the tracer information is combined with the local, or intrinsic, removal rate to produce the wetland outlet concentration. There are many candidate models that may be used, which typically involve series and parallel combinations of two idealized flow elements: perfectly mixed units and plug flow sections (Figure 6.16). It is clear from numerous studies that treatment wetlands are neither plug flow nor well-mixed. The tanks-in-series (TIS) model captures the important features of wetland DTDs that produce the skewed bell-shaped response. The TIS model requires two parameters: the number of "tanks" ($N$), and the mean tracer detention time ($\tau$). As the model networks increase in complexity, such as the parallel path and finite stage models, they are able to resolve the

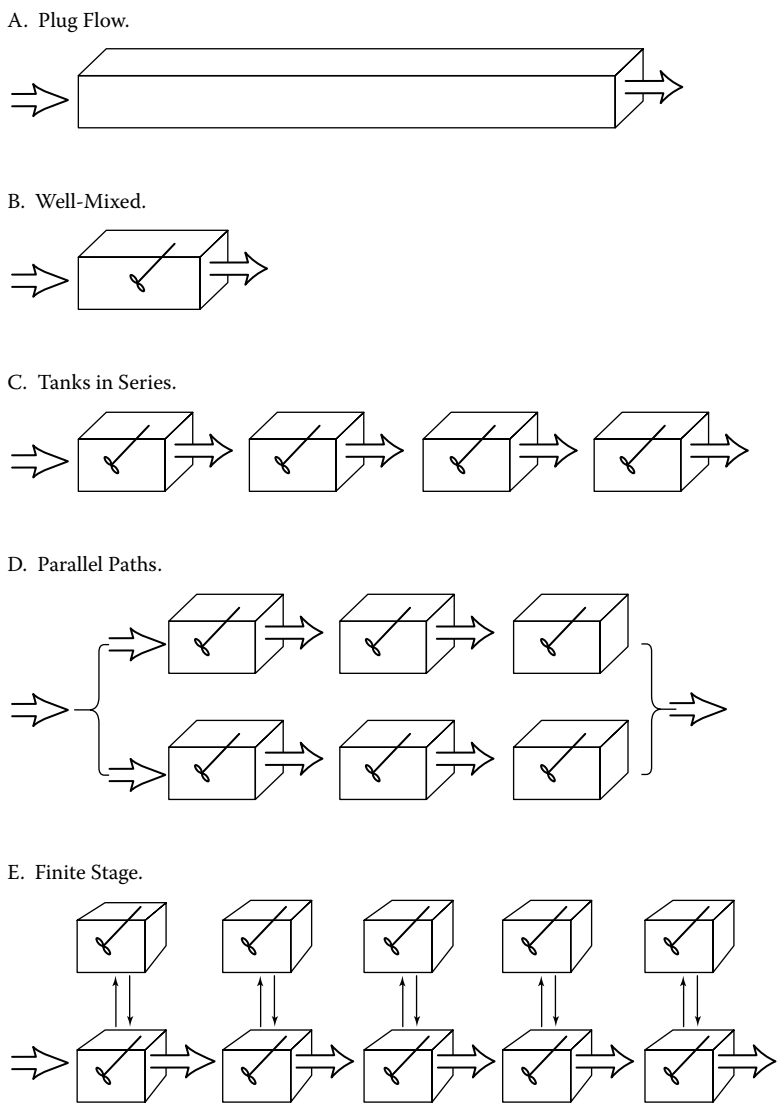



**FIGURE 6.16** A sample of various models to represent wetland tracer responses. The plug flow model (A) produces an impulse output at one detention time. The well-mixed model (B) produces an exponential decline. Models (C), (D), and (E) produce skewed bell-shaped responses.

last bits of detail in the responses, but do so at the expense of adding more calibration parameters. In the extreme, it is possible to use complicated computer codes to model wetland tracer responses (Martinez and Wise, 2003a; Keefe *et al*., 2004). In this work, the TIS model is utilized as a spreadsheet compromise between too many parameters and too little detail.

## Extreme Models

The two extremes of models are the single stirred tank and plug flow. Much of the literature about flow through lakes assumes that the lake behaves as a single well-mixed unit (one tank). In contrast, rivers are often conceptualized as plug flow systems, possibly with some dispersion. Much of the early treatment wetland literature presumed plug flow, for unspecified reasons (U.S. EPA, 1988b; Water Environment Federation, 1990). The wetland tracer studies of the early 1990s made it apparent that neither extreme applied to FWS wetlands, and in many instances did not apply to HSSF systems either. Kadlec and Knight (1996) knew that plug flow did not apply, but reasoned that the plug flow assumption would be "conservative," provided that a background concentration was acknowledged. It is now known that the plug flow assumption is not always conservative (Kadlec, 1999a).

The danger in the plug flow model results from its propensity to forecast extremely low effluent concentrations, when in reality, even minor amounts of short-circuiting preclude that from happening. Therefore, the probability of design mistakes at long detention times is very high. The temptation to calculate plug flow rate constants is huge: just put numbers into Equation 6.33:

$$k = q \cdot \ln\left(\frac{C_i}{C_o}\right) \tag{6.33}$$

Other models generally require curve fitting, and are therefore more time consuming. There are two major difficulties with

such calculations: (1) Equation 6.33 does not apply to synchronous samples, because of transport delay; and (2) there is no indication of the amount of variability removed by this model. If no variability is removed, the model is a useless forecaster.

The plug flow model is often an acceptable interpolator on existing data sets (Kadlec, 1999a). Thus, if high flow and low flow performance for a given system are known, a plug flow interpolation is reasonable. The difficulties arise when the model is used for extrapolation to low outlet concentrations or for extrapolation from one configuration to another. In both cases, discrepancies of a factor of two to five may easily be encountered (Figure 6.17). The parameter $P$ in Figure 6.17 is a modification of the number of tanks in series, $N$, as discussed later in this chapter.

Despite these shortcomings, the wetland literature continues to espouse the plug flow formulation (e.g., Water Environment Federation, 2001; Rousseau *et al.*, 2004; Crites *et al.*, 2006). In this book, models that include the hydraulics and configuration are used. There is no loss of the ability to include near-plug-flow in those situations where it is warranted.

### *Tanks in Series*

The TIS model is a gamma distribution of detention times:

$$g(t) = \frac{N}{\tau\,\Gamma(N)}\left(\frac{Nt}{\tau}\right)^{N-1}\exp\left(-\frac{Nt}{\tau}\right) \tag{6.34}$$

where

$\Gamma(N)$ = gamma function of $N$, = $(N-1)!$, factorial, if $N$ is an integer, $d^{-1}$
$N$ = number of tanks (shape parameter), unitless
$t$ = detention time, d
$\tau$ = mean detention time, d

When $N = 1$, the gamma distribution becomes the exponential distribution. Both the gamma distribution and the gamma function are readily available in handbooks (e.g., Dwight, 1961), or as computer spreadsheet tools (e.g., GAMMADIST and GAMMALN in Excel™). Equation 6.34 may easily be fit to tracer data by selecting $N$ and $\tau$ to minimize error (e.g., SOLVER in Excel™). This is a gradient search procedure, in which $N$ and $\tau$ are selected to minimize the sum of the squared errors (SSQE) between the DTD model and the data. Old textbook methods involve computation of the first and second moments of the experimental outlet concentration distribution, which are related to tracer detention time and the number of TIS, respectively. A serious failing of that moment method is that minor concentration anomalies on the "tail" of the concentration response curve may yield spurious parameter values, and bad fits of the main part of the DTD. The mode of the distribution (peak time), and its height, are also useful in determining $N$ and $\tau$, but the peak may not be well defined. For purposes of parameterization, it is noted that for the TIS model or gamma DTD distributions:

$$\sigma_\theta^2 = \frac{1}{N} \tag{6.35}$$

$$\frac{\tau - \tau_{peak}}{\tau} = \frac{1}{N} \tag{6.36}$$

Examples of least squares gamma fits of tracer data are shown previously in Figures 6.13 and 6.15. It is to be noted that although gamma distributions describe TIS mixing, the converse is not true. A gamma distribution of detention times does not imply the existence of turbulent mixing. Indeed, a gamma distribution may also arise from totally unmixed, separate travel paths with different velocities (Kadlec, 2000).

In the limit as $N$ becomes very large, the gamma distribution becomes the plug flow (PF) distribution, with all water departing after exactly one detention time. This limiting case does not exist for treatment wetlands. Reported literature values are $N = 4.1 \pm 0.4$ (mean ± SE) for FWS wetlands, and $N = 11.0 \pm 1.2$ for HSSF wetlands (Tables 6.1–6.2). However,

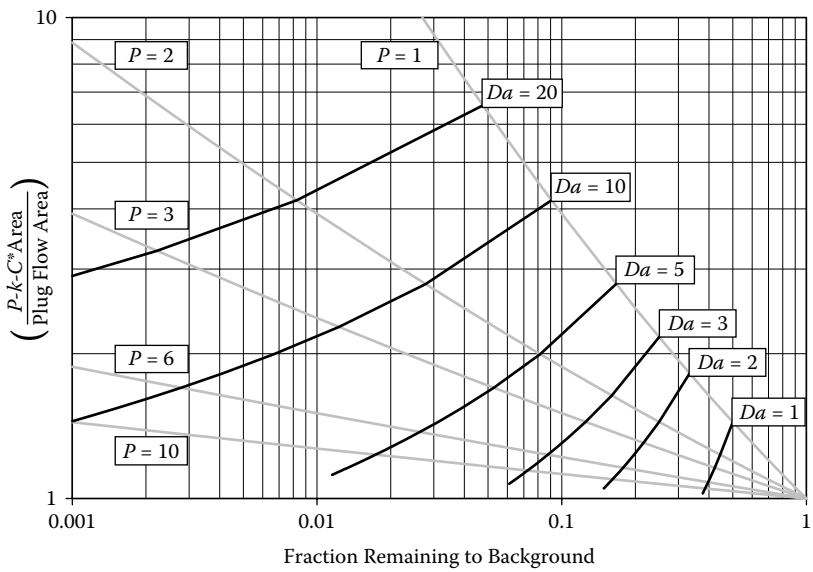



**FIGURE 6.17** Comparison of plug flow and $P$-$k$-$C^*$ areas required for specified percentage reductions. Note that the areas are much larger for low values of $P$, and for higher Damköhler numbers ($Da = k/q$). The fraction remaining to background is $(C - C^*)/(C_i - C^*)$.

**TABLE 6.1**
**A Sampling of Tracer Study Results from Various Sizes of FWS Wetlands**

| State or Country | Project | Vegetation | Name | Size ($m^2$) | Depth (cm) | Tracer | Recovery (%) | nHRT (days) | Tracer HRT (days) | Volumetric Efficiency (%) | *N*TIS | Source |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Florida | Everglades | SAV | Mesocosms | 4 | 40 | Lithium | 75 | 3.6 | 4.0 | 111 | 4.0 | 1 |
| Florida | Everglades | SAV | Mesocosms | 4 | 80 | Lithium | 90 | 7.2 | 7.7 | 107 | 7.7 | 1 |
| Florida | Everglades | SAV | Mesocosms | 4 | 120 | Lithium | 96 | 10.8 | 9.0 | 83 | 9.0 | 1 |
| Florida | Everglades | Algae | Mesocosms | 6 | 44 | Lithium | 86 | 11.1 | 13.1 | 118 | 1.4 | 1 |
| Florida | Everglades | Algae | Mesocosms | 18 | 34 | Lithium | 75 | 6.5 | 16.6 | 256 | 1.4 | 1 |
| Australia | Richmond | Open Water | Open Water | 400 | 0.5 | RWT | — | 5.70 | 4.63 | 81 | 1.6 | 2 |
| Australia | Richmond | SAV | SAV | 400 | 0.5 | RWT | — | 6.30 | 5.00 | 79 | 5.5 | 2 |
| Florida | Champion | Mixed Emergent | E | 1,000 | 34 | Lithium | 82 | 0.97 | 0.88 | 91 | 10.7 | 3 |
| Florida | Champion | Mixed Emergent | F | 1,000 | 68 | Lithium | 105 | 3.38 | 2.50 | 74 | 2.0 | 3 |
| Florida | Everglades | SAV | Test Cells | 2,000 | 71 | RWT | 83 | 5.6 | 5.9 | 105 | 5.9 | 1 |
| Florida | Everglades | Algae | Test Cells | 2,000 | 48 | Lithium | 82 | 10.6 | 11.6 | 109 | 3.1 | 1 |
| Florida | Champion | Mixed Emergent | C | 2,000 | 34 | Lithium | 90 | 2.59 | 1.12 | 43 | 4.0 | 3 |
| Florida | Champion | Mixed Emergent | D | 2,000 | 57 | Lithium | 58 | 11.83 | 9.66 | 82 | 3.5 | 3 |
| Florida | Everglades | Cattails | Test Cells | 2,457 | 36 | Lithium | 95 | 3.6 | 2.1 | 60 | 4.3 | 1 |
| California | Sacramento | Bulrushes | 7B | 2,926 | 55 | Lithium | 76 | 4.8 | 4.7 | 98 | 4.2 | 4 |
| California | Sacramento | Bulrushes | 9B | 2,926 | 56 | Lithium | 78 | 4.8 | 6.4 | 133 | 5.2 | 4 |
| Florida | Champion | Mixed Emergent | A | 4,000 | 34 | Lithium | 64 | 7.48 | 4.68 | 63 | 6.3 | 3 |
| Florida | Champion | Mixed Emergent | B | 4,000 | 48 | Lithium | 37 | 21.45 | 17.40 | 81 | 3.6 | 3 |
| Arizona | Tres Rios | Bulrushes | H2 | 9,100 | 49 | Bromide | 95 | 4.7 | 2.8 | 60 | 5.8 | 5 |
| Arizona | Tres Rios | Bulrushes | H1 | 9,200 | 49 | Bromide | 80 | 4.3 | 3.3 | 77 | 8.6 | 5 |
| Arizona | Tres Rios | Bulrushes | C2 | 12,800 | 52 | Bromide | 99 | 3.0 | 1.8 | 62 | 7.2 | 5 |
| Arizona | Tres Rios | Bulrushes | C1 | 13,400 | 67 | Bromide | 79 | 3.0 | 2.4 | 79 | 6.1 | 5 |
| Illinois | Des Plaines | Cattails | EW3 | 20,000 | 60 | Lithium | 98 | 12.3 | 8.4 | 68 | 2.7 | 6 |
| Florida | Iron Bridge | Mixed Emergent | 3 | 56,680 | — | Bromide | 91 | 2.57 | 1.34 | 52 | 1.4 | 7 |
| Florida | Iron Bridge | Mixed Emergent | 4 | 56,680 | — | Bromide | 95 | 8.97 | 2.28 | 25 | 2.1 | 7 |
| Florida | Iron Bridge | Mixed Emergent | 7 | 117,409 | — | Bromide | 97 | 13 | 2.63 | 20 | 3.1 | 7 |
| Florida | Iron Bridge | Mixed Emergent | 8 | 121,457 | — | Bromide | 112 | 2.95 | 1.24 | 42 | 0.3 | 7 |
| Florida | Iron Bridge | Mixed Emergent | 1 | 230,769 | — | Bromide | 109 | 4.53 | 1.38 | 30 | 1.3 | 7 |

| State or Country | Project | Vegetation | Name | Size ($m^2$) | Depth (cm) | Tracer | Recovery (%) | nHRT (days) | Tracer HRT (days) | Volumetric Efficiency (%) | *N*TIS | Source |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Florida | Lakeland | Cattails | Cell 4 | 300,000 | — | Lithium | 84 | — | 35.00 | — | 2.8 | 8 |
| Florida | Lakeland | Cattails | Cell 2 | 770,000 | — | Lithium | 67 | — | 5.70 | — | 0.3 | 8 |
| Florida | Lakeland | Mixed Emergent | Cell 1 | 810,000 | — | Lithium | 75 | — | 0.70 | — | 1.5 | 8 |
| Florida | Everglades | SAV | ENR4 | 1,470,000 | 74 | RWT | 83 | — | 4.3 | — | 3.5 | 1 |
| Florida | Everglades | SAV | ENR4 | 1,470,000 | 76 | RWT | 60 | — | 2.8 | — | 1.2 | 1 |
| Florida | Everglades | Mixed Emergent | ENR2 | 4,140,000 | 83 | Lithium | — | — | 11.3 | — | 5.6 | 1 |
| Florida | Everglades | Mixed Emergent | ENR1 | 5,780,000 | 58 | Lithium | — | — | 13.4 | — | 7.9 | 1 |
| | | | | | | **Mean** | **84** | **6.7** | **6.5** | **82** | **4.1** | |
| | | | | | | **SD** | **3** | | | **8** | **0.4** | |

*Sources:*

1. Unpublished data, South Florida Water Management District
2. Bavor *et al.* (1988)
3. Unpublished data, Champion Paper Co.
4. Nolte and Associates (1998b)
5. Unpublished data, City of Phoenix
6. Kadlec (1994)
7. Wang and Jawitz (2006)
8. Unpublished data, City of Lakeland

**TABLE 6.2**
**A Sampling of Tracer Study Results from Various Sizes of HSSF Wetlands**

| State or Country | Project | Vegetation | Name | Size ($m^2$) | Depth (cm) | Tracer | Recovery (%) | nHRT (days) | Tracer HRT (days) | Volumetric Efficiency (%) | *N*TIS | Source |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Quebec | Saint-Damien | *Phragmites* | T2 | 1.0 | 25 | Lithium | 63 | 4.30 | 5.13 | 119 | 3.4 | 1 |
| Quebec | Saint-Damien | Cattails | T3 | 1.0 | 25 | Lithium | 65 | 5.95 | 4.54 | 76 | 2.5 | 1 |
| Tennessee | Baxter | Bulrush | B1 | 5.9 | 45 | Lithium | 94 | 1.75 | 2.01 | 115 | 5.3 | 2 |
| Tennessee | Baxter | Bulrush | B2 | 5.9 | 45 | Lithium | 100 | 1.00 | 0.83 | 83 | 6.0 | 2 |
| Tennessee | Baxter | Bulrush | K | 5.9 | 45 | Lithium | 59 | 4.88 | 6.71 | 138 | 4.9 | 2 |
| Tennessee | Baxter | Bulrush | I | 5.9 | 45 | Lithium | 160 | 1.61 | 1.67 | 103 | 5.6 | 2 |
| Tennessee | Baxter | Bulrush | N | 11.8 | 45 | Lithium | 89 | 1.75 | 1.80 | 103 | 7.2 | 2 |
| California | UC–Davis | Cattails | Old | 15 | 95 | Bromide | 94 | 9.7 | 8.66 | 89 | 23.4 | 3 |
| California | UC–Davis | None | New | 15 | 95 | Bromide | 99 | 9.7 | 11.13 | 115 | 24.1 | 3 |
| Spain | Barcelona | *Phragmites* | A1 | 55 | 50 | Bromide | 86 | 5.13 | 5.25 | 102 | 3.4 | 4 |
| Spain | Barcelona | *Phragmites* | A2 | 55 | 50 | Bromide | 106 | 5.13 | 5.17 | 101 | 3.4 | 4 |
| Spain | Barcelona | *Phragmites* | B1 | 55 | 50 | Bromide | 99 | 5.13 | 4.50 | 88 | 5.3 | 4 |
| Spain | Barcelona | *Phragmites* | B2 | 55 | 50 | Bromide | 92 | 5.13 | 7.00 | 137 | 8.3 | 4 |
| Spain | Barcelona | *Phragmites* | C1 | 55 | 50 | Bromide | 94 | 5.13 | 5.50 | 107 | 6.7 | 4 |
| Spain | Barcelona | *Phragmites* | C2 | 55 | 50 | Bromide | 105 | 5.13 | 6.54 | 128 | 11.1 | 4 |
| North Carolina | New Hanover | Bulrush | SSF-R | 61 | 60 | Lithium | 98 | 5.3 | 4.61 | 87 | 6.8 | 5 |
| North Carolina | New Hanover | None | SSF-C | 61 | 60 | Lithium | 96 | 2.9 | 2.58 | 89 | 7.2 | 5 |
| New Zealand | Paraparaumu | — | Point | 132 | 78 | RWT | — | 4.00 | 2.55 | 64 | 4.5 | 6 |
| New Zealand | Paraparaumu | — | Manifold | 132 | 78 | RWT | — | 4.00 | 3.60 | 90 | 5.5 | 6 |
| Minnesota | Grand Lake | Cattails | Cell 1 | 182 | 60 | Bromide | 84 | 15.1 | 12.10 | 80 | 4.8 | 7 |
| Australia | Richmond | Bulrush | Trench 3 | 400 | 69 | RWT | — | 4.23 | 4.02 | 95 | 13.8 | 8 |
| Australia | Richmond | Cattails | Trench 5 | 400 | 68 | RWT | — | 4.20 | 3.08 | 73 | 21.0 | 8 |
| Australia | Richmond | None | Trench 2 | 400 | 45 | RWT | — | 3.10 | 3.41 | 110 | 25.2 | 8 |
| France | Curienne | *Phragmites* | Mar-00 | 605 | 72 | Chloride | 96 | 3.00 | 2.00 | 67 | 10.0 | 9 |
| France | Curienne | *Phragmites* | May-00 | 605 | 72 | Chloride | 82 | 1.48 | 1.79 | 121 | 16.0 | 9 |
| France | Curienne | *Phragmites* | Jul-00 | 605 | 72 | Chloride | 87 | 4.34 | 3.29 | 76 | 14.0 | 9 |
| France | Curienne | *Phragmites* | Dec-00 | 605 | 72 | Chloride | 78 | 2.99 | 1.88 | 63 | 7.0 | 9 |
| France | Curienne | *Phragmites* | Mar-01 | 605 | 72 | Chloride | 91 | 1.28 | 1.29 | 101 | 11.0 | 9 |
| France | Curienne | *Phragmites* | May-01 | 605 | 72 | Chloride | 93 | 3.03 | 1.50 | 50 | 9.0 | 9 |
| France | Curienne | *Phragmites* | Jul-01 | 605 | 72 | Chloride | 79 | 4.16 | 3.33 | 80 | 14.0 | 9 |
| France | Curienne | *Phragmites* | Dec-01 | 605 | 72 | Chloride | 92 | 3.33 | 1.83 | 55 | 7.0 | 9 |

| State or Country | Project | Vegetation | Name | Size ($m^2$) | Depth (cm) | Tracer | Recovery (%) | nHRT (days) | Tracer HRT (days) | Volumetric Efficiency (%) | *N*TIS | Source |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| New York | Minoa | *Phragmites* | Cell 3 | 1,000 | 76 | Bromide | 81 | 0.60 | 0.47 | 78 | 10.6 | 10 |
| New York | Minoa | *Phragmites* | Cell 1 | 1,700 | 76 | Bromide | 128 | 0.85 | 0.64 | 75 | 14.3 | 10 |
| New York | Minoa | *Phragmites* | Cell 2 Bottom | 1,750 | 76 | Bromide | 95 | 0.85 | 0.66 | 78 | 10.7 | 10 |
| New York | Minoa | *Phragmites* | Cell 2 Top | 1,750 | 76 | Bromide | 100 | 0.85 | 0.64 | 77 | 16.8 | 10 |
| New York | Minoa | *Phragmites* | Cell 2AB | 1,750 | 76 | Bromide | 116 | 0.83 | 0.73 | 88 | 21.6 | 10 |
| New York | Minoa | *Phragmites* | Cell 2 Clogged | 1,750 | 76 | Bromide | 45 | 5.43 | 4.50 | 83 | 34.4 | 10 |
| | | | | | | **Mean** | **92** | | | **91** | **11.0** | |
| | | | | | | **SD** | **4** | | | **4** | **1.2** | |

*Sources:*

1. Chazarenc *et al.* (2004)
2. George *et al.* (1998)
3. Grismer *et al.* (2001)
4. García *et al.* (2004b)
5. Rash and Liehr (1999)
6. Shilton and Prasad (1996)
7. Kadlec *et al.* (2003b)
8. Bavor *et al.* (1988)
9. Chazarenc *et al.* (2003)
10. Marsteiner (1997)

the range of values is quite large, and depends strongly on wetland configuration, which will be discussed in subsequent chapters.

This TIS quantification of internal hydraulics forms the basis for development of reaction models for treatment wetlands. The TIS hydraulic model is flexible enough to describe both mixing and preferential flow paths for a wide range of hydraulic efficiencies.

*Plug Flow with Dispersion*

Another model uses a dispersion process superimposed on a plug flow model (PFD). Mixing is presumed to follow a convective diffusion equation. A 1-D spatial model is chosen, because analytical expressions are available for computation of pollutant removal for the 1-D case (Fogler, 1992). A 2-D version requires the 2-D velocity field, which has yet to be determined for any operating treatment wetland. The tracer mass balance equation includes both spatial and temporal variability:

$$\left( D\frac{\partial^2 C}{\partial x^2} - \frac{\partial (uC)}{\partial x} \right) = \frac{\partial C}{\partial t} \tag{6.37}$$

where

$u$ = velocity, m/d
$D$ = dispersion constant, $m^2/d$
$x$ = distance from inlet toward outlet, m

The appropriate wetland boundary conditions for this mass balance are known as the *closed-closed* boundary conditions (Fogler, 1992). These imply that no tracer can diffuse back from the wetland into the inlet pipe, nor back up the exit structure at the wetland outlet. These are different from the *open-open* boundary conditions that are appropriate for river studies. There are analytical, close-form solutions to the latter case, which has led to their repeated misapplication to wetlands (Bavor *et al.*, 1988; Stairs, 1993). There are no closed form solutions to the wetland case, but numerical solutions to the closed-closed tracer mass balance have been available for more than three decades (Levenspiel, 1972). It is possible to calculate the dispersion constant that fits a particular data set, although there are issues of accuracy. This model is not advocated here, because the PFD model is only marginally applicable to treatment wetlands (see Appendix B).

The dimensionless parameter that characterizes Equation 6.37 is the Peclet number (Pe), or its inverse, the wetland dispersion number ($\mathfrak{D}$):

$$\mathfrak{D} = \frac{D}{uL} = \frac{1}{\mathrm{Pe}} \tag{6.38}$$

where

Pe = Peclet number, dimensionless
$\mathfrak{D}$ = wetland dispersion number, dimensionless
$L$ = distance from inlet to outlet, m

The two results of interest from modeling of the pulse test are the tracer detention time and the dimensionless variance:

$$\tau = \frac{L}{u} \tag{6.39}$$

$$\sigma_\theta^2 = 2\mathfrak{D} - 2\mathfrak{D}^2(1 - e^{-1/\mathfrak{D}}) \tag{6.40}$$

The principal problems with application of the PFD model to wetlands have to do with meeting the assumptions implicit in the model. Levenspiel (1972) notes:

> In trying to account for large extents of backmixing with the dispersion model we meet with numerous difficulties. With increased axial dispersion it becomes increasingly unlikely that the assumptions of the dispersion model will be satisfied by the real system.

The condition of an intermediate amount of axial dispersion is nominally taken to be $D/uL > 0.025$ (Levenspiel, 1972), which corresponds to about twenty TIS. Therefore, on average, FWS wetlands are not within the acceptable mixing range, but SSF systems may be marginally within range (see Tables 6.1–6.2). However, a bigger obstacle to accepting the PFD model consists of the concentration profiles that are predicted for reactive constituents, which will be addressed in Section 6.6.

## 6.6 REACTION RATE MODELS

In this section, the concepts of local pollutant reduction are blended with wetland hydraulic considerations, and environmental and ecosystems features, to develop pollutant removal models.

### Intrinsic Chemistry

The removal of a contaminant may depend upon the local concentrations of that contaminant in any of a number of ways, depending upon the mechanism(s) or pathways involved. Additionally, other substances may be involved in the conversion process.

### Zero Order

The most simplistic quantitative model for contaminant reduction is a constant rate of removal, termed *zero-order removal*, because it does not depend upon how much of the contaminant is present at a given location. The local load removal is given by

$$J = \text{constant} \tag{6.41}$$

where

$J$ = removal per unit area, or load removed, $g/m^2 \cdot d$

Such a constant rate of consumption has been postulated in only a few instances for treatment wetland situations (Seidel, 1966; Horne, 1995; Kadlec and Srinivasan, 1995).

### First Order

As seen earlier in this chapter, many individual wetland processes are basically first order, such as mass transport, volatilization, sedimentation, and sorption. It is, therefore, not unreasonable to presume that these behave similarly in combination, at least over some range of pollutant concentration. The local removal rate equation is:

$$J = kC \tag{6.42}$$

where
$C$ = concentration, g/m$^3$
$J$ = removal per unit area, or load removed, g/m$^2$·d
$k$ = rate coefficient, m/d

This rate equation is the most prevalent in treatment wetland literature, although in many instances, it is only presumptively advocated.

### Saturation: Monod

Many biologically mediated reactions are first order only for concentrations lower than a saturation value. The premise is based upon the limited ability of the biological community to respond to increases in chemical availability, and this concept is implemented in models of other wastewater treatment technologies (Metcalf and Eddy Inc. 1991). Such a model interpolates between zero and first-order limits:

$$J = k\left(\frac{C}{K + C}\right) \tag{6.43}$$

where
$K$ = half-saturation constant, g/m$^3$

For low values of $C << K$, this is a first-order model. For high values of $C >> K$, it is a zero-order model. This model has been infrequently implemented for treatment wetlands. Kadlec (1997a) reported that this model was appropriate for phosphorus removal in FWS wetlands, with a half-saturation concentration of 0.8 mg/L. Mitchell and McNevin (2001) suggested that a Monod model was appropriate for BOD removal, but did not offer any calibrations or half-saturation concentrations.

### Limiting Reactants

Some removal processes require a second reactant to achieve the transformation or removal process. Nitrification requires oxygen, and denitrification requires a carbon source. In such cases, removal rates may be limited by the supply of the second reactant, in addition to the concentration of the contaminant in question. Other treatment technologies utilize removal models that incorporate such supply limitations. For instance, for microbial nitrification

$$k = k_{max}\left(\frac{C_{DO}}{K_{DO} + C_{DO}}\right) \tag{6.44}$$

where
$K_{DO}$ = dissolved oxygen half-saturation constant, g/m$^3$

The value of $K_{DO}$ is suggested to be about 1.0 mg/L (U.S. EPA, 1993b). In a similar fashion, the carbon limitation for denitrification may be expressed as

$$k = k_{max}\left(\frac{C_{OrgC}}{K_{OrgC} + C_{OrgC}}\right) \tag{6.45}$$

where
$K_{OrgC}$ = organic carbon half-saturation constant, g/m$^3$

The value of $K_{OrgC}$ is suggested to be about 0.1 mg/L (U.S. EPA, 1993b). As a result, the effect of the carbon supply is not large, unless that supply is very low.

In virtually all of the treatment wetland literature, supply limitations are implicit in the overall rate constants that are reported. Nonetheless, there are some calibrations available for specific situations (McBride and Tanner, 2000; Langergraber, 2001).

#### *Return Fluxes and Background Concentrations*

For many chemicals, the return rate to the water from the static compartments of the ecosystem—the soils and biomass—can be a significant (negative) contribution to the net rate. There is at present no scientific study to provide guidance on modeling this transfer. Therefore, the simplest option is used here: a constant (zero-order) return rate. The lumped rate equation for the net reduction of a chemical with no precursors is therefore written as

$$J = kC - r^* = k(C - C^*) \tag{6.46}$$

where
$C^*$ = background concentration, g/m$^3$
$k$ = removal rate constant, m/d (or with unit conversion, m/yr)
$r^*$ = return rate of chemical, g/m$^2$·d

In the terminology of reaction engineering, the model is first order in the forward direction, and zero order in the reverse direction. The concentration $C^*$ is achieved when there is no net uptake or conversion of the chemical in question, and is therefore termed the "background" concentration. When inlet waters have $C > C^*$, there will be a decrease with travel or time to this background concentration. When inlet waters have $C < C^*$, there will be an increase up to this background concentration.

There are several possible reasons for the existence of a real or apparent nonzero background concentration for a

specific chemical constituent. First, there may be some portion of the incoming chemical that is resistant to storage or conversion in the wetland environment. This is particularly possible when the concentration measures a lumped set of species, one or more of which may be resistant to degradation in the wetland. For instance, total phosphorus in water may exist as particulate and dissolved, organic, and inorganic forms. Some portion of the organic phosphorus may be highly resistant to uptake by the biogeochemical cycle. An extreme example would be the organophosphate pesticide diazinon, which is not efficiently degraded in wetlands. However, more benign sources may contain a biologically unavailable fraction, by virtue of the size and character of the molecules embodying the phosphorus (Proctor *et al.*, 1999). Such phosphorus fractions may pass through the system untouched.

The second reason for a nonzero background concentration is the association of the chemical with particulates. Because the chemical is associated with (sorbed or incorporated in the structure) suspended particulate matter, a nonzero background level of TSS entails a nonzero background level of the chemical. For instance, at the Des Plaines, Illinois, wetlands, the export of 8 mg/L of TSS carried 16 μg/L of phosphorus, which is due to a phosphorus content of 0.2% in the exported TSS. Although TSS is notoriously difficult to measure inside wetlands, background levels of 5–10 mg/L are commonly found in densely vegetated systems (see Chapter 7).

The third reason is a set of wetland processes that provide inputs distributed across the entire areal extent of the system. Groundwater discharge and rainfall may bring a specific compound into all portions of a wetland (Raisin *et al.*, 1999). The chemical may be utilized in the biogeochemical cycle, which is also distributed across the entire wetland area. That same cycle can produce return of the substance to the water column, usually by the processes of decomposition and leaching (Kadlec, 1997a).

Fourth, there is seasonality. Dry seasons may be accompanied by loss of surface water, and dry-out of the surface sediments in the wetland. The organics in the surface sediments may then be oxidized, resulting in the mineralization of previously organic-bound substances. Upon subsequent rewetting, these mineralized materials may dissolve and contribute to surface water concentrations.

Another factor influencing concentration gradients, and the possibility of plateaus, is hydraulic bypass of the reactive wetland environment. Bypassed water carries with it the inlet substances, which may reblend with treated water at downstream wetland locations (Kadlec, 2000). This process cannot create a true plateau or background, but may easily lead to an inferred background concentration, derived from extrapolation of gradients in the upstream portion of the system. For some chemicals, very few treatment wetlands extend beyond the zone of total containment, and such extrapolation, via curve fitting, is the norm rather than the exception. As shown in Kadlec (2000), the nature of internal flow patterns leads to a data-fitted background concentration, which varies strongly with the hydraulic loading rate to the wetland. Higher loading rates lead to higher background concentrations for flow through wetlands. In this chapter, methods will be set forth to minimize this effect.

## Batch versus Flow Systems

There is a strong but incorrect presumption often made that batch and continuous flow wetland systems are equivalent if travel time is exchanged for batch time. There are two potential reasons for major differences: internal hydraulics and ecosystem gradients.

A batch system will tend to be spatially uniform. The concepts of short-circuiting and dead zones do not apply. The components of the ecosystem, including plants, algae, and microbes, are exposed to a time-changing water chemistry, which may foster time variable consortia of microbes (Stein *et al.*, 2003). Consequently, the hydraulic model is always presumed to be a well-mixed batch. In laboratory mesocosm environments, the water mass balance often is not influenced by water losses or gains. In field situations, the hydraulic efficiency is 100%, because the entire wetland is filled with water. Full-scale batch treatment wetlands operated at Humboldt, Saskatchewan (Lakhsman, 1981), and continue to be operated by Ducks Unlimited Canada at Oak Hammock, Manitoba. For a case of no water losses or gains, the batch contaminant mass balance is

$$V\frac{dC}{dt} = JA = kA(C - C^*) \tag{6.47}$$

where

$A$ = wetland area, $m^2$

Over a given time period, this mass balance integrates to

$$\frac{(C - C^*)}{(C_i - C^*)} = \exp\left(-\frac{kt}{h}\right) \tag{6.48}$$

where

$C_i$ = starting concentration, $g/m^3$

$h$ = wetland free water depth, m

This model has been calibrated for batch microcosms, and the ramifications of different statistical fitting procedures discussed by Stein *et al.* (2006b).

### *Continuous Flow Wetlands*

A continuous flow system will not be spatially uniform. Plants, algae, and microbes vary in type and density along the path of water travel. The concepts of short-circuiting and dead zones do apply. The hydraulic model must account for these effects—through the use of the TIS model, for instance.

## The TIS Model

Water passes through $N$ tanks in series, and loses contaminant in each (Figure 6.18). For the case of no water losses or gains,

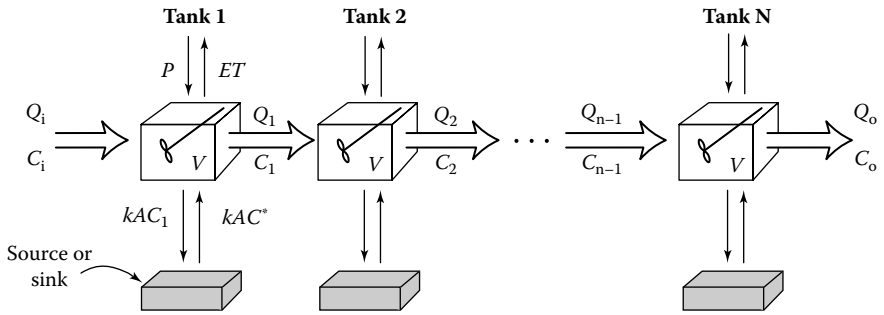



**FIGURE 6.18** The tanks-in-series (TIS) model for wetland hydraulics and contaminant removal.

the steady flow contaminant mass balance for the *j*th tank is

$$(QC_{j-1} - QC_j) = kA(C_j - C^*) \tag{6.49}$$

where
$C_j$ = concentration in and leaving tank $j$, g/m$^3$

For the entire sequence of tanks, these mass balances combine to

$$\frac{(C - C^*)}{(C_i - C^*)} = \left(1 + \frac{k\tau}{Nh}\right)^{-N} \tag{6.50}$$

Note that there are two reaction parameters in this model: the rate constant ($k$) and the hydraulic parameter ($N$).

In Equation 6.50, it has been presumed that the rate constant ($k$) does not vary with the time of exposure to the wetland. This is typically not the case for mixtures of contaminants, such as BOD, and modification is required for such situations, as will subsequently be set forth.

## The PFD Model

The first-order concentration reduction produced by the PFD model is well known (see, e.g., Fogler, 1992):

$$\frac{(C - C^*)}{(C_i - C^*)} = \frac{4b \exp\left(\frac{\text{Pe}}{2}\right)}{(1+b)^2 \exp\left(\frac{b\text{Pe}}{2}\right) - (1-b)^2 \exp\left(\frac{-b\text{Pe}}{2}\right)} \tag{6.51}$$

$$\text{(a)} \qquad b = \sqrt{1 + 4\frac{Da}{\text{Pe}}}$$

$$\text{(b)} \qquad Da = \frac{k\tau}{h} \tag{6.52}$$

$$\text{(c)} \qquad \text{Pe} = \frac{uL}{D}$$

where
$D$ = dispersion coefficient, m$^2$/d
$L$ = wetland length, m
$u$ = average water velocity, m/d

Note that there are also two reaction parameters in this model: the rate constant ($k$) and the dispersion coefficient ($D$).

Although the PFD model has been advocated for wetlands (e.g., Pardue *et al.*, 2000), it is doubtful that it is the most appropriate model of comparable complexity. The DTDs for FWS wetland systems are characterized by a large amount of apparent dispersion, with $0.07 \le D/uL \le 0.35$ (Kadlec, 1994a). The PFD model is not suitable under those circumstances (Levenspiel, 1995). The dispersion coefficient describes eddy transport of water elements both upstream and downstream. In FWS wetlands, such mixing may not occur because flow is often predominantly laminar.

## Longitudinal Profiles

Equations 6.50 and 6.51 represent the input–output concentration behavior of different hydraulic models with first-order rate expressions. Longitudinal concentration profiles may also be derived from these models. In the case of the TIS model, the result is:

$$\frac{(C - C^*)}{(C_i - C^*)} = \left(1 + \frac{k\tau y}{Nh}\right)^{-N} \tag{6.53}$$

where
$y$ = fractional distance through the wetland, dimensionless

This profile is a smoothly decreasing concentration that starts at the inlet concentration and levels off at a plateau value of $C^*$. In theory, the parameters of the model could be determined from an analysis of longitudinal transect data for $C$ and $y$. In practice, there are nearly insurmountable difficulties that arise from three principal reasons. First, if there is time variability in the inlet concentration, profiles reflect that effect, together with a transport delay. In such a dynamic situation, profile data must be averaged over a sufficiently long period to determine mean behavior. Second, it is typically impossible to determine where to take a sample across the width of the wetland so that it is spatially representative. There is usually a bias toward open water areas, because of the ease of obtaining the sample. However, such open water is often a preferential flow path (short-circuit). This effect has been clearly elucidated by (Dierberg *et al.*, 2005).

It is easily seen that any given sample location may be either inside or outside of a preferential flow path (see Figure 6.12). Third, it is not enough to collect a set of spatially uniform samples in an effort to gain access to all flow paths. Spatially uniform sampling across the flow direction will produce an average that is substantially different from the mean (flow-weighted) concentration at that distance (Levenspiel and Turner, 1970; Kadlec, 1999d).

If there is a reasonably long period of averaging, the first obstacle may be overcome. The second and third obstacles may be eliminated by sampling of deep zones perpendicular to flow along the transect, but there is no guarantee. Figure 6.19a shows such deep zone sampling for ammonia, averaged over half a year at Sacramento, California. Decreasing profiles result that display a smooth decline, but there is an abrupt change at the system outlet, indicating some remaining difficulties. Figure 6.19b shows such deep-zone sampling for oxidized nitrogen, averaged over two years at Tres Rios, Arizona. On average, there is a slight increase in the inlet region of the wetland that may be partially the result of nitrification of the very small amount of ammonia that enters. However, individual profiles vary greatly, and it would be misleading to attempt model calibration from any one profile.

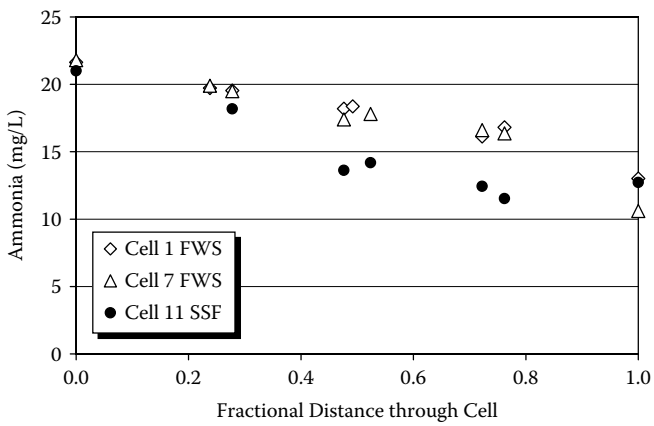



**FIGURE 6.19** Examples of transect data for treatment wetlands. (a) Ammonia nitrogen for the Sacramento wetlands, spanning spring and summer 1997 (Data from Nolte and Associates, 1998a). (b) Oxidized nitrogen for the Tres Rios FWS wetland cell H1. The mean represents 32 transects over two years, and two extremes are shown. (Unpublished data from city of Phoenix.)

Longitudinal profiles may be used to test the validity of alternative modeling assumptions. For instance, the PFD model forecasts the concentration profile through the wetland to be given by

$$\frac{C(x)}{C_i} = \frac{2\exp\left(\frac{xPe}{2}\right)\left[(1+b)\exp\left(\frac{+bPe(1-x)}{2}\right)-(1-b)\exp\left(\frac{-bPe(1-x)}{2}\right)\right]}{(1+b)^2\exp\left(\frac{bPe}{2}\right)-(1-b)^2\exp\left(\frac{-bPe}{2}\right)} \tag{6.54}$$

where

$x$ = distance in flow direction, wetland, m
$C(x)$ = concentration at length $x$, g/m$^3$

Interestingly, this predicted profile has not been examined in any of the literature pertaining to applications to ponds or wetlands. However, in a treatment wetland, this model conceives of swirls, which cannot move back into the inlet distribution works, nor move forward into the outlet collection works. Consequently, the longitudinal concentration profile is predicted to display an instantaneous drop at the wetland inlet. For $D/uL = 0.2$ and $k\tau/h = 3$, the decrease at the inlet is 30%. This unrealistically large concentration drop has not been observed in practice, and hence the PFD model is not an acceptable alternative.

## Mixtures, Weathering, and the *P-k-C** Model

Equation 6.50 represents the reduction of a single compound on transit through a treatment wetland. However, many contaminants are, in fact, mixtures. In almost all instances, water quality parameters are measured by procedures that lump individual chemical compounds into an overall or total concentration for that class of materials. BOD and TSS are examples of such lumping. It is clear that the individual components of such mixtures may be degraded or removed at different rates, and that there is a corresponding difference in removal rate constants (Crites and Tchobanoglous, 1998; Tchobanoglous *et al.*, 2000; Shepherd *et al.*, 2001; Kadlec, 2003a). There is, therefore, a distribution of rate constants across the various mass fractions of the mixture. Such a distribution may be discrete, in the case of a countable and very small number of individual compounds, or it may be continuous in the case of a very large and possibly uncountable number of constituents. Combinations of both types of distribution are also common, such as for total nitrogen (TN). TN consists of a few separately identifiable compounds (nitrate, ammonia) and lumped classes of compounds (particulate N, organic N). Total phosphorus (TP) comprises particulate (PP), dissolved organic (DOP), and soluble reactive (SRP) forms. As water containing such a mixture passes through the wetland, its composition changes because different fractions of the mixture are

reduced at different rates. The mixture becomes *weathered*, a term coined to describe the selective stripping of light volatile materials upon exposure to outdoor environments.

Each fraction of the lumped material will, in general, possess its own *k*-value. Therefore, there is a distribution of *k*-values, designated by $f(k)$:

$$f(k)dk = \text{mass fraction of material with rate constant in the range } k \text{ to } k + dk \quad (6.55)$$

This *k*-value frequency distribution across the mass fractions of the lumped material is termed the *k*VD. It may also be shown for gamma distributions of *k*-values (Kadlec, 2003a) that the average *k*-value at any time during the reduction process is

$$k = \frac{k_i}{(1+\beta t)^n} \quad (6.56)$$

where

$k$ = rate constant during the weathering process, m/yr
$k_i$ = inlet rate constant, m/yr
$n$ = mixture *k*-value distribution breadth parameter, dimensionless
$t$ = length of time the mixture has weathered in the wetland, d
$\beta$ = mixture *k*-value distribution weathering parameter, $d^{-1}$

In the wetland environment, the DVD and the *k*VD interact to produce the overall observed reduction in a lumped category of pollutants. However, batch testing eliminates the DVD effect as there is no distribution function for batch time. The DVD effect is also removed in theory for the (unachievable) ideal of true plug flow.

It has been noted that observed weathering behavior in real wetland situations may be represented by the TIS model Equation 6.50, wherein the parameter values are relaxed to become fitting parameters (Kadlec, 2003a). The *relaxed TIS concentration model* is, therefore, defined to be

$$\left(\frac{C-C^*}{C_i-C^*}\right) = \frac{1}{(1+k/Pq)^P} = \frac{1}{(1+k_V\tau/P)^P} \quad (6.57)$$

**TABLE 6.3**
**Apparent TIS Numbers (*P*-Value) for Hypothetical Double Continuous Distributions**

| | $n = 1$ | $n = 4$ | $n = 8$ | $n = \infty$ |
|---|---|---|---|---|
| $N = 1.5$ | 0.54 | 0.99 | 1.13 | 1.50 |
| $N = 2$ | 0.68 | 1.37 | 1.62 | 2.00 |
| $N = 4$ | 0.86 | 2.19 | 2.86 | 4.00 |
| $N = 8$ | 0.95 | 2.90 | 4.29 | 8.00 |
| $N = \infty$ | 1.00 | 4.00 | 8.00 | — |

*Note:* The relaxed TIS model has been fit to doubly distributed behavior, with $N$ stirred tanks and a *k*VD parameter of $n$.

where

$k$ = modified first-order areal rate constant, m/d
$k_V$ = modified first-order volumetric rate constant, $d^{-1}$
$P$ = apparent number of TIS

A gamma DTD/*k*VD model was used to generate concentration profiles for an incoming $k_o = 0.2\ d^{-1}$, and ranges $1 \le n \le \infty$ and $1.5 \le N \le \infty$ (Table 6.3). That value of $k_o$ is appropriate for nutrients, such as TP or TN. That computer data was then fit with Equation 6.57, generating *k*- and *P*-values for each profile. The value of the rate constant was constant across all distributions, $k = 0.198 \pm 0.006$ (mean ± SD). The apparent number of TIS varied systematically, but always $P \le N$ and $P \le n$ (Table 6.3). In other words, the presence of a weathering mixture will cause a reduction in the *N*-value determined from an inert tracer experiment.

To illustrate the effect of model choice, the progression of concentration reduction along a flow direction is compared for plug flow, *N*TIS, and *P*TIS models (Figure 6.20). Plug flow produces a linear decline on the semi-logarithmic plot for first-order disappearance kinetics. If the hydraulics are represented by four TIS, in a tracer test, and the contaminant experiences first-order decay, the decline is no longer linear, because rates slow at longer detention time because

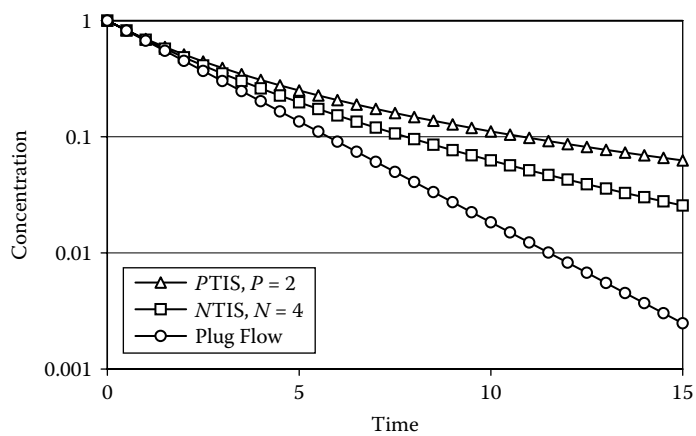



**FIGURE 6.20** Comparison of time progressions along a flow path for three models: plug flow, *N*TIS, and *P*TIS.

of the early exit of some fraction of the material. If, in addition, the material is a mixture that undergoes weathering, the apparent number of tanks is reduced, for instance, to *P*TIS = 2. There is an even greater slowing of the decline because the rate constant decreases as travel time progresses.

As seen in Figure 6.20, there is not much difference in concentration reduction among the various models for low removals, e.g., when removal is less than 50%. However, there is a very large difference when removals are in the high range, and there can be as high as a factor of 10 in concentrations when reductions are above 99%. The specific illustration of Figure 6.20 can be expanded to include a variety of *N*TIS or *P*TIS (see Figure 6.17). It is seen that the achievement of large removals, e.g., 99%, requires very large areas if the parameter *P* is small, e.g., *P* = 1 or 2. This large sensitivity to the *P*-value implies that high pollutant reductions cannot be achieved unless the wetland is hydraulically very efficient.

The first "half" of a wetland performance model is deterministic, and is represented by Equation 6.57. This representation of the *P-k-C** model would be used to describe the central tendency (average performance) of the wetland reactor. Probability distributions represent the second "half" of the mathematical model, which in total is written as

$$C = C^* + \frac{(C_i - C^*)}{(1 + k/Pq)^P} + E \tag{6.58}$$

The random part of the outlet time series is typically given by the intrasystem probability distribution graph or table.

Variability information may also be configured as a multiplier on the deterministic portion of the prediction. Kadlec and Knight (1996) provide such monthly multipliers for the 100% probability for undetrended data for the limited data then available, mostly for lightly loaded wetlands:

$$C(P_\omega = 100\%) = C_{\text{model}} \times F(P_\omega = 100\%) \tag{6.59}$$

where

$C_{\text{model}}$ = deterministic model concentration, mg/L
$F$ = model multiplier to meet probability criterion, unitless
$P_\omega$ = probability that exceedances of frequency ω will not occur
ω = frequency of data averaging (e.g., weekly, monthly, etc.)

It is important to note that the probabilistic portion of the wetland performance model is not a "safety factor," as utilized in some wetland design procedures (Water Environment Federation, 2001). In the usual sense of the term, a safety factor provides extra capacity in design to accommodate unforeseen events and phenomena. The stochastic variability that exists in all treatment wetland outlet concentration data is not unforeseeable. In fact, these probabilistic variations are just as quantifiable as the deterministic variations caused by changes in detention time, and must be accounted in design apart from any considerations of safety factors. It is simply unacceptable to ignore this half of wetland behavior in design, as is the case in U.S. EPA (2000a) and other sources. It is possible to account for probabilistic variations in wetland performance even when using simple loading chart relationships, and this has been done in the wetland literature (Wallace and Knight, 2006; WERF database, 2006).

In subsequent chapters, the multipliers on the seasonal trend are presented for each of the common contaminants and the various wetland types. The random variation, *E*, can be captured through the use of a multiplier on the trend value, $C_{\text{trend}}$:

$$E = C_{\text{trend}} \Psi \tag{6.60}$$

$$C = C_{\text{trend}} (1 + \Psi) \tag{6.61}$$

where

$\Psi$ = stochastic portion of $C/C_{\text{trend}}$, dimensionless

Methods for incorporating stochastic modeling in design are discussed in Part II of this book.

### *Rate Constant Distributions*

The data from any one wetland may be used to calibrate the *P*TIS model. Then, the calibrations from a number of such wetlands may be used to determine the frequency distribution of those *k*-values. Here, the reduction of ammonia in FWS wetlands is used as an example. In Chapter 9, this aspect of nitrogen removal is dealt with in much more detail. Here, the apparent removal of ammonia is used as an illustration of the quantification of frequency distributions of parameter values for wetland models. For purposes of illustration, systems that produce ammonia from mineralization of organic nitrogen are excluded. Often, the *N*-value is not known, because a tracer test has not been run. So, for illustration purposes, it will be assumed that *P* = *N* = 4, which is a mean *N*-value for FWS wetlands (see Table 6.1). The value of *C** is presumptively taken to be zero in this analysis. Equation 6.57 may be used to calculate a long-term average *k*-value from the hydraulic loading as well as the inlet and outlet concentrations averaged over the period of record. By this procedure, temperature effects are lumped into the variability, although it is known that water temperature will be a significant component of the set of conditions which lead to the variability. Each wetland is accorded one value for its entire period of record, thus averaging over a number of annual periods that differs for each wetland.

The probability distribution of these *k*-values for a set of 131 FWS wetlands is quite broad (Figure 6.21). The mean is *k* = 18 m/yr, but the distribution contains some very high rate constants. Accordingly, the median *k* = 11.5 m/yr. However, the range is 0.8–308 m/yr, and the SE of the mean is 3 m/yr (SD = 30 m/yr). Clearly, this distribution is too broad to give much confidence in design for a mean *k*-value. It is obviously necessary to understand the components of the wetland environments and layouts that contribute to either

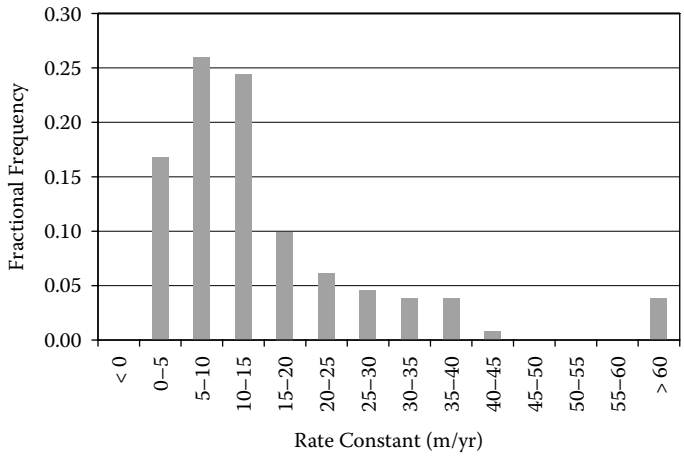



**FIGURE 6.21** Rate constants for ammonia removal in FWS wetlands. The presumption is for 4 TIS. Ammonia-producing systems are excluded. Each of the 131 wetlands is represented by one point, representing its entire period of record. The data were not temperature-corrected.

high or low *k*-values. Very importantly, it would be exceedingly dangerous to place great trust in any one wetland as a prototype for all others. In addition, it would be pure coincidence if any new wetland were to behave as the mean of the distribution.

As a result of these considerations, recommendations for design must go beyond the concept of a universal, or average, *k*-value that may be used for any wetland.

## Data Fitting

Equation 6.57 represents an alternative for quantifying wetland performance data. It has three potentially adjustable parameters, $C^*$, $k$, and $P$. Tracer information gives an upper bound for $P$. The value of $C^*$ does not have to accommodate both hydraulic and biogeochemical plateau effects for long detention times. The value of $P$ is a free-fitting parameter, subject to the constraint of $P < N$, where $N$ is the tracer TIS number. The value of $C^*$ represents the only biogeochemical background, because speciation effects have been removed to the parameter $P$. $C^*$ may be selected in one of two ways. It may be considered a free parameter, constrained by $C^* > 0$, or it may be selected to be the lowest concentration ever measured in a comparable situation, such as at far down-gradient locations in impacted pristine systems.

The best procedure would be to fit a three-parameter model to the data, adjusting $k$, $C^*$, and $P$. However, if the available data cover only a small reduction in the inlet concentration from values well above $C^*$, there is not sufficient information to gain a good estimate of $C^*$. Conversely, if most of the data are in the region near $C^*$, a good estimate of the $k$-value is not possible. These results suggest:

1. For high inlet concentrations ($C_i >> C^*$), it is better to guess $C^*$ and gain good estimates of $k$ and $P$.
2. For low inlet concentrations ($C_i < 3C^*$), it is better to guess $P < N$, and gain good estimates of $k$ and $C^*$.

It is not as onerous as it may seem to independently estimate $C^*$. Data from a wide range of treatment wetlands suggest that virtually all individual chemicals have zero wetland background levels. Exceptions include BOD, COD, organic nitrogen, and pathogens, as will be discussed in subsequent chapters.

Often, the worst fitting procedure is to choose $C^* = 0$, and select $N = \infty$, which is the plug flow model.

### Synoptic Error

There will typically be a set of contemporaneous values of wetland inlet and outlet concentrations, together with accompanying flows and other information. Calibration consists of selection of rate parameters that minimizes the error between those field observations and the calculated model values from Equation 6.57 or an equivalent.

It is clear that paired contemporaneous measurements of input and output flows and concentrations have little chance of providing a quantitatively accurate description of removals, because of the transport delay in the wetland. Typical hydraulic detention times are of the order of several days to more than a week, which implies a significant shift in event timing if the wetland is in plug flow. However, tracer testing of hundreds of treatment wetlands has shown conclusively that no treatment wetland exhibits plug flow; rather, the detention time distribution extends to three or more nominal detention times (Kadlec, 1994a). Therefore, there are remnant effects of inlet events at the outlet after three or more nominal detention times. The only chance of avoiding a transport delay artifact (synoptic error) is to compare inlet and outlet measurements averaged over more than those three nominal detention times.

The water leaving the wetland may have entered anywhere from about a tenth to three or four times the nominal detention time earlier. If there is a time series of changing inlet concentrations or flows, instantaneous, contemporaneous inlet–outlet data should not be used to calibrate the model. Rather, time averages over at least three detention times should be used.

## 6.7 OTHER FACTORS AFFECTING TREATMENT PERFORMANCE

### Definition of the Rate Constant

It is seen that the Damköhler number ($Da = k/q = k\tau/h$) grouping is common to many models, including the TIS and PFD models. There are differences in the interpretation of both $h$ and $\tau$, as well as different assumptions concerning the possible relations among $k$, $h$, $q$, and $\tau$.

### Which Detention Time?

As seen in Chapter 2 and the discussion of tracer testing in this chapter, there may be considerable difference between the nominal detention time and the mean tracer (actual) detention time, exemplified by the volumetric efficiency, $e_V$. Tables 6.1 and 6.2 show that on average, $\tau_{actual} = 0.82\ \tau_n$ for a sampling of FWS wetlands, and $\tau_{actual} = 0.91\ \tau_n$ for a sampling of SSF wetlands. During data analysis, either may be used, if a tracer test is available. The resultant calibrated values of $k$ will differ markedly, depending upon which alternative is chosen. In this book, the nominal detention time is chosen, because tracer testing may not be available to provide the actual detention time. This implies that the volumetric efficiency is absorbed into the rate constant.

### Areal versus Volumetric Rate Constants

A good portion of the treatment wetland literature (Crites and Tchobanoglous, 1998; U.S. EPA, 1999; Water Environment Federation, 2001) utilizes a volumetric rate constant, defined as:

$$k_V = \frac{k}{e_V h} \tag{6.62}$$

Those sources also estimate values of $0.65 < e_V < 0.75$, which are referred to as "porosity" (see Chapter 2). Consequently, $k_V$-values from those sources are based on such $e_V$-values.

Another fundamental difference between the use of $k$ and $k_V$ arises from the assumption of constancy of $k$-values or $k_V$-values, as follows:

- If $k_V$ is assumed constant, deeper water (greater $h$) provides more detention time, without any penalty of reduction of the value of $Da$ for the system.
- If $k$ is assumed constant, deeper water (greater $h$) provides more detention time, but with a penalty of reduction of the value of $Da$ for the system which compensates for, and removes the advantage of, the greater deep water detention.

Side-by-side studies have been conducted for both FWS and HSSF wetlands, in an effort to investigate the effect of depth increases at constant flow. Analyses of data from FWS studies at Arcata, California, show that $k$ was nearly constant, and therefore $k_V$ decreased with increasing depth (data in Gearheart *et al.*, 1983). As the depth, and hence detention, increased, the calculated $k_V$ decreased proportionately for BOD and fecal coliforms (Table 6.4). Side-by-side studies of HSSF wetlands at Les Franqueses del Valles, near Barcelona, Spain (García, 2003; García *et al.*, 2004b), produced removals that increased with decreasing depth (Table 6.5), thereby indicating that $k_V$-values would also increase with decreasing depth. Indeed, the García *et al.* (2004b) data produced *areal* $k$-values that increased with decreasing depth, ranging 22–33 m/yr for 27 cm depth, and 4–7 m/yr for 50 cm depth. García *et al.* (2004b) attributed these effects to differences in the chemical environment created by shallower water, in particular, the occurrence of denitrification in shallow systems.

**TABLE 6.4**
**Plug Flow Removal Rate Constants in Side-by-Side FWS Wetlands Operated at the Same Hydraulic Loadings**

| Flow ($m^3/d$) | Depth (m) | Increase in HRT (%) | BOD PF Rate Constant ($d^{-1}$) | Decrease in BOD $k_V$ (%) | Fecal Coliform PF Rate Constant ($d^{-1}$) | Decrease in FC $k_V$ (%) |
|---|---|---|---|---|---|---|
| 93 | 0.40 | — | 0.29 | — | 1.33 | — |
| 94 | 0.55 | 37 | 0.17 | 42 | 0.62 | 53 |
| 86 | 0.36 | — | 0.25 | — | 0.83 | — |
| 83 | 0.61 | 76 | 0.13 | 49 | 0.51 | 38 |
| 45 | 0.30 | — | 0.28 | — | 0.83 | — |
| 49 | 0.49 | 49 | 0.14 | 48 | 0.41 | 51 |
| 29 | 0.33 | — | 0.14 | — | 0.36 | — |
| 29 | 0.53 | 78 | 0.08 | 40 | 0.32 | 10 |
| 23 | 0.35 | — | 0.14 | — | 0.53 | — |
| 24 | 0.50 | 39 | 0.09 | 36 | 0.38 | 28 |

*Source:* Data from Gearheart *et al.* (1983) *City of Arcata Marsh Pilot Project, effluent quality results—system design and management*. Final Report to the North Coast Regional Water Quality Board (Santa Rosa, California) and State Water Resources Board (Sacramento, California).

**TABLE 6.5**
**Percent Removals in Side-by-Side SSF Wetlands Operated at the Same Hydraulic Loadings**

| Parameter | Unit | Deep (50 cm) | Shallow (27 cm) |
|---|---|---|---|
| COD | mg/L | 63.5 ± 1.4 | 74.5 ± 6.4 |
| $BOD_5$ | mg/L | 56.5 ± 2.7 | 77.5 ± 9.2 |
| Ammonia | mg/L | 26.5 ± 2.3 | 44.5 ± 9.2 |
| Dissolved Reactive Phosphorus | mg/L | 5.2 ± 3.1 | 16.0 ± 8.5 |
| Relative HLR | — | 1.00 | 1.00 |
| Relative HRT | — | 1.00 | 0.54 |

*Note:* The shallow systems do much better, despite the fact that they have less detention time.

*Source:* Data from García *et al.* (2004b) *Ecological Engineering* 23(3): 177–187.

In FWS wetlands, the biogeochemical processes, which remove and sequester pollutants, are closely associated with plants, biofilms, and sediment interfaces. In total, these represent a "biomachine" that processes contaminants and nutrients. Such action zones are typically apportioned to wetland area to a greater extent than to wetland water volume. Thus, when the wetland area is doubled at constant depth, the detention time is doubled, and the biomachine is doubled. But, when the water depth is doubled at constant area, there are not more plants, and biofilms and interfaces do not necessarily increase in proportion to water depth (Figure 6.22). In fact, it is often not possible to deepen the water and retain the same ecology, because of the hydropattern requirements of vegetation (see Chapter 3).

In contrast, properly designed SSF wetlands can support vegetation at virtually any water depth, because the plants are situated on top of the media and root into the water just below the surface. The media also provide the majority of the interfacial and biofilm area, which is in proportion to water depth. However, plant roots typically do not penetrate more than 20–30 cm, as discussed earlier in this chapter. There is debate over how much effect plants and their roots have on treatment potential, ranging from the suggestion that they completely control the degree of treatment (Reed *et al.*, 1995) to the idea that they have no effect at all (Langergraber, 2001).

Either $k$ or $k_V$ can be used to represent a data set or be used in design. However, the use of $k_V$ for FWS or HSSF wetlands requires the accompanying information on water depth ($h$) in the sizing equations because of the high probability of important depth dependence. This depth dependence usually means that more detention time created by deeper water is counteracted by a decrease in the volumetric rate constant. Conversely, data indicate that (the areal) $k$ is nearly independent of depth. Nonetheless, data analysis and design require knowledge of the water depth, because of the implied changes in the type of vegetation that may be fostered and maintained.

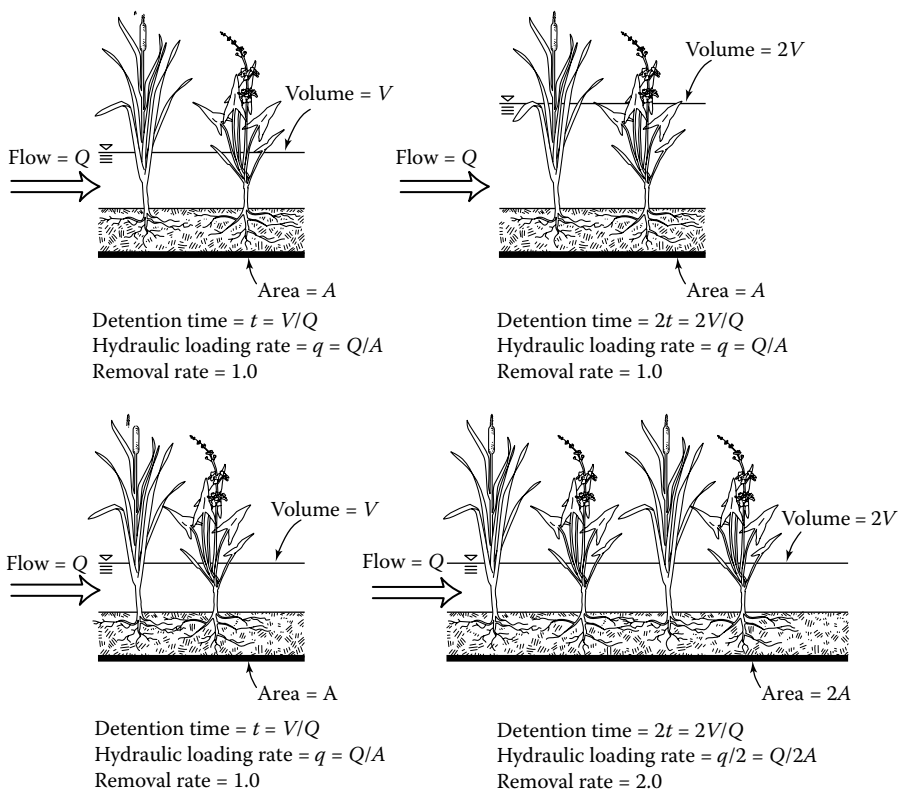



**FIGURE 6.22** Conceptual distinction between increasing detention time with deeper water and with more area.

### Temperature and Season

Temperature effects on $k$ or $k_V$ have often been summarized by use of the modified Arrhenius equation:

$$k_T = k_{20}\,\theta^{(T-20)} \tag{6.63}$$

where $k_T$ is the rate constant at temperature $T = T$°C, and $k_{20}$ is the rate constant at 20°C. Values of the temperature correction factor (θ) may be estimated for data sets with adequate operational temperature data. However, it should be noted that incorrect historical presumptions still pervade the treatment wetland literature.

Microbial processes have been studied in several types of "conventional" wastewater treatment devices, including activated sludge plants and trickling filters. The Arrhenius factors for some contaminant reductions have been documented for those situations (see, e.g., Metcalf and Eddy, Inc., 1991). This prior information for other technologies lead early wetland workers to assume that the temperature coefficients would be the same for wetlands, which they viewed as microbial reactors. For instance, a BOD temperature coefficient of θ = 1.06 was *assumed* by Reed *et al.* (1995), and has been propagated in other subsequent publications (Crites and Tchobanoglous, 1998; Water Environment Federation, 2001; Crites *et al.*, 2006). However, examination of treatment wetland data (and other natural system data) does not yield such a value for BOD (Kadlec and Reddy, 2001). For 23 FWS wetlands, they found an average θ = 0.983; for three ponds, θ = 1.005; and for two overland flow systems, θ = 1.012. The situation in wetlands is apparently more complicated than just microbial processing.

Many of the variables that go into a mechanistic BOD model are temperature-dependent, such as diffusion coefficients and the biofilm rate constant ($k_b$). However, the apparent rate constant ($k$) is a combination of those parent variables, and therefore exhibits a different temperature dependence. The theta model is used here to explore the consequences. For illustration, assume:

1. The rate constant $k_b = k_{b20}\,\theta^{(T-20)}$. The value of θ = 1.05, after Polprasert and Agarwalla (1994).
2. The diffusion coefficients have $\theta_w = 1.025$ and $\theta_b = 1.00$, after Polprasert and Agarwalla (1994).
3. The relative amounts of biofilm surfaces are greater in winter, after litterfall has occurred. Assume there is 25% more litter during winter than spring, and 25% less litter during summer.
4. Climatic conditions give a winter temperature of 5°C and a summer temperature of 25°C.
5. The values of the 20°C parameters are those determined by Polprasert and Agarwalla (1994).

The resulting θ-value for the overall first-order areal uptake coefficient ($k$) is then 0.999. The temperature dependencies cancel each other.

The situation for nutrients is even more complex because light nutrient loadings are strongly influenced by plant and algal uptake on a seasonal basis. Microbial activity follows an annual pattern with a peak in midsummer, but uptake peaks earlier, in spring (Figure 6.23). To the extent that growth has first claim on nutrients, removal will be out of phase with the annual temperature cycle. The loading region sensitive to growth uptake has been postulated to be about 120 g/m²·yr for nitrogen (Kadlec, 2005d). Phosphorus removal would be expected to be in phase with growth, since microbial conversions do not remove total phosphorus. When growth effects are dominant, the $k$-values change seasonally:

$$k = k_j \quad j = 1, 2, 3, \ldots 12 \tag{6.64}$$

where $j$ is the month number. Although higher frequency might be used for the seasonal change pattern, monthly values will be sufficient in cases where detention time lags are on the order of a week.

### Variability and Data Folding

Many factors contribute to random variability in the outlet concentrations from treatment wetlands. This variability is

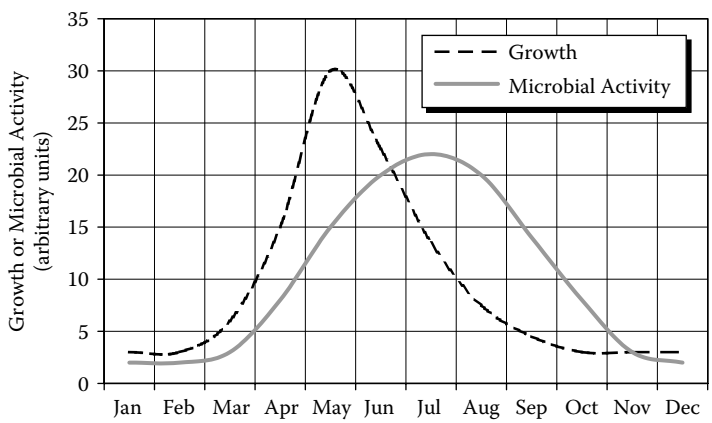



**FIGURE 6.23** Annual patterns of growth and microbial activity in a temperate climate. Growth peaks in spring, when temperatures and microbial activity are at moderate levels.

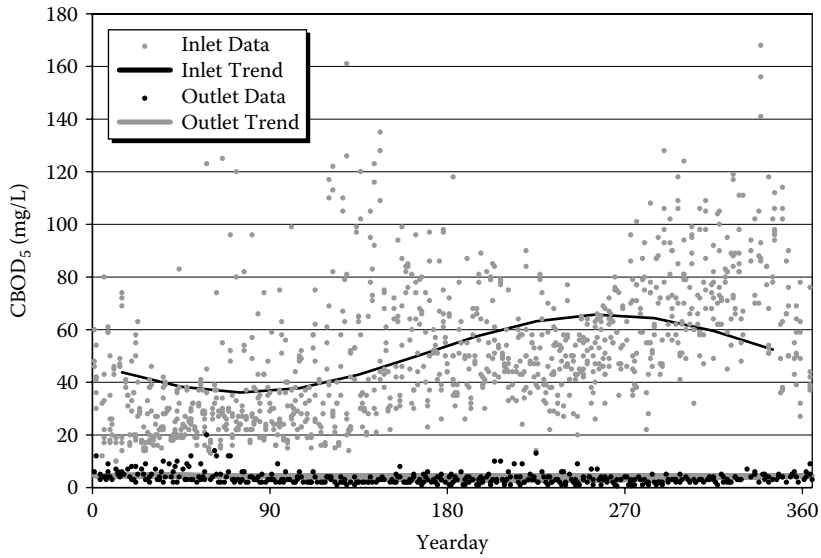



**FIGURE 6.24** Annualized progression of $CBOD_5$ in the Arcata, California, wetland system. The inlet is to the treatment wetlands, and the outlet is from the enhancement wetlands. The period of record is 1992–1999. (Data from TWDB database (2000) *Treatment Wetland Database (TWDB).* Website developed for U.S. EPA. http://firehole.humboldt.edu/wetland/twdb.html. Last updated November 2000. Compiled by B. Finney. U.S. EPA: Washington, D.C.)

typically not small, with coefficients of variation of 20%–60% being common. The deterministic models presented above reproduce the central tendency of performance, but not the random variability. Whether there is microbial or vegetative control, seasonal patterns of wetland variables are the rule, accompanied by a random variable term (Kadlec, 1999a).

$$C = C_{avg}\left[1 + A \cdot \cos\left(\omega(t - t_{max})\right)\right] + E \qquad (6.65)$$

where

$A$ = fractional amplitude of the seasonal cycle, dimensionless
$C$ = instantaneous outlet concentration, mg/L
$C_{avg}$ = average (trend) outlet concentration, mg/L
$E$ = random portion of the outlet concentration, mg/L
$t$ = time of the year, Julian day
$t_{max}$ = time of the year for the maximum outlet concentration, Julian day

The deterministic portion of this representation may also be modeled by the *k*-rate technique with appropriate rate constants and background concentrations, both of which may respond to temperature and season. The stochastic portion ($E$) will have a probability distribution, which will be different depending upon sampling frequency and sample averaging period.

The existence of the error term ($E$) means that sampling must either be at high frequency, or cover many annual cycles before meaningful trend averages can be determined. Data from several years may be "folded" to create an annualized grouping, distributed across the year according to Julian day. This use of many annual cycles has the advantage of including year-to-year variations in climate, flow, and ecosystem condition.

As an example, consider the $CBOD_5$ behavior of the Arcata, California, treatment and enhancement marshes (Figures 6.24–6.26). Weekly sampling produces a cloud of data which spans a wide range of values at any particular time of the year (Figure 6.24). Although compressed by the vertical scale required to show inlet concentrations, the variability in the outlet $CBOD_5$ is also quite high. Because the mean outlet concentration over the 1992–1999 period of record was quite low, just under 4 mg/L, there occurs a skewed distribution of departures from the trend behavior (Figure 6.25). This leads to a nearly log-normal distribution of fractional errors, which is to be expected whenever concentrations are limited on the low end by analytical accuracy near detection limits. Because of the luxury of eight years' worth of data, it is possible to determine a *k*-value, which describes the relation between the average inlet and average outlet concentrations. The *P-k-C** model is well calibrated by $P = 8$, $C^* = 3$ mg/L, and $k = 57$ m/yr (Figure 6.26). The high value of $P$ is not unexpected, because there are five wetlands in series in the Arcata system. Thus each wetland unit behaves like less than two tanks in series.

## Water Losses and Gains

In general, literature values of rate constants have not been corrected for water losses and gains. In some instances, water budget information was not collected; in other cases atmospheric losses and gains were not significant. Therefore, water mass balance effects are the cause of some fraction of the variability in rate constant data.

It is readily possible to adjust the *P*TIS model to account for excessive rain or evapotranspiration. The wetland is broken into $P$ segments, corresponding to the model (see Figure 6.18). First, the water mass balance for the first

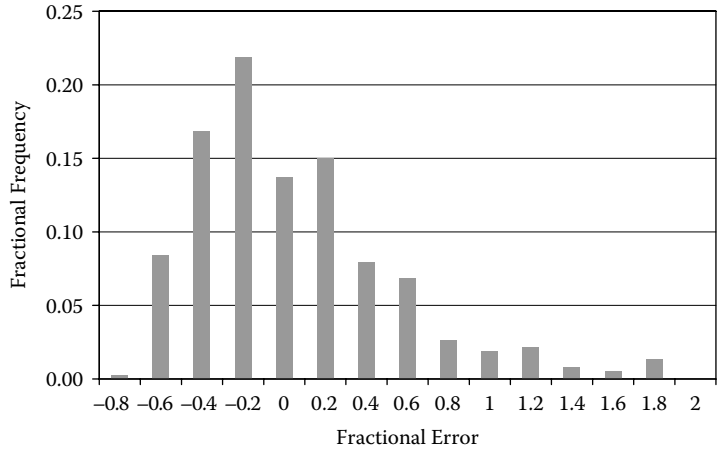



**FIGURE 6.25** Distribution of departures from the annual trend for $CBOD_5$ at the Arcata, California, treatment wetland. The period of record is 1992–1999. (Data from TWDB database (2000) *Treatment Wetland Database (TWDB).* Website developed for U.S. EPA. http://firehole.humboldt.edu/wetland/twdb.html. Last updated November 2000. Compiled by B. Finney. U.S. EPA: Washington, D.C.)

of the wetland segments, designated by the subscript "1," for steady-state, nonuniform flow is

$$Q_1 = Q_i + A_1 \cdot (P - ET - I) \tag{6.66}$$

where

$A_1$ = area of the first segment (tank), $m^2$
$ET$ = evapotranspiration, m/d
$I$ = infiltration, m/d
$P$ = precipitation, m/d
$Q_i$ = inlet flow rate, $m^3/d$
$Q_1$ = outlet flow rate from segment #1, $m^3/d$

The pollutant mass balance for the same first segment, for steady-state, nonuniform flow is:

$$Q_1 C_1 = Q_i C_i - (I \cdot A_1 C_1) - (k \cdot A_1 \cdot (C_1 - C^*)) \tag{6.67}$$

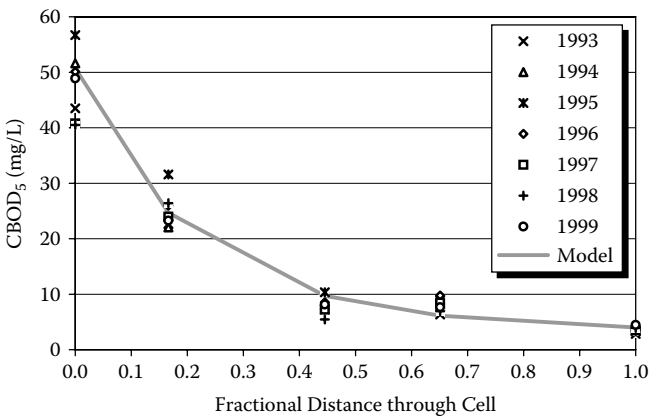



**FIGURE 6.26** Progression of $CBOD_5$ through the Arcata, California, treatment wetland complex, including both treatment and enhancement wetland systems. The period of record is 1992–1999. (Data from TWDB database (2000) *Treatment Wetland Database (TWDB).* Website developed for U.S. EPA. http://firehole.humboldt.edu/wetland/twdb.html. Last updated November 2000. Compiled by B. Finney. U.S. EPA: Washington, D.C.)

where

$C_1$ = concentration out of and in the first segment (tank), mg/L
$C_i$ = concentration into the first segment, mg/L
$C^*$ = background concentration, mg/L

Here, rainfall has been assumed to have a zero pollutant concentration, and infiltration is assumed to occur at the outlet concentration. Combining Equation 6.66 with Equation 6.67 gives the concentration exiting hypothetical segment number one:

$$C_1 = \frac{Q_i C_i + (k \cdot A_1 \cdot C^*)}{Q_i + (A_1 \cdot (P - ET)) + (k \cdot A_1)} \tag{6.68}$$

This computation is then repeated for the remaining segments, in each case using the outlet concentrations and flows from the preceding unit. The wetland outlet concentration is that exiting from the final hypothetical segment.

### Interactions with Solids

The water detention time does not appropriately reflect transport delays due to storage in or on stationary compartments of the wetland: the substrate, sediments, and biota. The contaminant residence time will vary markedly depending on how many times it has been "parked" and recycled in various active or passive storage compartments during its passage through the wetland.

As an example, consider the hypothetical compartmentalization and storages in a wetland for nitrogen (Figure 6.27). The overall nitrogen content of the various solid storages totaled over 200 $gN/m^2$. Nitrogen removal rates were of the order of 2 $gN/m^2 \cdot d$, corresponding to hydraulic detention times of about five days. Therefore, the nominal displacement time for solid phase nitrogen is about 100 days. However, first-order turnover is more likely than displacement, and not all the solid phase nitrogen is available for turnover. Reported turnover times are rapid, of the order of one day, for sorbed ammonium (Sikora *et al.*, 1995;

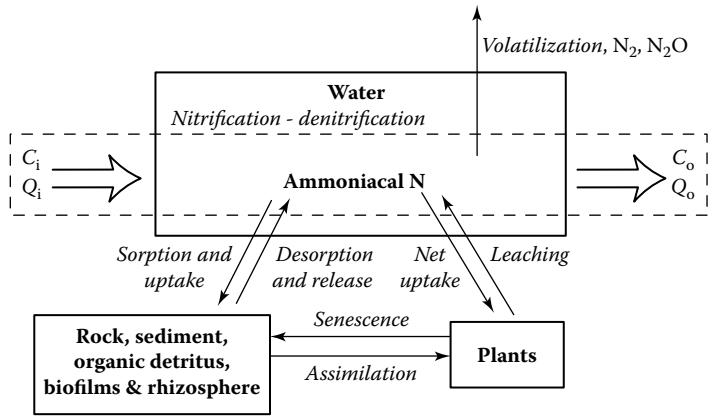



**FIGURE 6.27** Simplified nitrogen processing schematic for high ammonium loading and low removal. (From Kadlec *et al.* (2005) *Ecological Engineering* 25: 365–381. Reprinted with permission.)

Tanner *et al.*, 1999). However, plant detritus turns over much more slowly, with half-lives of up to a year or more (Hietz, 1992; Tanner, 2001a). These approximations for nitrogen storage and fluxes indicate that the nitrogen detention time in the constructed wetland is far greater than the water detention time.

The importance of the large pools of solid phase nitrogen lies in the fact that small percentage changes in those storages can have large consequences for the water concentrations. For instance, the seasonal senescence of biomass is capable of adding back a portion of the nitrogen stored during the growing season (Sartoris *et al.*, 2000; Tanner, 2001a), thus adding an element of seasonality to performance.

A simplified model of nitrogen tracer ($^{15}N$) movements in the wetland environment is possible when some of the more minor intercompartmental transfers are ignored, and the solids compartments are lumped (Kadlec *et al.*, 2005). This model allows for unidirectional plant uptake, reversible exchange with wetland solids, and microbial conversion of ammonium to nitrogen gases. It has been calibrated to treatment of dairy wastewater with high ammonium in gravel bed mesocosms with good success ($R^2 = 0.97–0.99$) (Tanner *et al.*, 1999). However, that calibration was for batch operation, whereas here we are interested in the dynamics of tracer movement through a spatially variable, longitudinal SSF wetland. The plug flow spatial and dynamic equations are:

$$\varepsilon \frac{\partial C}{\partial t} + u \frac{\partial C}{\partial y} = -k_{Vu} C + k_{Vr} X - k_{Vg} C \qquad (6.69)$$

$$(1-\varepsilon) \frac{\partial X}{\partial t} = k_{Vu} C - k_{Vr} X \qquad (6.70)$$

where

$C$ = water phase concentration, g/m$^3$
$k_{Vg}$ = growth and conversion rate constant, d$^{-1}$
$k_{Vr}$ = release rate constant, d$^{-1}$
$k_{Vu}$ = uptake rate constant, d$^{-1}$
$t$ = time, d
$u$ = water velocity, m/d
$X$ = solid phase concentration, g/m$^3$
$y$ = distance, m
$\varepsilon$ = porosity

It should be noted that a volumetric solid phase concentration ($X$) has been used. It is related to the commonly used mass concentration by

$$X = \rho \cdot C_S \qquad (6.71)$$

where

$C_S$ = solid phase concentration, g/kg
$\rho$ = solid density, kg/m$^3$

The model may be implemented in a sequence of segments, thus converting it to a *P*TIS model. Figure 6.28 presents rough calibrations to the field scale wetlands for $P = 5$. About 85% of the incoming tagged ammonia was not converted, and ultimately exited the system. Only a tiny portion of the tagged ammonium nitrogen goes directly to the wetland exit (<1%). The $^{15}N$ detention time for the field scale wetlands is about 50 days, or 12 times the water detention time.

The correct concept for the longitudinal transport of interactive substances, such as nitrogen and phosphorus, includes major interactions with wetland solid compartments. Transport with water is rapid, but the exiting compounds originated from storage, and not from the water that entered only one or a few detention times earlier. Pulses of incoming nutrients must work their way through solid storages. Consequently, there is a second type of synoptic error that must be accounted, which results from the wetland storage delay.

## System Start-Up

The models discussed in the previous text pertain to fully developed treatment wetlands. Plants are ordinarily introduced into constructed treatment wetlands as seeds or propagules at biomass densities far less than those that will ultimately develop. This process of grow-in typically takes a period of one to three years, depending upon climate. The amounts of nutrients taken up during the initial grow-in are substantial in many cases, and represent a removal process that is not sustainable in the long term. Development of a litter layer takes even longer. Because submerged plant material creates attachment substrate for microbes, and a carbon source for metabolism for some of them, the wetland may not

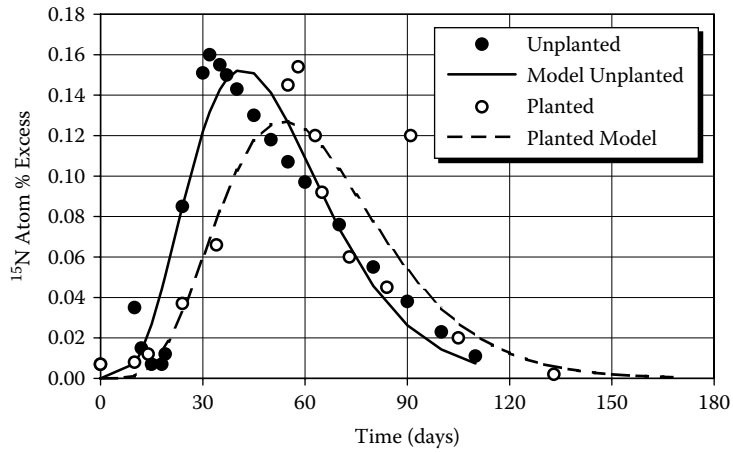



**FIGURE 6.28** Response of planted and unplanted SSF wetlands to an impulse of $^{15}NH_4$. The water detention time in these systems was about five days. (From Kadlec *et al*. (2005) *Ecological Engineering* 25: 365–381. Reprinted with permission.)

be fully functional during start-up. Nonnutrient chemicals are likely to be less affected, but processes such as sorption still require the full ecosystem to reach their potential.

These large ecosystem start-up effects can dominate nutrient removal for many months (Busnardo *et al*., 1992; Tanner *et al*., 1998; Tanner, 2001a). Therefore, calibration of steady-state models during such a period will not produce rate constants that are representative of long-term sustainable operation.

Figure 6.29 shows an illustration of start-up effects for an aerated HSSF designed to remove ammonia from landfill leachate (Nivala, 2005). Upon implementation of wetland aeration, it took the microbial community approximately six months (under winter operating conditions) to adapt to the ammonia loading.

## 6.8 DANGERS OF EXTRAPOLATING WETLAND PERFORMANCE DATA

The models discussed in the previous text are all quite simplistic, in that they purport to describe general features of complicated processes. Many ecosystem compartments are lumped together, and the models are therefore termed "highly aggregated." The decision to use highly aggregated design models (all those under consideration fit that description) carries the implied penalty of great risk in extrapolation beyond the calibration conditions. The safest criterion for extrapolation is to not do it; only interpolation on the calibration sets should be allowed. A less conservative approach would be to avoid extrapolation to conditions that are known to threaten the integrity of the vegetative community that typified the calibration. In any case,

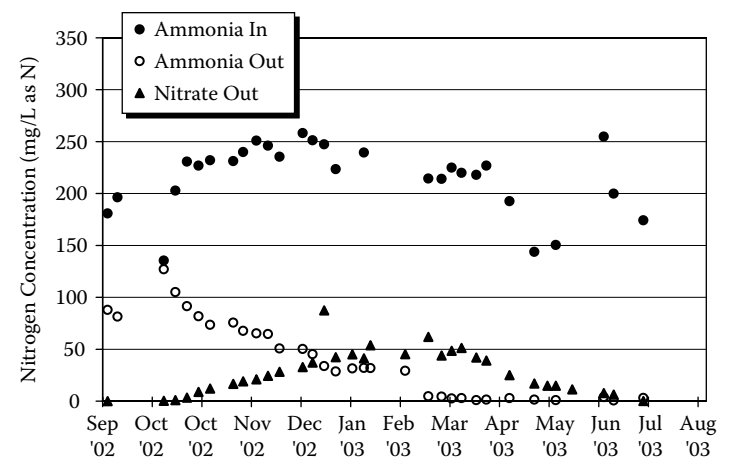



**FIGURE 6.29** Example of start-up effects for ammonia removal in a HSSF wetland. This wetland is mechanically aerated to remove oxygen transfer limitations. Note that effluent ammonia concentrations are declining despite lower water temperatures over the winter months. The hydraulic loading rate was relatively constant over the time period reported. (Data from Nivala (2005) Treatment of landfill leachate using an enhanced subsurface-flow constructed wetland. M.S. Thesis, Department of Civil and Environmental Engineering, University of Iowa, Iowa City.)

extrapolation should not be to water chemistries known to be outside the range of the calibration.

Hydraulic variables present difficulties in two ways: they may affect the performance of a specific vegetative community, and they may affect that community itself. For instance, removal performance may not differ too much from steady to pulsed operation. As long as the vegetation is not altered, we are faced only with the issue of how that ecosystem averages the event-driven environment. Depth variation may not be a large factor within certain ranges if the variations are not of long duration, but average depth is likely to be an important factor. The larger risk is that the ecological communities become affected, and change to communities that do not match the calibration. For example, cattails find it difficult to survive at long hydroperiods in deep water. SAV may suffer if hydroperiods are too short. Prolonged dry-out may pose problems under some design scenarios, or at least affect the subsequent treatment performance.

Internal flow patterns are controlled by topography, depth, and compartmentalization. Pollutant removal is known to be sensitive to those flow patterns. Therefore, the applicability of a specific TIS model is conditioned on the anticipated TIS for the extrapolation. There are very large implications, especially for high removals, as was shown in Figures 6.19 and 6.22.

Extrapolation from a wetland of one type to another is clearly not a reasonable step. The microbial communities, as well as the character and magnitude of the biogeochemical cycles, may differ markedly. Forested ecosystems will not necessarily perform with the same rate constants as emergent marshes, even if the hydrologies are comparable.

As a consequence of these considerations, it is not a good practice to use model parameters ($P$, $k$, $C^*$, $\theta$) in situations outside the ranges of operating values from which they were derived. There are not "universal" values of rate constants, as offered in many literature sources (e.g., Reed *et al.*, 1995; Water Environment Federation, 2001; Crites *et al.*, 2006). Nor should the reported central tendencies of parameters, such as rate constants (Kadlec and Knight, 1996), be interpreted as "universal" values. The parameter values obtained from various operating wetland systems vary widely, depending upon factors described in preceding sections. It is therefore prudent to examine the origin of a particular calibration set ($P$, $k$, $C^*$, $\theta$) before using it in design calculations. The questions to be addressed include:

- Do the inlet and outlet concentration ranges of the calibration set include the ranges to be considered in design?
- Do the ranges of detention times and hydraulic loadings of the calibration set include the ranges to be considered in design? Are the intended water depths in the calibration range?
- Is the configuration of the system under design comparable to that of the calibration wetland(s)? This includes aspect ratio, number of physical compartments, and size. Consideration should be given to the $N$-values (or $P$-values) of calibration and design systems. Further, small mesocosms may not be representative of a large system.
- Are the climatic conditions for the calibration set similar to those of the intended design? Extremes of rainfall, evapotranspiration, and temperature may affect the calibration values. For instance, the seasonal freeze-up of temperate wetlands separates them from subtropical systems.
- Is the ecology of the calibration wetland(s) comparable to that of the intended wetland under design? At a minimum, wetland types must correspond across the variants of emergent marshes, floating plants, submerged aquatic vegetation, and possibly open water fraction.

Insofar as the intended design departs from calibration conditions and information, an increasing degree of risk is engendered. Intersystem data is a valuable aid to quantifying that risk. However, many systems do not possess enough information to gain firm estimates of rate model parameters. Therefore, the risk assessment associated with transferability of available data is better accomplished via other methods of data representation, as will be described in subsequent chapters.

## SUMMARY

The following circumstances often apply to treatment wetland data analysis:

1. Steady flow
2. Negligible water gains and losses
3. Concentrations averaged over several nominal detention times

The central tendencies of wetland outlet concentrations are then often well represented by the $PkC^*$ model:

$$\frac{C - C^*}{C_i - C^*} = \frac{1}{(1 + k/Pq)^P}$$

for microbial control (high loading) $\quad k_T = k_{20}\theta^{(T-20)}$

for vegetative control (low loading) $\quad k = k_j \qquad j = 1, 2, 3, \ldots 12$

The $k$-values for a group of similar wetlands will display a frequency distribution. Choice of a high or low value from such a distribution for design purposes requires assessment of the desired degree of risk and other wetland factors.

This is the TIS model, with a modified number of tanks $P \leq N$, where $N$ = the tracer-determined number of tanks. For a single chemical compound in a mixed batch system, or for

long narrow wetlands with many compartments, $P = N \rightarrow \infty$, and the exponential form may be applied.

There are quantifiable excursions around this central tendency that can be included as a stochastic addition:

$$C = C^* + \left( \frac{(C_i - C^*)}{(1 + k/Pq)^P} \right) + E$$

Variability information may also be configured as a multiplier on the deterministic portion of the prediction.

$$C = C_{trend}(1 + \Psi)$$

where $(1 + \Psi)$ is the multiplier that creates the concentration exceeded not more than a specified fraction of the time.

# 7 Suspended Solids

A major function performed by wetland ecosystems is the removal of suspended sediments from water moving through the wetland. These removals are the end result of a complicated set of internal processes, including the production of transportable solids by wetland biota.

Low water velocities, coupled with the presence of plant litter (in FWS wetlands) or sand/gravel media (in HSSF and VF wetlands), promote settling and interception of solid materials. This transfer of suspended solids from the water to the wetland sediment bed has important consequences for the quality of the water, as well as the properties and function of the wetland ecosystem. Many pollutants are associated with the incoming suspended matter, such as metals and organic chemicals, which partition strongly to suspended matter. In FWS wetlands used for municipal wastewater treatment, the accretion of solids contributes to a gradual increase in the bottom elevation of the wetland. However, wetlands used to treat urban or agricultural stormwater, or those exposed to periodic ancillary flooding, may have rapid accretions in the inlet zone.

In HSSF and VF wetlands, incoming suspended matter is removed primarily through the mechanisms of interception and settling. Although particle resuspension due to wind, wave, or animal activity can play an important role in the sediment cycle of FWS wetlands, these mechanisms are minimized in HSSF and VF wetland systems. As a result, particulate matter tends to accumulate in HSSF and VF wetland beds, with profound consequences on hydraulic conductivity and system performance.

It should be noted that the concept of using VF filter beds to remove incoming total suspended solids (TSS) as the initial stage of a treatment process dates back to the 1960s. This concept originated with Dr. Kathe Seidel, and came to be known as the Max Planck Institute Process (MPIP) or Krefeld Process (Seidel, 1966; Liénard *et al*., 1990; Brix, 1994d; Börner *et al*., 1998). The MPIP system consisted of batch-fed vertical flow wetland beds followed by HSSF wetland stages for further effluent polishing.

## 7.1 SOLIDS MEASUREMENT

TSS are measured gravimetrically after filtration and drying (Method 2540D; APHA, 1998), and reported in mg/L. The organic content is characterized as volatile suspended solids (VSS), determined from the weight loss on ignition at 550°C. The TSS method has been subjected to considerable criticism by Gray *et al*. (2000) for use on "natural" waters, and these authors recommend a suspended sediment concentration (SSC) analysis as a replacement (Method D 3977.97; ASTM, 2000). One fundamental difficulty is the representativeness of aliquots, especially if they contain sand particles. A second difficulty is the wide variability of the TSS method in low concentration ranges. Gray *et al*. (2000) quote the Standard Methods precision as a 33% coefficient of variation at 15 mg/L. TSS measurements are likely to be biased low compared to SSC measurements.

Turbidity in water is caused primarily by suspended matter, although soluble colored organic compounds can contribute. Therefore, turbidity is sometimes used as a surrogate for gravimetric measurement of suspended matter. The measurement technique involves light scattering. The instrument is the turbidimeter, consisting of a nephelometer, light source, and photodetector. The standard unit is the nephelometric turbidity unit (NTU). The correlation between TSS and NTU is often good for a specific wetland system, but care must be taken in the extrapolation from one site to another (Table 7.1). From these results, it may be concluded that the NTU–TSS relationships for FWS wetland effluents differ substantially from those for activated sludge effluents, and vary somewhat between natural systems.

### Potential for Sampling Errors

It is sometimes virtually impossible to sample interior wetland waters for TSS because of the disturbance of sediments caused by sampling. Errors of one to two orders of magnitude can easily occur. This is the case in shallow zones of vegetated FWS wetlands. If the water is deeper than about 20 cm, accurate sampling is possible but not easy. Immersion of a sampler may cause disturbance of bed sediments, or the currents caused by water rushing into a sample bottle may disturb those sediments. Ideally, the sample should flow into the sample bottle at the local velocity of the water in the wetland. This is termed *isokinetic sampling*, and is necessary to prevent extraneous resuspension. It is often not possible to achieve undisturbed sampling for TSS, and therefore difficult to obtain proper flow-weighted or volume-weighted values of TSS at interior points. For this reason, much of the available TSS data from wetland treatment systems consists of input and output measurements in pipes and at structures.

This difficulty carries over to those chemical constituents which partition strongly to the solids, or form an integral part of them. Any interior water sample will likely contain an unrepresentative proportion of the locally agitatible, or transportable, sediments and particulates. Subsequent analysis for the total amount of a partitioned or contained substance will yield an inaccurately high value.

**TABLE 7.1**
**Regressions between Total Suspended Solids and Turbidity for Wetlands, Forced through the Origin (TSS = 0, NTU = 0)**

| | NTU/TSS | $R^2$ | TSS Range (mg/L) | Turbidity Range (NTU) | Number | Reference |
|---|---|---|---|---|---|---|
| Secondary effluent | 0.37–0.50 | — | — | — | — | Crites and Tchobanoglous (1998) |
| Secondary effluent | 0.42–0.43 | — | — | — | — | Metcalf and Eddy (1991) |
| Everglades | 0.25 | 0.80 | 1–18 | 0.4–3.4 | 126 | South Florida Water Management District, unpublished data |
| River water | 0.83 | 0.77 | 0–145 | 0–125 | 64 | Des Plaines River Project, unpublished data |
| River water | 0.66 | 0.95 | 50–1,400 | 100–1,000 | 23 | Harter and Mitsch (2003) |
| Agricultural runoff | 0.75 | 0.52 | — | — | 1,013 | Everglades Nutrient Removal Project, unpublished data |
| Submerged vegetation | 0.74 | 0.93 | 0–215 | 0–150 | >100 | James *et al.* (2002) |
| Water hyacinths | 1.39 | 0.54 | 4–18 | 6–21 | 12 | Crites and Tchobanoglous (1998) |
| Oxidation pond | 0.47 | 0.06 | 1–15 | 1–27 | 96 | Gearheart *et al.* (1983) |

Similar sampling problems exist for HSSF wetlands. Most of the solids present within a HSSF wetland bed are an accumulation of microbial biofilms, intercepted particulate matter, and plant-root networks. This accumulated material, collectively called a *biomat*, occurs either as material attached to the bed media and plant roots or as colloidal material within the media pores. Because the actual flow velocity, $v$ (see Chapter 2), in an HSSF bed is very low, sampling events can induce localized flow velocities at the point of sample collection that are much higher than ambient flow velocities. This disturbs the *in situ* biomat and leads to sampling errors.

Introduction of sampling probes within the HSSF bed disturbs the bed matrix, shearing biomat off bed particles, which interferes with sample accuracy. As a result, samples taken within the HSSF bed are typically done using sample ports fabricated from perforated pipe (the same applies for VF wetlands). These sample ports are installed during construction and are a permanent feature of the HSSF wetland bed. Depending on the orientation of the perforated section of the pipe (horizontal or vertical), these sample ports will produce a sample that is width-averaged or depth-averaged over a localized portion of the HSSF wetland bed. A typical HSSF sample port assembly is shown in Figure 7.1; installation of the ports within an HSSF wetland is shown in Figure 7.2. However, the use of such pre-installed internal sampling ports does not guarantee that samples will be representative, because solids may still be selectively aspirated into the port. Difficulties in sampling lead to large variability for interior TSS samples. For instance, the coefficient of variation for TSS samples from the HSSF bed at Minoa, New York, was

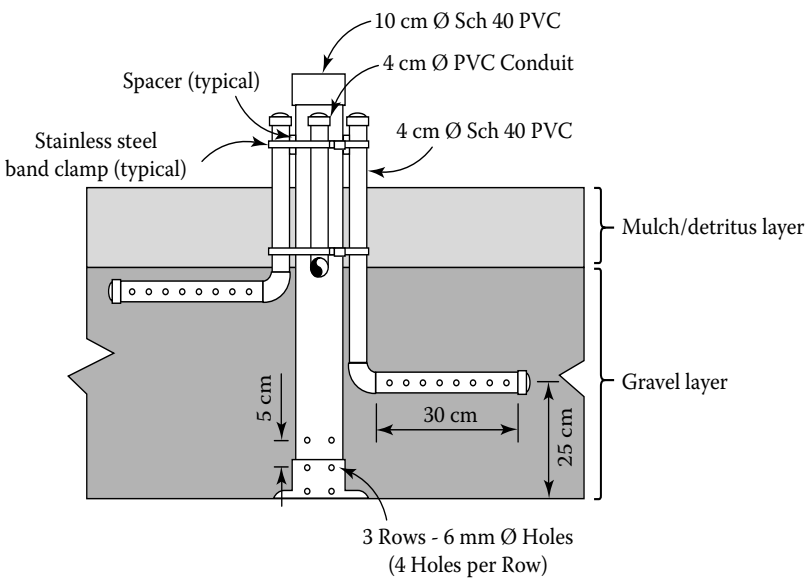



**FIGURE 7.1** Example of a HSSF wetland sampling port. This particular assembly is designed to allow sample collection at three different bed depths and installation of a thermocouple at the base of the mulch layer.



**FIGURE 7.2** Four-cell HSSF wetland at the University of Vermont. White pipes extending from the wetland beds are sampling ports.

72% (N = 534), with no apparent distance profiles. Similarly, the coefficient of variation was 145% (N = 215) in the Grand Lake, Minnesota, HSSF system.

As a consequence of these sampling difficulties, most of the samples collected in HSSF and VF wetlands consist of inlet and outlet samples, unless interior sampling ports were installed in the wetland at the time of construction. Because of the low flow velocities encountered in these systems, inlet and outlet works in contact with the water develop a biomat coating. Again, care must be taken not to disturb this biomat coating. If agitation of the water and sloughing of the biomat occurs, the sample will be contaminated and is no longer representative of the wastewater. As a result, high-energy devices such as dipping buckets and bailers should be avoided. The use of peristaltic pumps is one preferred sampling method, as the rate of sample withdrawal can be controlled, and the sampling tube can be carefully positioned to collect a representative sample. Small-diameter guide pipes are sometimes installed to facilitate placement of the sampler tubing away from side walls, tank bottoms, and other sources of sample contamination.

## Solids Characterization

The suspended solids entering a treatment wetland may display widely varying characteristics, according to the source water involved. Domestic wastewaters at all pretreatment stages contain suspended materials that are primarily organic. Runoff waters, both urban and agricultural, may contain high proportions of mineral matter. Other source waters may involve highly specific characteristics, such as the colloidal materials that discharge from milking parlors. The two principal ways of describing solids are: the soil type and the size distribution.

Soil fractions are often also applied to suspended matter, especially for situations involving mostly mineral materials. These fractions are: organic, clay, silt, and sand. The VSS fraction of the solids is usually taken to be a measure of the organic fraction (Table 7.2), and the remaining nonvolatile suspended solids (NVSS) are assumed to be the mineral fraction of the overall TSS. For incoming waters derived from runoff from mineral soils, the fraction organic may be rather low. At the Des Plaines site, river water entering averaged 11–16%

**TABLE 7.2**
**Organic Content of Various Source Waters Entering Treatment Wetlands**

| System | Influent Source | TSS Inlet (mg/L) | % NVSS |
|---|---|---|---|
| Houghton Lake, Michigan | Lagoon | 25 | 56 |
| Estevan, Saskatchewan | Lagoon | 27 | 40 |
| Des Plaines, Illinois | River | 80 | 24 |
| Tarrant, Texas | River | 276 | 10 |
| Tarrant, Texas | Sedimentation basin | 37 | 20 |
| Connell, Washington | Potato processing | 350 | 94 |

*Note:* NVSS = non-volatile suspended solids

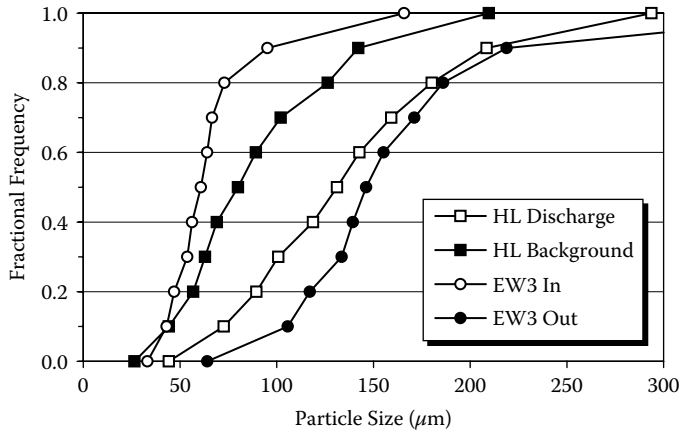



**FIGURE 7.3** Particle size distributions for two FWS wetlands. At Des Plaines (EW3), the outlet particles are larger than those entering. At Houghton Lake (HL), the discharge area particles are larger than those in wetland background areas. (From unpublished data.)

organic, whereas water leaving the treatment wetlands averaged 16–26% organic. Harter and Mitsch (2003) reported 9% organic for both entering and leaving waters from the Olentangy River wetlands. However, the Houghton Lake natural peatland showed 77% organic, and after lagoon wastewater addition showed 56% organic (unpublished data). As an extreme example, the fraction VSS in a potato wastewater treatment wetland was 94% (unpublished data). Obviously, no generalizations may be made across the spectrum of treatment wetlands and source waters, but it should be noted that organic materials may be subject to decomposition after deposition.

Mineral constituents may be defined by size ranges (Lane, 1947; Brix, 1998; Braskerud, 2003):

| | |
|---|---|
| Clay: | size < 2 μm |
| Silt: | 2 μm < size < 60 μm |
| Sand: | 60 μm < size < 2 mm |
| Gravel: | 2 mm < size < 64 mm |

These mineral particles have relatively high densities, $\rho_s \approx$ 2–2.5 g/cm$^3$, and the larger sizes settle readily. In contrast to organics, these materials accrete without decomposition.

Neither the particles entering the wetland nor those leaving are of a single size. Frequency distributions of particle sizes are always present (Figure 7.3). As a result, particle processing also becomes distributed, with large particles behaving differently from small.

## 7.2 PARTICULATE PROCESSES IN FWS WETLANDS

FWS wetlands process sediments and TSS in a number of ways (Figure 7.4). After the suspended material reaches the wetland, it joins large amounts of internally generated suspendable materials, and both are transported across the wetland. Sedimentation and trapping, and resuspension, occur en route, as does "generation" of suspended material by activities both above and below the water surface. For example, algal debris may form at one location and deposit downgradient in the wetland.

### Particulate Settling

### Single Particles

The slow-moving waters in the FWS wetland environment often permit time for physical settling of TSS. The settling velocity of the incoming particulates, combined with the depth of the wetland, gives an estimate of the time and travel distance for those solids.

Solids sink in water due to the density difference between the particle and water. For single, isolated spherical particles, the terminal velocity is reached quickly:

$$w^2 = \frac{4}{3}\frac{gd}{C_D}\left(\frac{\rho_s - \rho}{\rho}\right) \tag{7.1}$$

where

$d$ = particle diameter, m
$C_D$ = drag coefficient, dimensionless
$g$ = acceleration of gravity, m/s$^2$
$w$ = terminal velocity, m/s
$\rho$ = density of water, kg/m$^3$
$\rho_s$ = density of solids, kg/m$^3$

In turn, the drag coefficient is a function of the particle Reynolds number:

$$C_D = \left(\frac{24}{Re_p}\right)\left(1 + 0.15\,Re_p^{0.687}\right) \tag{7.2}$$

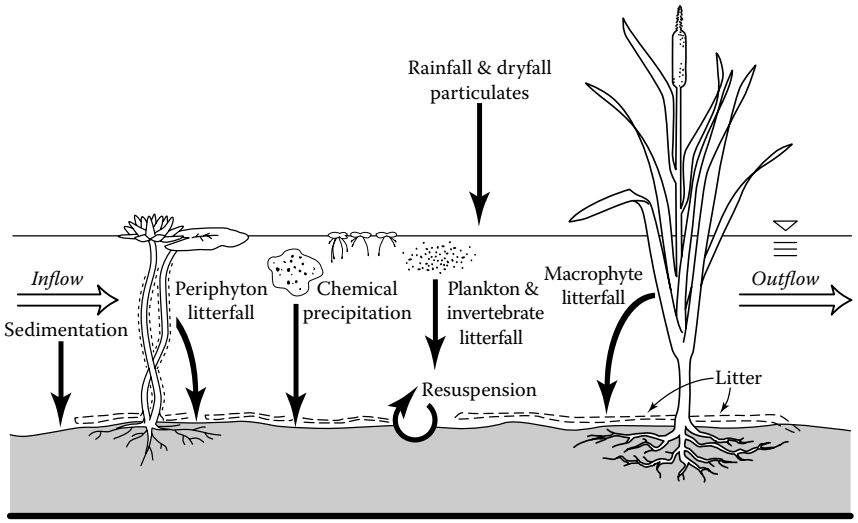



**FIGURE 7.4** Processes affecting particulate matter removal and generation in FWS wetlands. (Adapted from Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

where the particle Reynolds number is:

$$\mathrm{Re}_p = \frac{d\rho w}{\mu} \tag{7.3}$$

where

$\mathrm{Re}_p$ = particle Reynolds number, dimensionless
$d$ = particle diameter, m
$\rho$ = density of water, kg/m$^3$
$w$ = terminal velocity, m/s
$\mu$ = viscosity of water, kg/m·s (= 0.001$\mu$, in centipoise)

If all physical properties are known, Equations 7.1–7.3 combine to determine the settling velocity. This calculation is easily automated on a spreadsheet, with the results shown in Figure 7.5.

In the laminar flow region, $\mathrm{Re}_p < 1.0$, the drag coefficient is inversely proportional to the particle Reynolds number, and the settling velocity of the particle is then calculable from *Stokes law*:

$$w = \frac{gd^2}{18\mu}\left(\rho_s - \rho\right) \tag{7.4}$$

where

$d$ = particle diameter, m
$g$ = acceleration of gravity, m/s$^2$
$w$ = terminal velocity, m/s
$\rho$ = density of water, kg/m$^3$
$\rho_s$ = density of solids, kg/m$^3$
$\mu$ = viscosity of water, kg/m·s (= 0.001$\mu$, in centipoise)

In the wetland environment, neither the density nor the particle diameter is known, and the particles are not spheres or

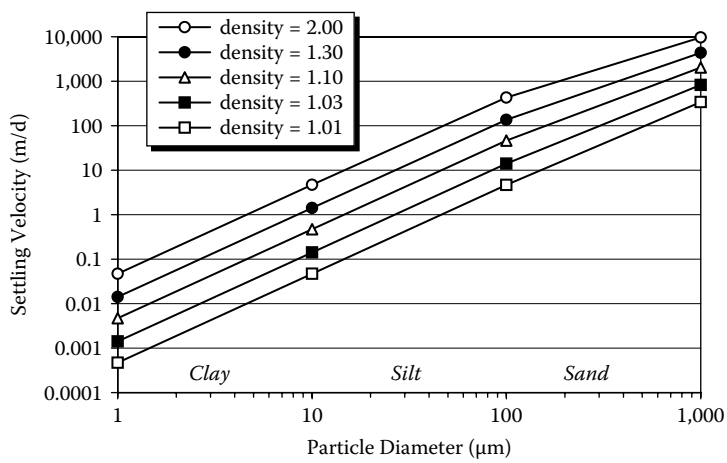



**FIGURE 7.5** Settling velocity of spherical particles in water at 20°C, for different particle densities.

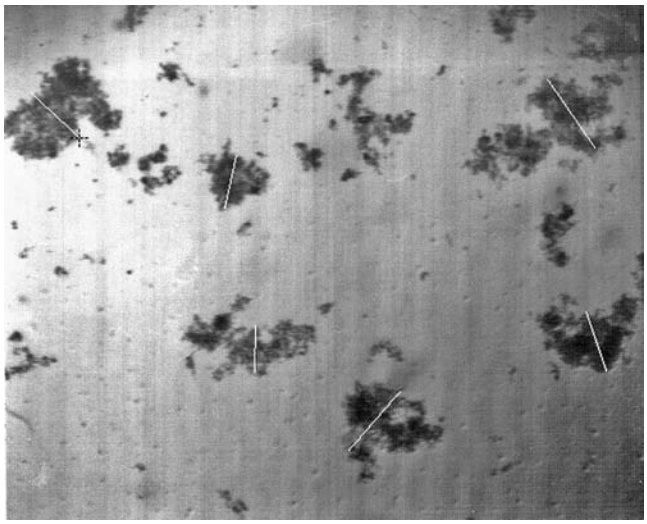


**FIGURE 7.6** Photomicrograph of suspended particulate matter in the effluent from Des Plaines wetland EW3. (From Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

discs (Figure 7.6). Although it is possible to correct for non-spherical shapes (Dietrich, 1982), there is not a convenient method for determination of the particle density. Further, particles may agglomerate to larger size, or be subject to interference from neighboring particles.

### Settling of Mixtures

Settling of particulate matter may be described by a first-order model (Equation 7.4) for each size fraction. In general, settling velocities are proportional to the square of particle size, with variation including shape factors and particle density. Particle mass may be estimated to be roughly proportional to the cube of size. The time of fall of a particle through a vertical distance ($h$) is determined from its velocity:

$$t_{fall} = \frac{h}{w} \tag{7.5}$$

where

$h$ = water depth, m
$t_{fall}$ = time to fall, s
$w$ = terminal velocity, m/s

If the water is moving through the wetland length ($L$) at velocity ($u$), the time of travel is:

$$t_{travel} = \frac{L}{u} \tag{7.6}$$

where

$L$ = wetland length, m
$t_{travel}$ = time to traverse wetland, s
$u$ = superficial water (flow) velocity, m/s

Theoretically, all particles of a size corresponding to a given fall velocity will be removed by settling if the travel time exceeds the settling time from the top of the water:

when
$$\frac{L}{u} > \frac{h}{w}$$
$$N_{fall} = \frac{Lw}{uh} > 1 \tag{7.7}$$

where

$N_{fall}$ = particle falling number, dimensionless

These concepts have been applied to mixtures in shallow overland flow in grass (Deletic, 1999), and in wetlands (Li *et al*., 2007), with mean particle diameter used to determine the settling velocity ($w$). Values of $N_{fall}$ were found to be above 10 for complete removal, reflecting the difficulty of settling of the small end of the particle size distribution (Figure 7.7).

These relations also allow the conversion of a size distribution to a settling velocity distribution, and ultimately to the size distribution remaining after some fixed settling time. Procedures for such calculations may be found in Crites and Tchobanoglous (1998); however, there is rarely sufficient information on particle properties available. Braskerud (2003) found considerable discrepancies when applying these procedures to mineral particles trapped in wetlands.

### Column Studies

Settling rates may also be determined experimentally. Typically, a large diameter column of water is charged with a well-stirred suspension of particles, and the concentration measured at a sequence of times at a series of depths below the water surface. Vertical profiles of TSS exist in differing shapes, depending on flocculation and particle–particle interference. A number of analytical techniques may be applied to such data (Font, 1991). Only the mean water column concentration of

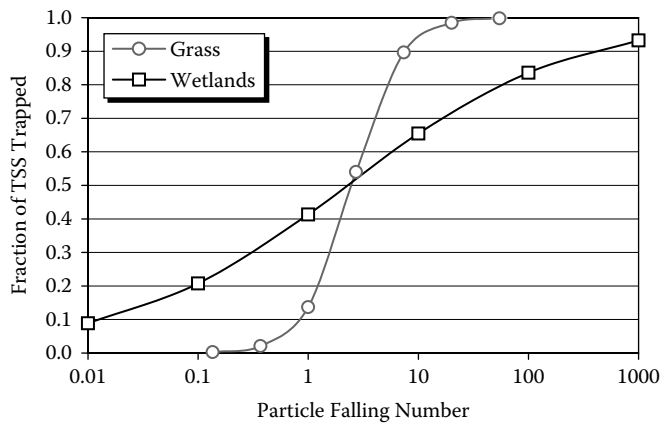



**FIGURE 7.7** Removal of TSS in shallow overland flow in grass. The particle falling number is ($Lw/uh$), in which $w$ is the terminal velocity of the mean particle diameter. Original data centered on a mean diameter of about 50 μm. (Data from Deletic (1999) *Water Science and Technology* 39(9): 129–136; and Li *et al.* (2007) *Journal of Hydrology* 338: 285–296.)

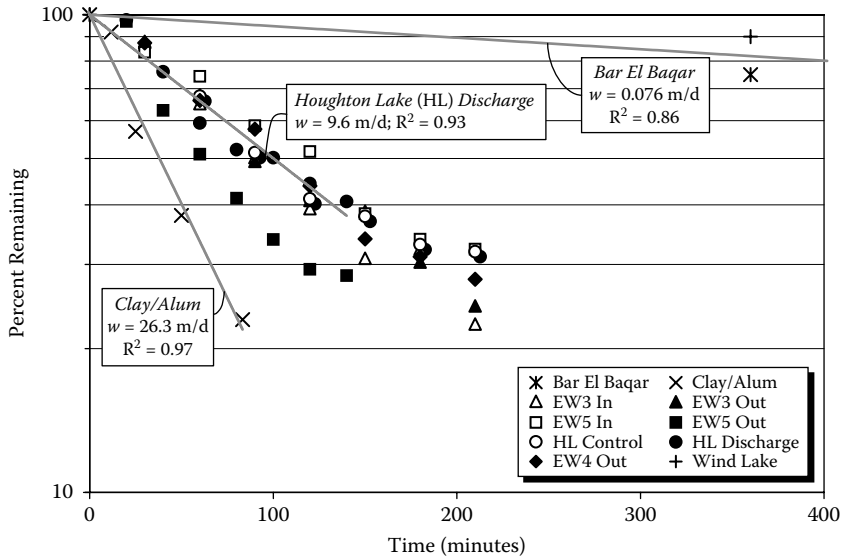



**FIGURE 7.8** Examples of settling characteristics of TSS derived from wetlands and other natural contributing sources. The mean settling velocities range from 0.076 m/d for the Wind Lake wetland TSS, to 26.3 m/d for the clay alum mix. (Data for HL Control, HL Discharge, EW3 In, EW3 Out, EW4 Out, EW5 In, EW5 Out, and Wind Lake: authors' unpublished data; data for Clay/Alum: ASCE (1975) *Sedimentation Engineering.* Vanoni (Ed.), American Society of Civil Engineers (ASCE): New York; data for Bar El Baqar: PLA (1993) *1993 Field Program for the Egyptian Engineered Wetland.* Report prepared for the United Nations Development Programme, New York, P. Lane and Associates, Ltd. (PLA).) (Graph from Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

TSS will be considered here. That concentration decreases as time progresses. Settling column data, for example, wetland waters and other sources, indicate an exponential decrease in concentration with time, and a time scale of a few hours for the majority of settling to occur (Figure 7.8). The settling velocities shown in Figure 7.8 range from $w = 0.076$ to 26.3 m/d. Interestingly, exponential decreases are found for the several sediments in Figure 7.8.

Caution must be used in those applications where colloidal materials may be present in the inflow, because these materials are stable or very slow to settle. Very fine clay suspensions and some milk processing wastewaters fall into this category. The settling velocity for planktonic solids was found to be on the order of $w = 0.076$ m/d for the Wind Lake, Wisconsin, wetland, which was dominated by algae.

Column settling data provide estimates of the removal time for TSS in the absence of dense vegetation. Confirmation of field applicability was found for wetland EW3 at Des Plaines in 1991. The inlet zone was essentially unvegetated, and the water velocity was on the order of 30 m/d. Settling column data (Figure 7.8) suggested that solids should essentially be gone in eight hours, or after a travel distance of about ten meters. Transect information confirmed this estimate.

## "Filtration" versus Interception

Conventional wisdom has it that the presence of dense wetland vegetation causes settling to be augmented by filtration. This is often not true in the usual sense of the term filtration. It is trapping of sediments in the litter layer that prevents resuspension, and thus enhances the net apparent suspended sediment removal. Macrophytes and their litter form a non-homogeneous "fiber bed" in the wetland context. The void fraction in the stems and litter is quite high; straining and sieving are thus not typically the dominant mechanisms. Submerged biomass additionally traps sediment in sheltered microzones, thereby lessening the potential for resuspension. Confirmation of sedimentation as the principal mechanism was provided in the laboratory studies of Schmid *et al.* (2005).

However, there are wetland circumstances in which the dominant mechanism is particles striking immersed objects and sticking. The three principal mechanisms of fiber-bed filtration are well known and documented in handbooks (see, e.g., Perry *et al.*, 1982; Metcalf and Eddy, 1991):

1. Inertial deposition or impaction—particles moving fast enough that they crash head-on into plant stems rather than being swept around by the water currents.
2. Diffusional deposition—random processes at either microscale (Brownian motion) or macroscale (bioturbation) which move a particle to an immersed surface.
3. Flow-line interception—particles moving with the water and avoiding head-on collisions, but passing close enough to graze the stem and its biofilm, and sticking.

The efficiencies of collection for these mechanisms depend on the water velocity, particle properties, and water properties, as well as the character of submerged surfaces. A typical wetland "fiber" is a bulrush stem of about 1 cm diameter.

A typical particle might be on the order of 1–100 μm. A typical water velocity is on the order of 10–100 m/d. Under these conditions, the collection efficiencies of Mechanisms 1 and 2 are predicted to be vanishingly small. There is evidence that Mechanism 3 is operative and significant. Lloyd (1997) examined the submerged surfaces of bulrushes (*Schoenoplectus* (*Scirpus*) *validus*) and found particles as small as 0.5–2.5 μm sticking to biofilms (Breen and Lawrence, 1998). Saiers *et al.* (2003) studied the movement of very small (0.3 μm), unsettleable particles of $TiO_2$ in the Florida Everglades. They concluded that 29% of the particle impacts on periphyton-coated stems resulted in sticking in a plant (*Eleocharis* spp.) density of 1,150 per $m^2$. These stems were only 0.2 cm in diameter, resulting in 99% porosity. Saiers *et al.* (2003) defined a first-order rate constant for removal by sticking, which on an areal basis is:

$$k = \frac{\eta u h^2}{\left[\frac{1}{n} - \frac{\pi d^2}{4}\right]} \tag{7.8}$$

where

$d$ = stem diameter, m
$h$ = water depth, m
$k$ = areal removal rate constant, m/hr
$n$ = stem density, #/$m^2$
$u$ = water velocity, m/hr
$\eta$ = sticking efficiency, dimensionless

## Resuspension

Settled particles may not "stay put" for a number of reasons. Hydrodynamic shear forces may tear particles loose from the sediment bed, which is a dominant mechanism in streams and rivers. However, wetlands provide an environment in which other processes may occur as well. Wind and wave action are major drivers of resuspension in lakes, and may also be operative in open water areas of FWS wetlands. Additionally, biological activity may result in the movement of particles from the sediments to overlying water.

### Unvegetated Surfaces

Much is known about the resuspension of particulates from flat surfaces (ASCE, 1975). Most interpretations are made in terms of the force per unit area (shear stress) required to tear a particle loose from the sediment surface. The concepts involve purely physical forces and apply most readily to mineral substrates and river systems. Most theoretical results are for planar sediment bed bottoms with no extraneous objects. Vegetated wetland bottoms do not fit these conditions.

In the treatment wetland environment, physical resuspension (due to high flow velocities) is not a dominant process. Water velocities are usually too low to dislodge a settled particle from either the bottom or a position on submerged vegetation. However, in design, it may be necessary to avoid wetland aspect ratios that produce excessively high linear velocities. The potential for erosive velocities exists for highly loaded wetlands with high length-to-width ratios. Estimation of the velocity required to foster resuspension may be based on the settling characteristics of the solids and the frictional characteristics of the wetland, combined with known correlations of the critical shear stress for particle dislodgment (ASCE, 1975). Modifications are needed for the case of laminar flow, which is the general case for wetlands (Mantz, 1977; Yalin and Karahan, 1979).

Velocities that cause erosion in open channels are high compared to wetlands. For instance, French (1985) lists recommended maximum (nonscouring) velocities for 14 canal materials in the range $0.46 < u < 1.83$ m/s. Such considerations resulted in a maximum canal velocity design constraint of 0.76 m/s for Everglades protection wetlands conveyance canals (Burns and McDonnell, 1996). In anticipation of more erodable particulates inside the wetlands, wetland velocities were limited to no more than 0.03 m/s (2,600 m/d). These large wetlands had lengths up to 2,500 m, which therefore created a design detention minimum of one day. The annual average design detention time was 30 days. No erosion has been noted in this project or its companions of comparable size and detention.

### Effects of Vegetation

It is known that vegetation increases the retention of particulates in both lake and stream environments. For instance, Horppila and Nurminen (2003) found that beds of submerged plant species—butter cup: *Ranunculus circinatus*; coontail: *Ceratophyllum demersum*; and pond weed: *Potamogeton obtusifolius*—in a lake environment effectively prevented resuspension, which they attributed to a reduction in wind and wave action. Horvath (2004) studied the effect of macrophytes—rushes: *Juncus* spp.; bur-reed: *Sparganium* spp.; forget-me-not: *Myosotis* spp.—on retention of particulate matter in a small stream, and found enhanced trapping in proportion to biomass.

It is logical that these same effects are prevalent in treatment wetlands. Dieter (1990) found about a threefold reduction in resuspension from open water to vegetated areas in a prairie pothole wetland. Hosokawa and Horie (1992) demonstrated enhanced removal in both laboratory channels with dowels and in field flumes in a reed bed (*Phragmites australis*). In fully vegetated wetlands, the litter and root mats provide excellent stabilization of the wetland soils and sediments. This limits, but does not eliminate, resuspension.

### The Floc Layer

Some treatment wetlands, such as those used for low-level nutrient removal, develop very flocculent sediment beds. These sediments are positioned on top of the consolidated soils, and may be interwoven with plant detritus. Bulk densities of such floc layers may range downward to 0.03–0.05 g/$cm^3$ of dry matter (James *et al.*, 2001; Coveney *et al.*, 2002). Depths of these loose and unconsolidated materials have been found to exceed 30 cm in some situations (Table 7.3).

**TABLE 7.3**
**Floc Thicknesses and Bulk Densities for the Everglades Nutrient Removal Project (ENRP), Lake Apopka, Florida Project, and the Sacramento California Demonstration Wetlands Project**

| | | Thickness (cm) | | | Bulk Density (g/mL) | | |
|---|---|---|---|---|---|---|---|
| Site | Years | Mean | SE | N | Mean | SE | N |
| Sacramento 4 | 2.6 | 17.2 | 1.4 | 8 | 0.068 | 0.015 | 12 |
| Sacramento 4 | 2.6 | 11.3 | 1.0 | 8 | 0.069 | 0.017 | 16 |
| ENRP 1 | 9.0 | 19.7 | 1.4 | 30 | 0.076 | 0.006 | 30 |
| ENRP 2 | 9.0 | 18.2 | 1.4 | 26 | 0.099 | 0.007 | 26 |
| ENRP 3 | 9.0 | 18.9 | 1.8 | 22 | 0.072 | 0.008 | 22 |
| ENRP 4 | 9.0 | 16.7 | 1.4 | 10 | 0.092 | 0.012 | 10 |
| Apopka | 2.4 | 33 | — | 48 | 0.051 | — | 48 |

*Source:* Data from Nolte and Associates (1997) *Sacramento Regional Wastewater Treatment Plant Demonstration Wetlands Project*. 1996 Annual Report to Sacramento Regional County Sanitation District, Nolte and Associates: Sacramento, California; Coveney *et al.* (2002) *Ecological Engineering* 19(2): 141–159; and South Florida Water Management District, unpublished data.

Despite low bulk density, the amount of floc dry matter is substantial. For instance, the Sacramento data in Table 7.3 convert to about 9,700 g/m$^2$ of dry matter present as the floc.

The origins of floc are not well understood, but it has been found to occur in both macrophyte-dominated (Sacramento) and SAV-dominated (ENRP Cell 4) wetlands. It likely contains a significant microbial detrital component, as well as algal and macrophyte detritus. Floc also occurs in the ultra-low nutrient, unimpacted Everglades (Gaiser *et al*., 2005), where it is presumably the result of an active periphyton biological cycle.

There is not an accepted common terminology for the floc. Nolte (1997) called it the "*A* layer," and described it as follows:

> The *A* layer consists of a slurry of dark, decomposing, loosely structured detrital material that pours out when the sampler is tipped. The material in the *A* layer has settled to the bottom, but has not been integrated into the matrix of the basin floor.

This material is not subject to transport under most ambient conditions, but is very mobile if disturbed. For example, disturbance resuspension tests were conducted at the Houghton Lake treatment wetland. A bottomless sharp-edged cylinder was twisted down into the soil, and the interior biomass (live, dead, litter) was removed. The remaining, isolated water was gently agitated, and then sampled for solids content. The mobile material averaged 880 ± 100 g/m$^2$ (mean ± SE).

### Other Resuspension Mechanisms

The wetland environment provides an opportunity for three other mechanisms of resuspension: wind-driven turbulence, bioturbation, and gas lift. In open water areas, wind-driven currents cause surface flow in the wind direction and return flows along the bottom in the opposite direction. These recirculation velocities can far exceed the net velocity from inlet to outlet. For wetlands with large open water zones, waves add to the overall process of resuspension. Lake studies suggest both processes are wind-dependent. For instance, Malmaeus and Hakanson (2003) suggest resuspension is proportional to the square of the wind speed. Additionally, fetch and water depth are controlling factors.

Animals of all types and sizes can cause resuspension to occur. Feeding carp (Kadlec and Hey, 1994) and nesting shad (APAI, 1995) have been observed to cause problems. The carp rooted in the sediments for food, and thus resuspended large amounts of sediments. Control was by drawdown and freezing. The shad fanned nests on the wetland bottom, and resuspended sediments. Control was by drawdown and avian predation. Beaver activity can cause stirring, often at the outlet of the wetland, in conjunction with attempts to dam the outlet. Human sampling activities in the interior of treatment wetlands may also result in locally-elevated concentrations of suspended solids. For instance, the passage of a drifting boat can cause extreme resuspension (Figure 7.9).

Gas lift occurs when bubbles of gas become trapped in or attached to particulate matter. Wetland sediments are often of near neutral buoyancy; so a small amount of trapped gas can cause "sinkers" to become "floaters." There are several gas-generating reactions in a wetland environment. Most important are photosynthetic production of oxygen by algae and production of methane in anaerobic zones.

### Chemical Precipitates

Several chemical reactions can produce particulate matter within wetlands under the proper circumstances. Some of the more important are the oxyhydroxides of iron, calcium carbonate under aerobic conditions, and divalent metal sulfides under anaerobic conditions. As conditions of chemical composition, pH, and redox change in the wetland, these and other compounds may undergo dissolution and be removed from the sediment bed.



**FIGURE 7.9** Passage of a drifting boat can stir up a cloud of floc. This site is in the interior of the A.R. Marshall Loxahatchee National Wildlife Refuge. The water was about 45 cm deep, and the vegetation was sparse.

**Iron Flocs.** The iron oxyhydroxides are typically flocs, with the possibility of coprecipitates. They may form under conditions of elevated dissolved ferric iron and oxygen-rich water. The processes may be represented as (Younger *et al*., 2002)

$$Fe^{2+} + \frac{1}{4}O_2 + H^+ \rightarrow Fe^{3+} + \frac{1}{2}H_2O \qquad (7.9)$$

$$Fe^{3+} + 2H_2O \rightarrow FeOOH_{(sus)} + 3H^+ \qquad (7.10)$$

$$FeOOH_{(sus)} \rightarrow FeOOH_{(sed)} \qquad (7.11)$$

These precipitates are characterized by an unmistakable blood-red color (Figure 7.10). As indicated by the chemistry, formation is inhibited by low pH and by low dissolved oxygen. Formation may be abiotic, or mediated by microorganisms such as *Thiobacillus ferrooxidans*. However, at pH > 9, the rate of the abiotic reaction is so fast that formation is controlled by the rate of oxygen supply (Younger *et al*., 2002). In the pH range 6 < pH < 8 that generally typifies treatment wetlands, rates are slow enough to be a design consideration. This set of reactions forms the basis for phosphorus removal by addition of ferric chloride to wastewaters, and the accompanying co-precipitation of the phosphorus. Consequently, the subsequent fate of these solids in polishing treatment wetlands is of considerable interest.

**Aluminum Flocs.** The aluminum oxyhydroxides are also typically flocs, with the possibility of co-precipitates. They may form under circumneutral pH conditions, and do not require oxygen. The processes may be represented as (Sobolewski, 1999):

$$Al^{3+} + H_2O \rightarrow Al(OH)_3 \downarrow + 3H^+ \qquad (7.12)$$



**FIGURE 7.10 (A color version of this figure follows page 550)** Venting groundwater at this Wellsville, New York, site contains iron, which oxidizes upon contact with air.

These precipitates are characterized by their formation of a "pin floc" material that does not readily settle in FWS wetlands (Bachand *et al*., 1999). This set of reactions also forms the basis for phosphorus removal by addition of alum to wastewaters, and the accompanying co-precipitation of the phosphorus. Consequently, the subsequent fate of these solids in polishing treatment wetlands is of considerable interest.

**Calcium Carbonate.** Calcium carbonates may be formed in wetlands, under conditions of elevated pH and dissolved calcium. The operative chemistry may be summarized as

$$Ca^{2+} + HCO_3^- + H_2O \rightarrow CaCO_3 \downarrow + H^+ \quad (7.13)$$

This reaction may occur abiotically, but perhaps more importantly it may be mediated by algae. Algal activity can drive up pH, and create conditions that foster creation of calcium-rich solids (Vymazal, 1995). Indeed, this process has contributed to the formation of marl prairies as a form of natural wetlands. New sediments in Everglades protection treatment wetlands contain a significant fraction of calcium compounds (Dierberg *et al*., 2002).

**Metal Sulfides.** Many metals form very insoluble sulfides, including mercury, lead, cadmium, and zinc, as further discussed in Chapter 11. These precipitates are important in the processes of metal removal in wetlands, and follow the general chemistry (Sobolewski, 1999):

$$SO_4^{2-} + 2CH_2O \rightarrow HS^- + H^+ + 2HCO_3^- \quad (7.14)$$

$$M^{2+} + HS^- \rightarrow MS + H^+ \quad (7.15)$$

However, for many treatment wetland applications, metals are present at only very low concentrations. Consequently, the formation of insoluble sulfides does not usually create measurable additions to the sediments of the wetlands.

### Biological Sediment Generation

Wetlands produce sediments via processes of death, litter fall, and litter attrition. This occurs for biota at a number of different size scales, ranging from macrophytes on down to bacteria. Algal productivity can be a major generator of suspended solids. A second set of processes adds pollen and seeds to the water. The TSS produced is organic in character, resulting in a high carbon content and a high proportion of VSS. The chlorophyll and pheophytin (dead chlorophyll) content is high if the algal pathway is dominant.

Some TSS originates from leaf and stem litter. For instance, annual leaf litterfall in a natural sedge-shrub peatland was found to be 60–70 g/m$^2$ (Chamie, 1976). Some part of this material contributes to TSS, either via direct attrition, or via microbial decomposition.

The generation of sedimentary material is a very important internal process in nutrient-rich treatment wetlands. The generous supply of nutrients assures a large production of a wide variety of transportable organisms and associated dead organic material. Such wetlands are characterized by high water chlorophyll content and high sediment accumulation. Bacterial and algal growth is promoted, and decomposition products form a new pool of suspendable material. A host of wetland invertebrates, such as *Daphnia* and waterboatman (*Corixidae*), also die and contribute to the sediments, and they may be present in pumped lagoon water.

These processes are virtually impossible to predict and quantify. But it is important to recognize that they exist, because they contribute to a background level of TSS in a wetland.

### Accretion

Trapped TSS, plus material generated within the wetland, will accrete as either movable sediment or the consolidated immovable new soil produced from the sediments. Not all of the dead plant material undergoes decomposition. Some small portions of both aboveground and belowground necromass resist decay, although these are typically shredded by microbial and other invertebrate processes. Underground processes form nonsuspendable accretions, some part of which is stable and does not fully decompose. The origins of new sediments may be from remnant macrophyte stem and leaf debris, remnants of dead roots and rhizomes, and from indecomposable fractions of dead microflora and microfauna (algae, fungi, invertebrates, bacteria).

### Measurement of Accretion

The processes above combine to determine the amount of sediment at various locations within the wetland as a function of time and the TSS concentration in the wetland effluent. Cup collectors may be placed on the wetland bottom (Jordan and Valiela, 1983; Fennessy *et al*., 1992; Braskerud, 2001a); these typically intercept the downward vertical flux of sediment but prevent shear-induced resuspension. Plate collectors may be placed on the wetland bottom, followed by sediment harvest above that horizon at a later time (Kozerski and Leuschner, 1999; Braskerud, 2001a). Alternatively, neutral density particulate material may be laid down in a layer, and retrieved by coring and sectioning (Harter and Mitsch, 2003). Another technique involves the elevation of a blunt-footed rod, which is lowered to the sediment surface. A reference rod, driven deep into stable soils, provides the local datum (Reeder, 1990). Other quantitative studies have relied upon atmospheric deposition markers such as radioactive cesium ($^{137}$Cs) or radioactive lead ($^{210}$Pb) (Kadlec and Robbins, 1984; Craft and Richardson, 1993; Robbins *et al*., 2004). These techniques require several years of continued deposition for maximum accuracy.

Cup collectors typically yield much more sediment than plate collectors. For instance, Schulz *et al*. (2003b) found $30 \pm 3$ g/m$^2$·d collected in cups in a riverine bed of *Sagittaria sagittifolia*, compared to $8 \pm 2$ g/m$^2$·d collected on plates. This is presumably due to the prevention of resuspension in cups, whether it be due to fluid shear or to bioturbation. For mineral sediments, the difference between cups and plates is less, probably because of the lesser importance of resuspension of heavier particles (Braskerud, 2001a).

**TABLE 7.4**
**Accretion Rates in FWS Wetlands**

| Location | Wetland | Reference | Method | Water $NH_3$-N (typical) (mg/L) | Accretion (cm/yr) |
|---|---|---|---|---|---|
| Louisiana | Salt marsh | DeLaune *et al.* (1978) | $^{137}Cs$ | Low | 1.1–1.35 |
| Louisiana | Forested | Conner and Day (1991) | Feldspar | Low | 0.84 |
| Louisiana | Forested | Rybczyk *et al.* (2002) | Feldspar | 0.05 | 0.14 |
| Xianghai, China | Open marsh | Wang *et al.* (2004) | $^{137}Cs$ + $^{210}Pb$ | Low | 0.35 |
| Xianghai, China | Isolated marsh | Wang *et al.* (2004) | $^{137}Cs$ + $^{210}Pb$ | Low | 0.65 |
| Michigan | Marsh | Kadlec and Robbins (1984) | $^{210}Pb$ | 0.1 | 0.2 |
| Norway Farm Runoff | Marsh CW | Braskerud (2001b) | Plate | 0.16 | 2 |
| Norway Farm Runoff | Marsh CW | Braskerud (2001b) | Plate | 0.37 | 4 |
| Everglades WCA2A | Marsh | Reddy *et al.* (1993) | $^{137}Cs$ | 0.3 | 0.5 |
| Everglades WCA2A | Marsh | Craft and Richardson (1993b) | $^{137}Cs$ | 0.3 | 0.4 |
| Everglades WCA3 | Marsh | Craft and Richardson (1993b) | $^{137}Cs$ | 0.1 | 0.3 |
| Everglades | Marsh | Robbins *et al.* (1999) | $^{210}Pb$ | 0.3 | 0.5 |
| Everglades | Marsh | Chimney (unpublished data) | Feldspar | 0.1 | 0.85 |
| Sacramento, California | Marsh CW | Nolte and Associates (1998b) | Visual | 16 | 1.5 |
| Houghton Lake, Michigan | Marsh NTW | Kadlec (unpublished data) | Resurvey | 10 | 1.0 |
| Chiricahueto Runoff, Mexico | Marsh | Soto-Jimenez *et al.* (2003) | $^{210}Pb$ | 14 | 1.0 |
| Louisiana | Forested NTW | Rybczyk *et al.* (2002) | Feldspar | 15 | 1.14 |

*Note:* CW = constructed wetland; NTW = natural treatment wetland.

## Amount and Distribution of Accretion

Accretions measured in various wetlands vary from a few millimeters per year to over a centimeter per year (Table 7.4). These accumulated solids represent the potential for filling of a constructed wetland. It is an easy calculation to allocate the removed TSS to the buildup of new solids in the FWS wetland. For municipal wastewater polishing, typical operations lead to an accumulation of 1–2 mm/yr of new solids (50 mg/L removed at $q$ = 5.5 cm/d at a bulk density of 0.5 g/cm$^3$ yields 2.0 mm/yr). But that material is augmented by internally generated solids and decreased by decomposition of the organic portion of sediments and soils. The net increase may total up to 10 mm/yr in a highly eutrophic marsh (Table 7.4). Even more accumulation can result from the trapping of mineral solids from urban or agricultural runoff.

For high amounts of sediment trapping compared to generation and resuspension, buildup typically occurs preferentially in the inlet section of the wetland. Therefore, a "delta" of accreted sediments builds in the inlet region of the wetland. For example, food processing wastewaters can contain very high TSS concentrations, which in turn can fill a treatment wetland with solids. Van Oostrom (1995) reported that one third of the volume of a floating *Glyceria* mat wetland was filled after 20 months of operation (Figure 7.11). The wastewater was

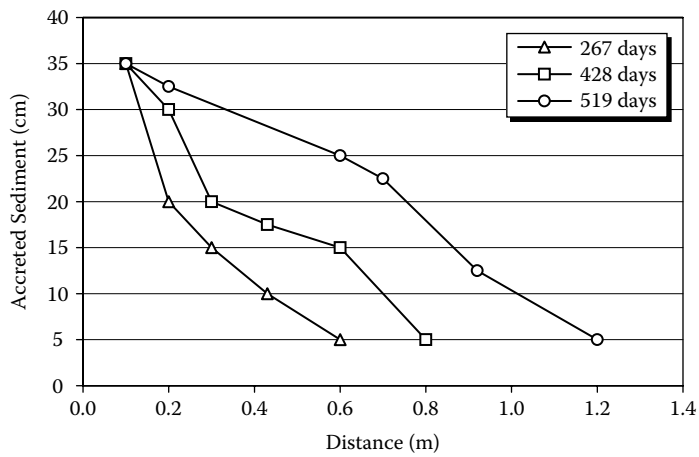



**FIGURE 7.11** The sediment "delta" developed in a small treatment wetland mesocosm. (Data from van Oostrom (1995) *Water Science and Technology* 32(3): 137–148.) (Graph from Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

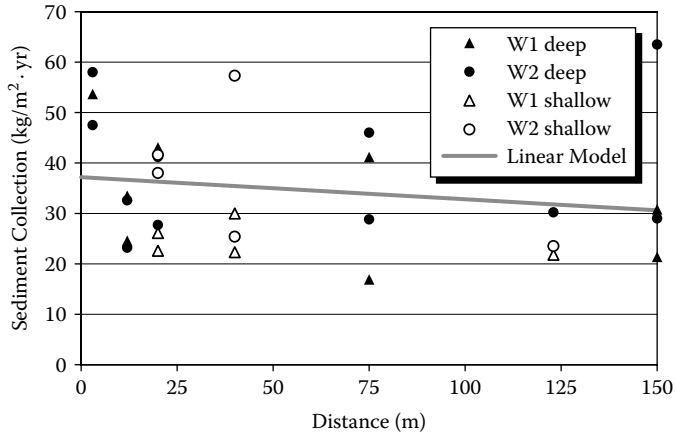



**FIGURE 7.12** Spatial distribution of plate sediment collection rates along the flow direction of a constructed marsh treating river water. (Data from Harter and Mitsch (2003) *Journal of Environmental Quality* 32(4): 325–334.)

a nitrified meat processing effluent, with incoming TSS of 269 mg/L, and the removal rate was 5,300 g/m$^2$·yr. Accreted sediments totaled 40% of the removed solids, 2,100 g/m$^2$·yr, and these were concentrated near the inlet end of the wetland. The density of the solids was very low, around 0.03 g/cm$^3$.

In contrast, lighter loadings and open water areas may foster the redistribution of suspendable material. For instance, Brueske and Barrett (1994) found a "delta" in a highly loaded wetland (around 3.6 g/m$^2$·d TSS), but little or no "delta" for a lower loading (around 0.8 g/m$^2$·d TSS). Both Harter and Mitsch (2003) and Brueske and Barrett (1994) found greater sediment accretion in open water areas, which may have been attributable to most of the flow traveling through such areas, or to bioturbation (Figure 7.12). In contrast, Benoy and Kalff (1999) found a linear relation between sediment accumulation and biomass for submerged species *Myriophyllum spicatum*, *Potamogeton* spp., *Ceratophyllum demersum*, and *Elodea canadensis* beds in Lake Memphremagog between Québec and Vermont. It is apparent that the processes involved in sediment accumulation in wetlands are too complicated to permit generalities.

In the long run, solids accretion may raise the elevation of the wetland bottom, and thus impact system hydraulics and treatment. U.S. EPA (2000a) suggests that accretion in municipal wastewater treatment wetlands results from both external and internal sources, which is conceptually correct. However, the U.S. EPA (2000a) estimate of accretion from external solids, 2–4 cm/yr, is based upon lagoon accumulation rates, and is excessively high. For example, the removal of 30 mg/L of TSS at a hydraulic loading rate of 10 cm/d results in solids storage of 1,095 g/m$^2$·yr. At a density of 0.2 g/cm$^3$, this gives 0.55 cm/yr if there is no decomposition. However, municipal TSS is about half mineral, and half-decomposable solids (VSS, see Table 7.2), and hence long-term external accretion would be about 0.27 cm/yr. U.S. EPA (2000a) estimates internal accretion as the annual deposition of macrophyte detritus to be 2.4 cm/yr. However, that material too is subject to decomposition, leaving an estimated residual long-term buildup of 20% of the input, or 0.48 cm/yr. In sum, the accretion in this example would be 0.75 cm/yr. This is consistent with the measured accretions in Table 7.4, for municipal systems. However, as the mineral content and loadings of TSS increase, so do accretions. Highly loaded wetlands treating mineral solids have been observed to accrete 2–8 cm/yr (Braskerud, 2001a).

Accretion is typically spatially nonuniform, due to gradients in deposition and productivity. This has been found to be true even in wetlands of very low nutrient status (Reddy *et al*., 1993). Inlet zones may therefore be subject to solids accumulations that are double the wetland average. However, some wetlands appear to redistribute solids fairly evenly from inlet to outlet.

To the authors' knowledge, only one municipal wastewater polishing FWS wetland has been serviced for solids removal, the Orlando, Florida Easterly Wetland inlet cells (White *et al*., 2004). The one removal of accumulations restored good hydraulic patterns, and restored original water quality performance.

It was suspected that uneven accumulations of new sediments were affecting flow patterns, and reducing efficiency (Sees, 2005). The inlet 9% of the wetland was excavated 45 cm, after 15 years of operation. This overexcavation restored more than the original freeboard, and resulted in a great improvement in hydraulic efficiency, from 34% to 74% (see Chapter 2). Two of the oldest facilities, Vermontville, Michigan (32 years, constructed), and Houghton Lake, Michigan (30 years, natural), have experienced accretions in the range of Table 7.4, but this has not jeopardized containment or operability. However, the Tucson, Arizona, Sweetwater wetland inlet cells have required solids removal after just a few years, because of the high suspended solids inlet water (see Figure 7.13).



**FIGURE 7.13** Excessive TSS can fill the inlet deep zone to a treatment wetland, as happened at the Tucson, Arizona, sweetwater wetland. Note the bird tracks that highlight the complete filling of the deep zone with relatively high density solids. Incoming waters had high TSS from filter backwashes at the secondary treatment plant that provided the source water.

## 7.3 TSS REMOVAL IN FWS WETLANDS

As for most treatment wetland water quality parameters, the utilization of input and output data to compute percent removals is an inadequate representation of the processes which lead to those removals. This is particularly true for the removal of TSS.

### Internal Cycling: Mass Balances

Models of sediment transport have been developed and verified for estuaries (Hayter and Mehta, 1986; Nakata, 1989, for example). These are 2- and 3-D models that allow for dispersion, settling, and resuspension; and generation is not usually an important term. These models may be adapted to the wetland situation. In the short term, there are significant fluctuations in TSS storage within the water column in response to the variations in settling, resuspension, and generation. Childers and Day (1990) state: "Our results affirm the variability of short-term sediment transport and depositional processes...." Over a long period, however, changes in water column storage are negligible compared to other inputs and outputs. The water column TSS mass balance then assumes the character of a steady state model. There is an accompanying sediment bed balance, in which the change in storage is the dominant feature. The long-term, time-average profiles calculated from the vertically averaged mass balances for TSS in a linear flow wetland are (see Figure 7.14):

$$uh\frac{\partial C}{\partial x} = G + R - S \tag{7.16}$$

$$\frac{\partial (B+P)}{\partial t} = S - R - D \tag{7.17}$$

where

$B$ = transportable solids bed, g/m$^2$
$C$ = concentration, g/m$^3$ = mg/L
$D$ = decomposition rate of transportable solids, g/m$^2$·d
$G$ = generation rate, g/m$^2$·d
$h$ = water depth, m
$P$ = permanent soils and sediments, g/m$^2$
$R$ = resuspension rate, g/m$^2$·d
$S$ = settling rate, g/m$^2$·d
$t$ = time, d
$u$ = superficial water velocity, m/d
$x$ = distance, m

In general, the settling rate may be written as:

$$S = w\,C \tag{7.18}$$

where

$w$ = solids settling velocity, m/d

It is possible to derive two very useful results from these mass balances.

### The *w*-*C** Model

First, in a spatially uniform wetland, as may occur after inlet settling effects no longer prevail, there will be no concentration gradient, and:

$$wC^* = G + R \tag{7.19}$$

where

$C^*$ = uniform downgradient concentration, g/m$^3$ = mg/L

Second, if it is assumed that generation and resuspension are constant over the entire wetland, Equation 7.16 may then be written, for the plug flow assumption, as

$$u\,h\frac{dC}{dx} = w(C^* - C) \tag{7.20}$$

Integration from inlet to outlet then gives

$$\frac{(C_o - C^*)}{(C_i - C^*)} = \exp\left(-\frac{wL}{uh}\right) = \exp\left(-\frac{w\tau}{h}\right) \tag{7.21}$$

where

$C_o$ = concentration, g/m$^3$ = mg/L
$C_i$ = concentration, g/m$^3$ = mg/L
$L$ = wetland length, m
$\tau$ = nominal detention time, d

The tanks-in-series (TIS) equivalent is (see Chapter 6):

$$\frac{(C_o - C^*)}{(C_i - C^*)} = \left(1 + \frac{wL}{Nuh}\right)^{-N} = \left(1 + \frac{w\tau}{Nh}\right)^{-N} \tag{7.22}$$

where

$N$ = number of TIS

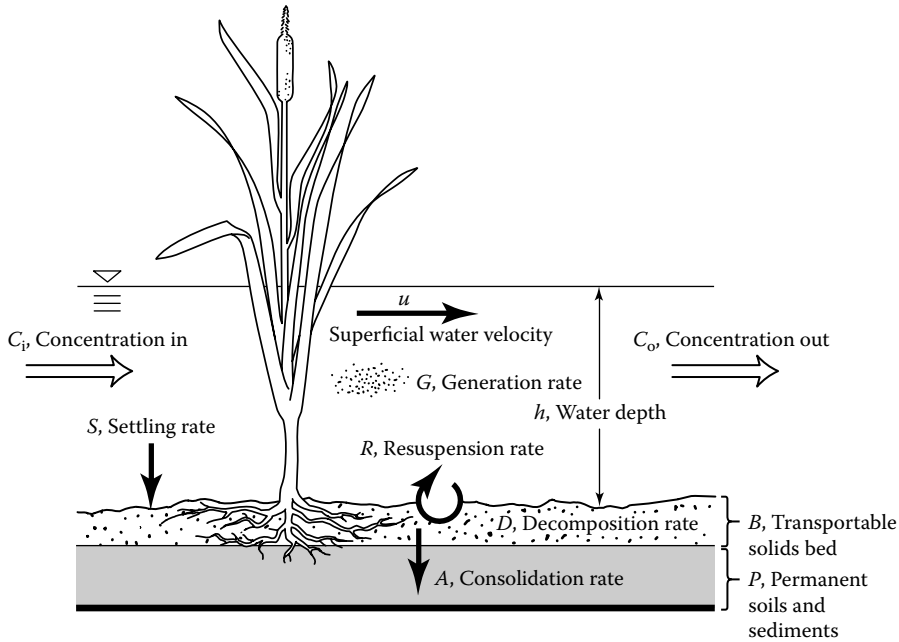



**FIGURE 7.14** Framework for mass balances on suspendable materials in the wetland environment. (Adapted from Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

Equation 7.21 contains a subtle message that bears on the removal of nearly all pollutants in wetlands, not just TSS. The right-hand numerator contains the settling velocity times the wetland length. An increase in either will cause a faster approach to $C^*$. The denominator contains the water velocity times the depth ($uh$). An increase in either of those will cause a slower approach to $C^*$. The detention time does not appear directly in this simplified mechanistic model, and the reason is easy to understand. If the water depth is doubled, for the same incoming volumetric flow rate and wetland area, the detention time will be doubled. But the particles do not fall any faster and now have twice as far to travel to the bottom. The extra detention time is used up by a greater vertical travel time. On the other hand, doubling the area of the wetland, all else being equal, will also double the detention time. The vertical settling distance is not increased, and the extra time causes greater removal.

A detailed gradient study to provide calibration of the $k$-$C^*$ model (as discussed in Chapter 6) was done at the Hallam Valley wetlands in Melbourne, Australia (Wong *et al*., 2006). Exceedingly high water flows (nominal HRT < three hours) were required to detail the rapid decrease of TSS. Model fits were excellent, with $w$-values in the range of 16–21 m/d, for both vegetated and unvegetated channels. However, the $C^*$-value for the unvegetated channel was about double that for that for the vegetated channel (60 versus 33 mg/L). This is consistent with resuspension being greater in the open channel (Equation 7.19). The rates of TSS removal in other continuous flow through wetlands are not quite exponential (Figure 7.15) The rapid initial declines in concentration prevail for only a brief time of travel, after which declines follow a slower pace. (The Hallam Valley study did not contain a long portion of wetland that could display such a slow decline.)

Thus it is clear that the TSS leaving an FWS treatment wetland of moderate to long detention is more reflective of generation and resuspension than of unsettled incoming solids. Therefore, for nearly all FWS data sets, the parameter $w$ cannot be determined accurately.

## Internal Cycling

The second feature of the mass balances is the ability to measure individual components of solids processing, and to combine them to infer other results. Data from the Des Plaines may be used in this way. Wetland EW3 was heavily loaded when the pump was operating and contained relatively sparse emergent vegetation. Independent measurements were made in settling columns, yielding $w$ = 9.7 m/d. Measurements of $R$ were made utilizing sediment cups plus input and output data, which gave $R = 46.0$ g/m$^2$·d. Estimates of $G = 1.6$ g/m$^2$·d (WRI, 1992). Accordingly, from Equation 7.19, the expected value of $C^* = 4.9$ g/m$^3$. Thus both $C^*$ and $w$ were estimated independently from the transect data for TSS. The predicted drop in TSS agreed quite well with the measurements.

This same data gives allows an approximation for the resuspension rate, and the net accretion rate (gross accretion less decomposition; Figure 7.16). The generation rates in this balance were estimated from measurements of productivity of the organisms in the water column and from biomass measurements. The striking feature of the mass balance is the large amount of solid material that is cycled, compared to inputs, outputs, or removals. Other studies have produced similar results (Table 7.5).

It may be concluded that in most instances, the effluent TSS from a FWS treatment wetland is determined by

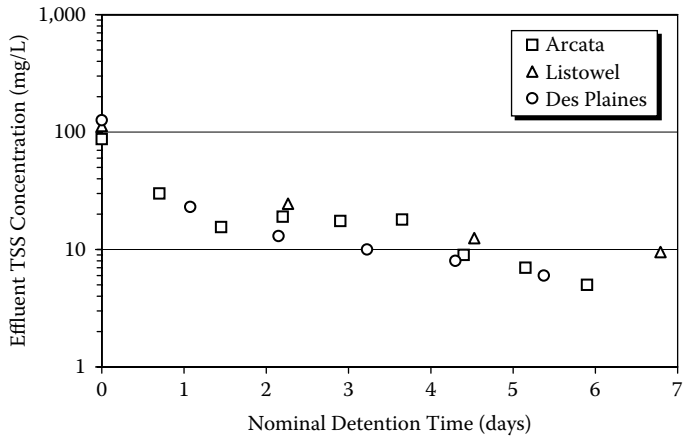



**FIGURE 7.15** Gradients in suspended solids along the flow direction in treatment wetlands. (Data for Arcata, California: Gearheart *et al.,* (1989) In *Constructed Wetlands for Wastewater Treatment: Municipal, Industrial, and Agricultural*. Hammer (Ed.), Lewis Publishers, Chelsea, Michigan, pp. 121–137; data for Listowel, Ontario: Herskowitz, (1986) *Listowel Artificial Marsh Project Report*. Ontario Ministry of the Environment, Water Resources Branch: Toronto, Ontario; data for Des Plaines, Illinois: unpublished data).

internal biological processes, and not by the removal efficiency for incoming TSS. As a corollary, the solids leaving the wetland will very often not be related to the solids entering, but rather to the detrital fragments originating internal to the system.

### Seasonal and Stochastic Effects

Because wetland effluent TSS is strongly related to internal ecosystem processes, random physical and biological events have pronounced effects on effluent concentrations. In addition, season and temperature are modifiers of the processes that generate and cycle solids. These effects may be separated by detrending the data, which typically follow a mild annual cycle with superimposed variability. The trend may be determined most accurately if there are data spanning many annual cycles, which may then be "folded" into one multiyear display and averaged.

TSS data time series often display some degree of sinusoidal behavior through the course of a calendar year. Therefore,

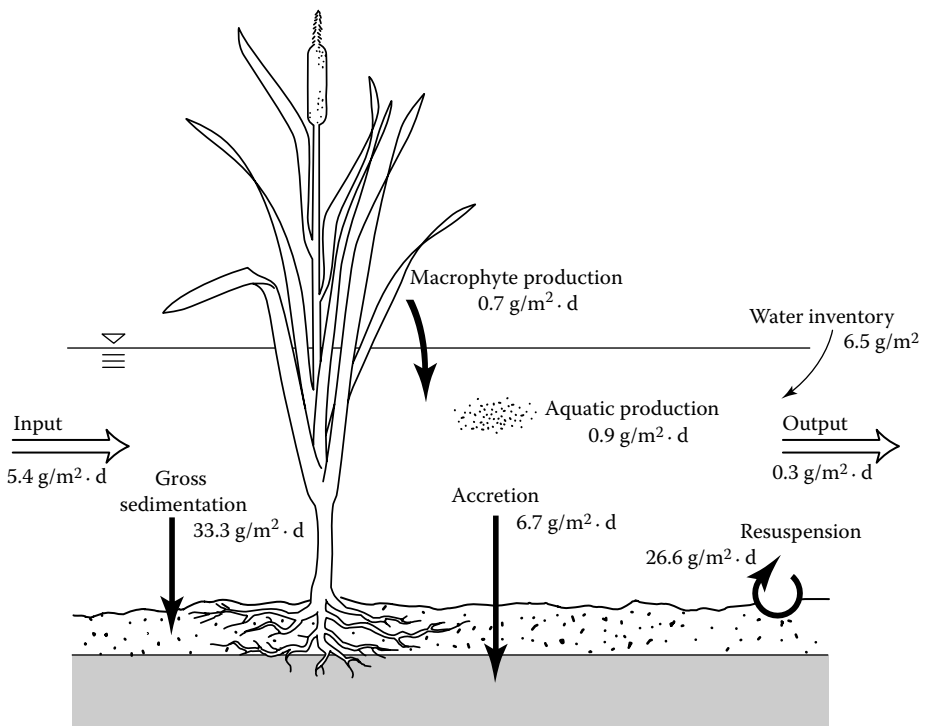



**FIGURE 7.16** Components of the sediment mass balance for wetland EW3 at Des Plaines, Illinois. The balance period is the 23-week pumping period in 1991. (Data from WRI (1992) *The Des Plaines River Wetlands Demonstration Project.* Report to U.S. EPA, July 1992. Wetlands Research Inc. (WRI), Chicago, Illinois.) (Adapted from Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

**TABLE 7.5**
**Cycling and Removal of TSS in FWS Wetlands**

| Site | Inflow ($g/m^2 \cdot d$) | Outflow ($g/m^2 \cdot d$) | Removed ($g/m^2 \cdot d$) | Generation ($g/m^2 \cdot d$) | Cycled ($g/m^2 \cdot d$) |
|---|---|---|---|---|---|
| Des Plaines EW3 | 5.4 | 0.3 | 5.1 | 1.6 | 26.6 |
| Houghton Lake Pre-discharge | 4 | 1 | 3 | 6 | 53 |
| Olentangy 1 | 4.7 | 2.7 | 2.0 | — | 95.3 |
| Olentangy 2 | 4.8 | 2.7 | 2.1 | — | 102.2 |
| Houghton Lake Discharge | 13 | 3 | 10 | 60 | 160 |

*Note:* The amounts cycled are far greater than the amounts removed.

*Source:* Data for Olentangy, Ohio: Harter and Mitsch (2003) *Journal of Environmental Quality* 32(4): 325–334; for Des Plaines, Illinois, and Houghton Lake, Michigan: unpublished data.

detrending may be accomplished by fitting the (folded) time series to

$$C = C_{\text{mean}}\left[1 + A\cos\left(\omega(t - t_{\max})\right)\right] + E \qquad (7.23)$$

where

$A$ = amplitude fraction
$C$ = concentration, $g/m^3$ = mg/L
$C_{\text{mean}}$ = concentration, $g/m^3$ = mg/L
$E$ = stochastic departure (error) of an individual measurement, mg/L
$t$ = Julian time, d
$t_{\max}$ = Julian time of TSS maximum, d
$\omega$ = annual frequency, 2/365, radians/d

The scatter of TSS data is large, and the trend typically accounts for less than 50% of the variability. An example of this model fit to data from the Arcata treatment marshes is given in Figure 7.17, for which $R^2 = 0.26$, implying that only 26% of the variability is accounted by the trend. The amplitude of the annual cycle for Arcata treatment wetlands was 0.32 times the mean. Examples may be found of both weaker and stronger annual trends, as indicated by lesser and greater $R^2$, with an average for the nine systems in Table 7.6 of $R^2 = 0.20 \pm 0.07$ (mean ± SE).

There is no strong indication of seasonality for the peaks of effluent TSS. These range from winter for Columbia, Missouri; Brighton, Ontario; Imperial, California; and Brawley, California, to autumn for Arcata, California; Cannon Beach, Oregon; and Estevan, Saskatchewan. Listowel, Ontario, peaks in the summer. Outlet peaks correspond only roughly to inlet peak times, with displacements of up to two months. It does not appear that either temperature or season alone is a sufficient predictor of the maximums and minimums of TSS. The temperature coefficient ($\theta$) set forth in Kadlec and Knight (1996) for wetland effluent TSS concentrations was derived from the Listowel, Ontario, data, and appears to be specific for that system. Based on information collected over the last ten years, it is apparent that effluent TSS concentrations vary

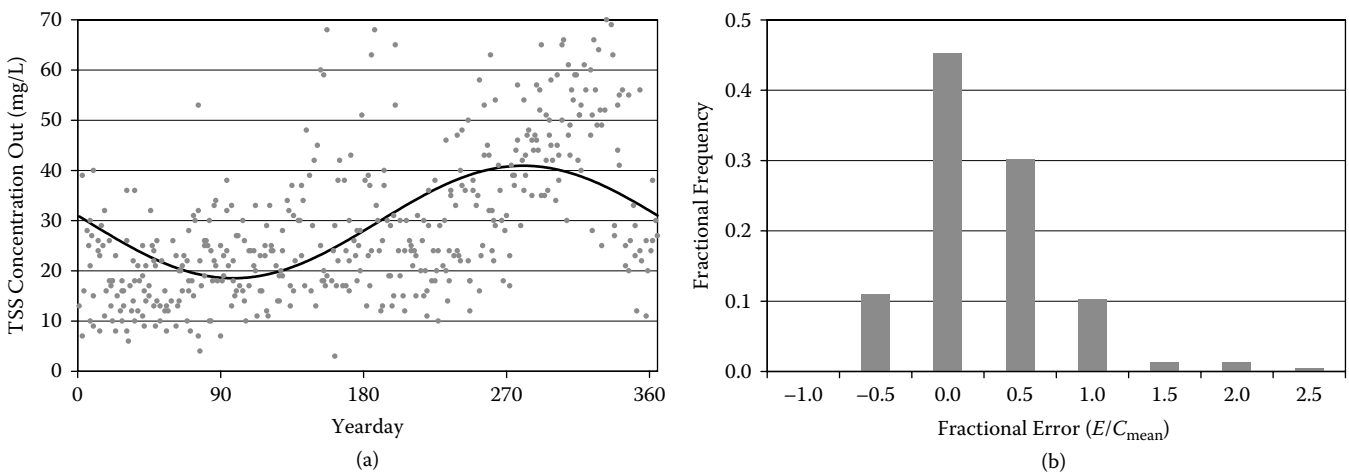



**FIGURE 7.17** Suspended solids leaving the Arcata treatment marshes versus day of the year (a). The departures from the sinusoidal trend line extend to 2.5 times the trend values, and are approximately log-normally distributed (b). Thirteen years of weekly data are represented (N = 443). (Data from TWDB database (2000) *Treatment Wetland Database (TWDB).* Website developed for U.S. EPA. http://firehole.humboldt.edu/wetland/twdb.html. Last updated November 2000. Compiled by B. Finney. U.S. EPA: Washington, D.C.)

**TABLE 7.6**
**Annual Trends in Wetland Effluent TSS**

| Site | | Period | Mean (mg/L) | Amplitude Fraction | Max (mg/L) | Min (mg/L) | $t_{max}$ (Julian day) |
|---|---|---|---|---|---|---|---|
| Arcata, California Treatment | I | Annual | 59 | 0.32 | 78 | 40 | 243 |
| Weekly 13 | O | | 29.7 | 0.38 | 41 | 19 | 280 |
| Arcata, California Enhancement | I | Annual | 27.2 | 0.24 | 34 | 21 | 284 |
| Weekly 14 | O | | 2.8 | 0.30 | 4 | 2 | 337 |
| Columbia, Missouri | I | Annual | 13.2 | 0.12 | 15 | 12 | 319 |
| Monthly 3 | O | | 8.1 | 0.43 | 12 | 5 | 20 |
| Brighton, Ontario | I | Annual | 14.3 | 0.63 | 23 | 5 | 47 |
| Weekly 4 | O | | 7.7 | 0.32 | 10 | 5 | 27 |
| Imperial, California | I | Annual | 35.9 | 0.16 | 42 | 30 | 116 |
| Weekly 3 | O | | 10.3 | 0.39 | 14 | 6 | 57 |
| Brawley, California | I | Annual | 18.1 | 0.42 | 26 | 10 | 92 |
| Weekly 3 | O | | 8.1 | 0.76 | 14 | 2 | 52 |
| Listowel 4, Ontario | I | Annual | 111 | 0.20 | 133 | 89 | 244 |
| Monthly 4 | O | | 7.2 | 0.64 | 12 | 3 | 176 |
| Cannon Beach, Oregon | I | Dry[a] (summer) | 56.0 | 1.3 | 71 | 31 | 212 |
| Monthly 16 | O | | 6.6 | 0.16 | 8 | 6 | 218 |
| Estevan, Saskatchewan | I | Summer[b] | 21.3 | 0.84 | 63 | 7 | 330 |
| Weekly 10 | O | | 9.5 | 0.11 | 11 | 9 | 330 |

*Note:* The frequency of sampling is either weekly or monthly as noted. The period record ranges from 3 years (Brawley and Imperial) to 16 years (Cannon Beach). The trend in each time series is presumed to be sinusoidal: $C = C_{mean}(1 + A\cos[\omega(t - t_{max})]) + E$

[a] The means of the full annual cycles are 31.0 and 6.6 mg/L.

[b] The means of the full annual cycles are 41.9 and 10.0 mg/L.

between FWS wetlands. Given this variability in performance response, it can be deduced that performance varies seasonally between FWS wetlands, in ways that are not directly related to temperature. *As a result, it is the current recommendation that no such temperature coefficient be used; essentially,* $\theta = 1.0$ *for TSS in FWS wetland systems.*

Because stochastic variability dominates the effluent TSS patterns, that variability requires quantification. For example, in the Arcata treatment marshes, the relative departures from the sinusoidal trend ($E/C_{mean}$) are approximately log-normally distributed (Figure 7.17). That type of distribution also prevails for other wetland sites, for TSS, and other water quality parameters. This occurs by virtue of the "squeeze" for low data values created by the nearness to the zero level (method detection limit, or MDL) of the parameter (Berthoux and Brown, 2002).

Because wetland effluent TSS distributions are only weakly seasonal, it is possible to ignore these trends, and to lump seasonal effects into the total variability. This is frequently done in the treatment wetland literature (e.g., U.S. EPA, 1999; Wallace and Knight, 2006). The frequency distributions of the inlet and outlet TSS measurements are displayed graphically. Figure 7.18 shows an example of this procedure, derived from the same data as Figure 7.17. Note that the 50th percentile represents the median of the data, not the mean. Further note that these are not paired point graphs, so that reductions cannot be computed at any specified frequency level.

It is useful to examine the multiplier factors associated with the various (higher) percentiles of the effluent distributions, because these may well be involved in permitting or licensing of the treatment wetland. Examples of these outlet multipliers are shown in Table 7.7, for a sampling of wetlands spanning a range of inlet concentrations from 1 to 100 mg/L. It may be seen that in several instances, excursions of outlet concentrations exceed the average inlet concentration, despite long-term average concentration reductions. It is only when the inlet TSS reaches about 25 mg/L that not more than 10% exceedances of the inlet concentration occur.

## Input–Output Relations

Suspended solids have been measured at inlets and outlets for a large number of FWS wetlands. It is instructive to examine this large interwetland data set, to ascertain the existence

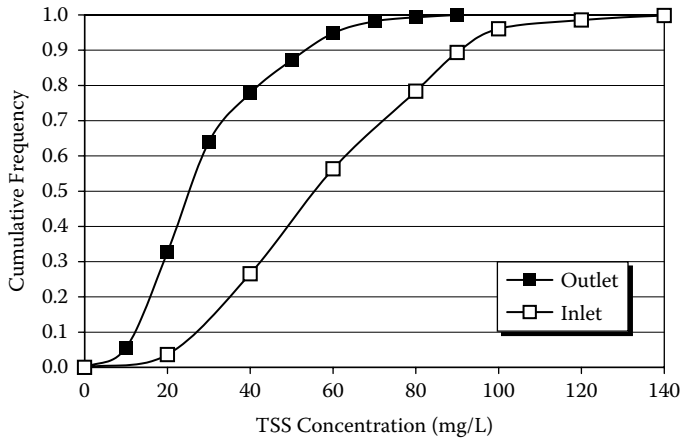



**FIGURE 7.18** Probability distributions for inlet and outlet TSS for the Arcata treatment wetlands. The median inlet TSS was 56 mg/L; the median outlet TSS was 25 mg/L. Data were weekly for 13 years. (Data from TWDB database (2000) *Treatment Wetland Database (TWDB).* Website developed for U.S. EPA. http://firehole.humboldt.edu/wetland/twdb.html. Last updated November 2000. Compiled by B. Finney. U.S. EPA: Washington, D.C.)

of trends among systems. A popular method of TSS data representation is the quotation of percentage removal, or removal efficiency. However, the presence of a background TSS level constrains removal efficiency to be below a level dictated by the inlet and background concentrations. As a consequence, percent removal is an inadequate measure for many treatment wetlands. Indeed, some efficiencies are negative, in situations where pretreatment includes removal of TSS prior to the wetland, because influent TSS concentrations are below the wetland background concentrations.

For these reasons, it is preferable to consider graphical exposition of intersystem data, and to derive generalities therefrom. Two choices exist:

1. The input–output concentration graph
2. The outlet concentration–inlet loading graph

Intersystem outlet concentrations apparently increase with the areal loading of TSS to the wetland, with higher outlet concentrations at higher loading rates (Figure 7.19). U.S. EPA (2000a) found a similar pattern for a restricted set of

**TABLE 7.7**
**Trend Multipliers for TSS Distribution of FWS Wetland Effluents**

| Percentile | Inlet 50 (mg/L) | Outlet 50 (mg/L) | Excursion Frequency 80% | 90% | 95% | 99% |
|---|---|---|---|---|---|---|
| *Orlando Easterly, Florida* | *1* | *1* | *2.52* | *4.20* | *8.54* | *19.63* |
| *Commerce Township, Michigan* | *1* | *9* | *1.60* | *1.96* | *2.44* | *3.62* |
| Tres Rios, Arizona H1 | 3 | 3 | 2.00 | 2.33 | 3.63 | 5.05 |
| Brighton, Ontario | 10 | 6 | 2.00 | 2.50 | 3.11 | 6.21 |
| Estevan, Saskatchewan | 11 | 6 | 2.17 | 3.33 | 4.30 | 9.12 |
| Columbia, Missouri | 12 | 6 | 1.64 | 2.26 | 3.70 | 5.30 |
| New Hanover, Michigan | 13 | 8 | 1.33 | 1.73 | 1.88 | 3.86 |
| Brawley, California | 18 | 8 | 1.69 | 1.79 | 1.82 | 1.83 |
| Listowel, Ontario 3 | 18 | 6 | 1.87 | 2.72 | 3.66 | 4.72 |
| Arcata, California Enhancement | 26 | 3 | 1.23 | 1.28 | 1.29 | 1.30 |
| Imperial, California | 32 | 10 | 1.33 | 1.37 | 1.39 | 1.40 |
| Tarrant, Texas WC1 | 39 | 5 | 1.74 | 2.48 | 2.76 | 3.78 |
| Arcata, California Treatment | 56 | 25 | 1.70 | 2.12 | 2.44 | 2.99 |
| *Cannon Beach, Oregon* | *58* | *6* | *2.00* | *2.43* | *2.88* | *4.39* |
| Des Plaines, Illinois EW3 | 83 | 7 | 1.64 | 1.83 | 2.09 | 3.03 |
| *Listowel, Ontario 4* | *100* | *5* | *1.60* | *3.08* | *3.88* | *5.80* |
| **Mean (ex. Orlando)** | | | **1.70** | **2.21** | **2.75** | **4.16** |

*Note:* The 50th percentile is the median, not the mean. Frequencies are weekly, or monthly (italics). Orlando Easterly data is strongly left-censored, with an MDL of 1.0 mg/L. Trend multiplier is $(1 + \Psi)$; see Equation 6.61.

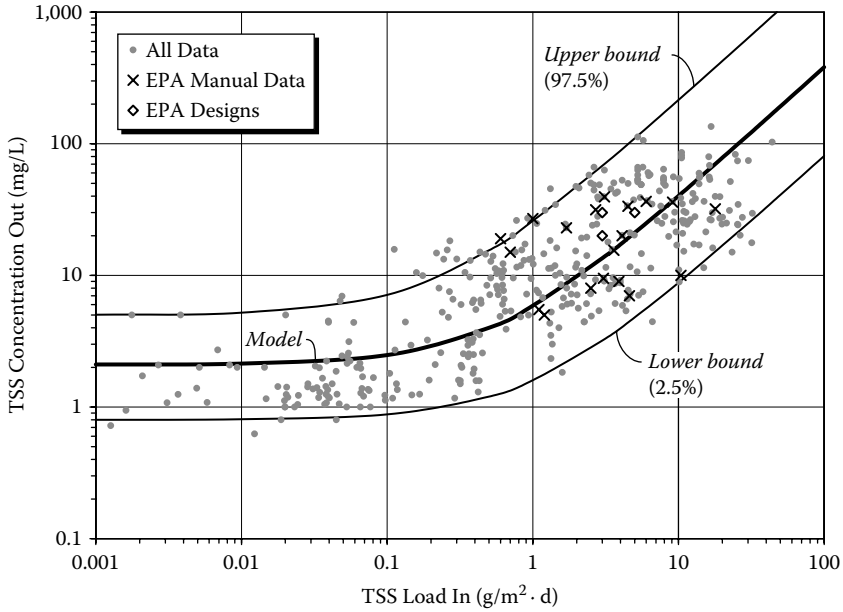



**FIGURE 7.19** Load response for TSS in FWS wetlands. Data represent one point for one wetland for one year. N = 388, for 136 wetlands. Diamonds represent EPA design recommendations (U.S. EPA, 2000a). The central line is a linear regression, with $R^2 = 0.65$ for the logarithmic basis shown. The upper line represents 97.5% bound of the data; the lower line represents the 2.5% bound. Fitting parameters for TSS = (A + (B × Loading)) are:

| | A (mg/L) | B (mg/L)/(g/m²·d) |
|---|---|---|
| Upper bound | 5 | 21 |
| Central tendency | 2.1 | 3.8 |
| Lower bound | 0.8 | 0.8 |

data, which are also shown on Figure 7.19. At any given loading rate, the data cloud spans about a factor of 10 in outlet concentrations. The central tendency and upper and lower bounds are shown, together with the corresponding regression equations. However, this view of system performance is very misleading.

When data from a given site are examined, a different picture emerges. Figure 7.20 shows results from six different side-by-side tests at four locations, each of a year or more duration. Different TSS loadings were achieved by varying the hydraulic loading. Depth, source water, and meteorology and other site factors were invariant within each group of data. In each group, the spread of the inlet TSS loadings was a factor of 5–10. An interesting and important observation is that there is essentially no increase in outlet TSS with TSS loading within each group. Therefore, *TSS loading is an inappropriate correlating parameter for prediction of outlet TSS.*

*This means* the *k-C* model* (*as described in Chapter 6*) *is dominated by C**. For the *k-C** model, we expect to see an "S" curve on the loading graph, with *C** as one asymptote and $C_i$ as the other asymptote. In contrast, the FWS wetlands analyzed in Figure 7.19 never approached $C_i$ as the hydraulic retention time ($\tau$) was decreased; the wetlands continued to return an outlet TSS concentration that is a function of internal TSS generation ($G + R$), which is represented by *C**.

Other factors most responsible for the large differences in outlet TSS concentrations at the same inlet loading and the likely candidates are inlet TSS concentration and inlet nutrient status. These two factors often go hand-in-hand, and there is not yet a study that has identified the relative importance. High inlet TSS concentrations could be partially short-circuited to the wetland outlet, or high nutrients could cause more internal generation of TSS.

At this point in the history of treatment wetland technology, we are only left with the possibility of input–output regression relationships to predict output TSS concentrations. A large intersystem data set for annual values is shown in Figure 7.21, together with regression lines for the central tendency, and upper and lower bounds set to confine the middle 95% of the data. Regression of the annual information produces the following correlation:

$$C^* = C_o = 1.5 + 0.22\,C_i \qquad (7.24)$$

where

$R^2 = 0.65$ for logarithmic data, N = 443
$0.2 < C_i < 1{,}910$ mg/L
$0.6 < C_o < 135$ mg/L

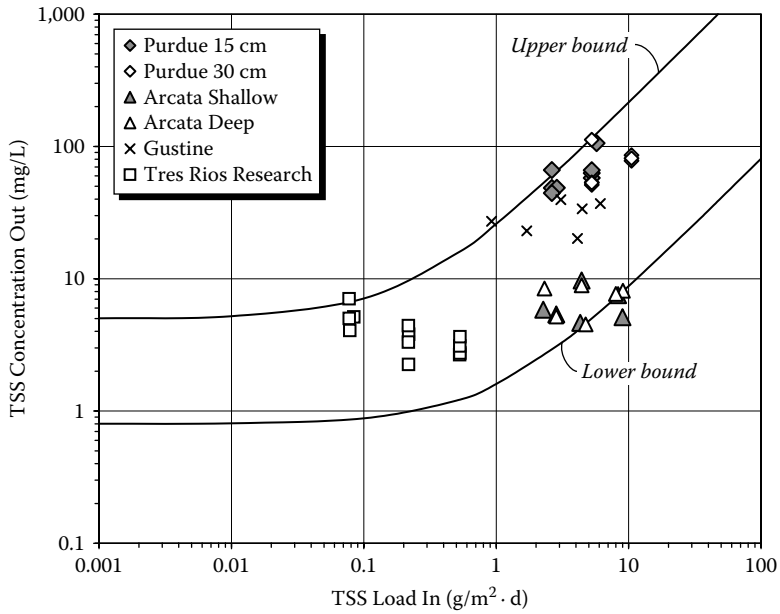



**FIGURE 7.20** Intrasystem behavior of FWS wetlands in response to loading changes. Each point represents one year of data, and all cell clusters were run side by side.

| Site | No. Cells | Depth (cm) | HLR (cm/d) | TSS Inlet (mg/L) | $NH_3$-N Inlet (mg/L) |
|---|---|---|---|---|---|
| Purdue 15 cm | 8 | 15 | 2–4 | 135–145 | 400 |
| Purdue 30 cm | 7 | 30 | 4–8 | 135–145 | 400 |
| Gustine | 6 | 45 | 1–4 | 75–150 | 14–20 |
| Arcata Shallow | 6 | 30–40 | 6–25 | 36 | 9–14 |
| Arcata Deep | 6 | 49–61 | 6–25 | 36 | 9–14 |
| Tres Rios Research | 12 | 41–62 | 2–15 | 4 | 2 |

## Open Water Areas

Curiously, the subject of inclusion of open water areas in FWS treatment wetland systems has been bifurcated into

1. Deep zones inside the wetland
2. Ponds preceding wetlands

An inlet deep zone inside the wetland is essentially a pond located inside the wetland boundary. Ponds function to settle incoming TSS, but are conducive to the production of TSS via algal cycling, as discussed in Chapter 3. Internal and outlet open water areas are settling zones, but are subject to wind resuspension and algal growth.

U.S. EPA (2000a) suggests that open zones be incorporated into treatment wetlands as a means of enhancing TSS removal, along with other purposes. The reason given is that these "…can provide conditioning and transformation processes which may improve overall removal of TSS…." "Open zones" may contain submerged vegetation, or be devoid of plants. This differentiation is very important, because open zones with submerged aquatic vegetation (SAV) will provide TSS reduction benefits, whereas unvegetated open zones will not, and may in fact increase TSS.

## Pond–Wetland Combinations

Because incoming TSS is rapidly settled and filtered in the wetland environment, it is possible and desirable to provide a first element of the treatment wetland complex that traps the fastest settling fraction of the suspended material. A pond provides for that presettling and is more easily cleaned than an emergent or submergent macrophyte bed. It is further desirable to collect solids and their partitioned metals and chemicals in a location that is not foraged by sediment-feeding vertebrates. This presettling pond may require infrequent dredging to remove the accumulated deposits.

Data from the Tarrant County, Texas, site (APAI, 1995) illustrates the mean performance of three parallel marsh wetland cell trains of three cells each, following two parallel unvegetated settling ponds. The settling ponds occupied 15% of the area, but accounted for 94%–97% of the solids removal (Figure 7.22). The first wetland cell completes the solids removal; the remaining two cells do not reduce TSS any further. The last wetland cells were, however, needed for phosphorus removal. The performance of five pond–wetland systems is summarized in Table 7.8. The large majority of

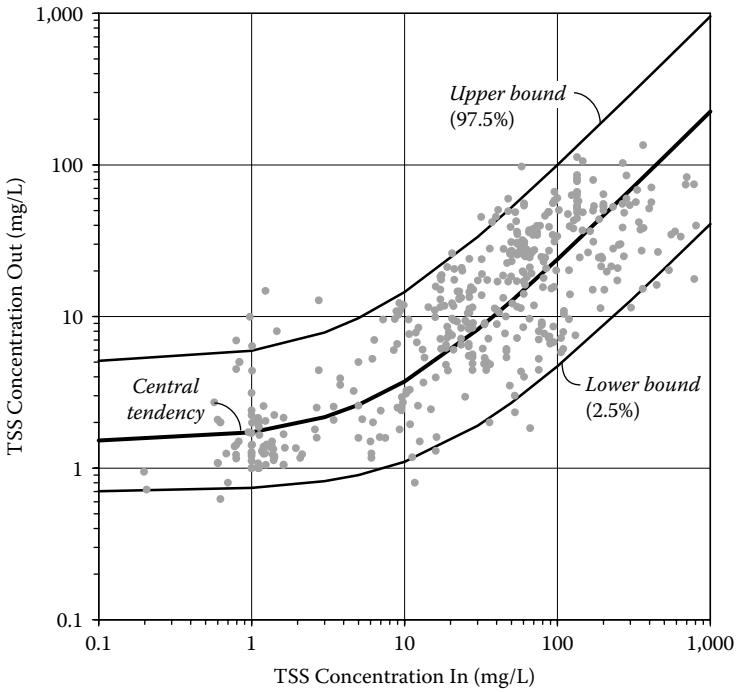



**FIGURE 7.21** Input–output plot for TSS in FWS wetlands. Data represent one point for one wetland for one year, N = 443, for 142 wetlands. The central line is a linear regression, with $R^2 = 0.65$ for the logarithmic basis shown. The upper line represents 97.5% bound of the data; the lower line represents the 2.5% bound. Fitting parameters for $C_o = (A + (B \times C_i))$ are:

| | A (mg/L) | B (Dimensionless) |
|---|---|---|
| Upper bound | 5 | 0.95 |
| Central tendency | 1.5 | 0.22 |
| Lower bound | 0.7 | 0.04 |

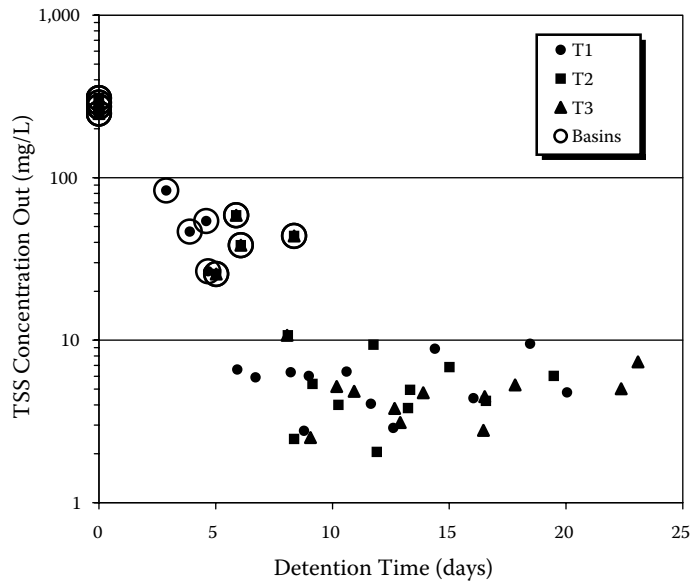



**FIGURE 7.22** Profiles of TSS along the flow direction in the Tarrant treatment trains over a four-year period. Note that the wetlands exhibit a plateau, or background TSS, below 10 mg/L. (From APAI (1995) *The use of constructed wetlands for protection of water quality in water supply reservoirs*. Final report by APAI (Alan Plummer and Associates, Inc.) to the American Water Works Association Research Foundation and the Tarrant County Water Control and Improvement District No. 1, AWWA: Denver, Colorado.)

**TABLE 7.8**
**TSS Removal in Systems with a Presettling Basin Followed by a Wetland**

| | | | Sedimentation Basin | | Wetland | |
|---|---|---|---|---|---|---|
| Site | Sed Basin (% Area) | TSS In (mg/L) | TSS Out (mg/L) | Load Removed ($g/m^2 \cdot yr$) | TSS Out (mg/L) | Load Removed ($g/m^2 \cdot yr$) |
| Tarrant, Texas 1 | 12 | 276 | 46 | 20,570 | 6 | 1,312 |
| Tarrant, Texas 2 | 15 | 276 | 37 | 21,993 | 11 | 1,181 |
| Tarrant, Texas 3 | 15 | 276 | 28 | 22,871 | 6 | 742 |
| Brawley, California | 25 | 216 | 35 | 21,585 | 12 | 858 |
| Imperial, California | ≈30 | 200 | 18 | 10,055 | 7 | 1,418 |

*Note:* All systems were run for more than three years, and had four to seven days' detention in the sedimentation basins.

TSS in these systems was retained in a minority fraction of the total footprint.

The placement of a pond as the final element in a wetland treatment system is generally not desirable from the standpoint of TSS reduction. The planktonic production in such a pond is typically quite high, leading to the reintroduction of high-chlorophyll microdetritus, much of which remains in suspension. An example of this phenomenon was the Lakeland, Florida, system. Entering TSS was reduced in the first marsh cells but was regenerated in later, open water cells because of planktonic activity (Bays *et al*., 1993).

## Deep and Open Water: Unvegetated Zones

A treatment wetland may include an inlet deep zone, within the footprint of the wetland. Such a feature is in effect a pretreatment pond element, as discussed in the previous section. In contrast, the wetland may also contain internal deep zones, ranging from narrow ditches to large expanses of open water. Both the number and size of such zones have been variable across a number of projects. Here, we examine the efficacy of such zones in TSS reduction.

Moore and Niswander (1997) operated a set of six treatment wetlands receiving diluted dairy wastewater for two years. Two wetlands had a central internal deep zone comprising 45% of the area, whereas the other four had no internal deep zone. Data for the second year, past the start-up period, were analyzed for differences in performance. The wetland areas were all identical, and hence the hydraulic loading to all six wetlands was uniform at 3.95 cm/d. For the two cells with deep zones, the average TSS influent of 653 mg/L was reduced to 202 mg/L For the four cells without deep zones, the average TSS influent of 653 mg/L was reduced to 195 mg/L. The authors concluded that deep center sections did not show any significant impact on treatment efficiency.

Knight *et al*. (1994) operated a set of six treatment wetlands receiving treated paper pulp mill effluent for two years. Two internal deep zones comprising 25%, 35%, and 45% of the area were included in three of the six. There was variability in flow (different HLRs) and aspect ratios (2.5, 5, and 10). If the pairs of cells with the same aspect are compared on the basis of deep zones, the concentrations produced were lower, and the load removed was higher, in wetlands with deep zones in two of the three pairs.

Eidson *et al*. (2005) studied side-by-side wetlands at Augusta, Georgia. Six 12-ha cells were studied, two of a marsh–pond–marsh configuration (60% marsh), and four of a marsh–ditch configuration (82% marsh). Incoming water was very low in TSS, averaging 3.3 mg/L over a one-year period. The marsh–pond–marsh cells averaged 12.2 mg/L at their outlets, whereas the marsh–ditch cells averaged 4.1 mg/L. The effect of the large open water areas in the marsh–pond–march cells was to raise the TSS concentration, presumably due to generation and resuspension. The marsh–ditch systems created a negligible increase.

At the Tres Rios, Arizona, project, Kadlec (2007) studied the effect of deep zones in a triplicated side-by-side study. The research site contained a set of twelve 0.12-ha (24 m wide × 50 m long) wetlands, built in a triplicate design with 2, 3, 4, and 5 deep zones, including one at the inlet and one at the outlet. No significant difference could be found in outlet TSS concentrations ($\alpha = 0.05$).

Further insights can be gained from systems that lost their vegetation over the course of time. A FWS treatment wetland in Commerce Township, Michigan, was "eaten out" by muskrats and waterfowl, leaving virtually no emergent or submergent vegetation. Before the loss, the effluent TSS was 5 mg/L; after, it was 13 mg/L.

The Tres Rios Hayfield wetlands are a second example (Kadlec, 2006). During spring and summer of the third year after start-up, the vegetation essentially all died, for reasons that have not been resolved, and regrowth did not occur. Incoming TSS was low (3 mg/L), and remained relatively low during the vegetative period (2 mg/L) (Figure 7.23). After loss of vegetation, effluent TSS climbed to 27 mg/L. This wetland had unvegetated deep zones. In both cases it is apparent that the effluent solids could not have derived from incoming TSS, but rather were the result of internal generation and resuspension.

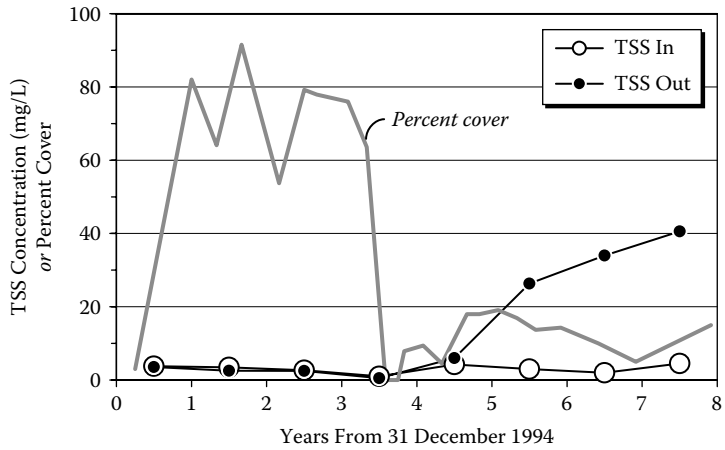



**FIGURE 7.23** The transition of Tres Rios wetland H1 from a vegetated to an unvegetated state. Outlet TSS averaged 2 mg/L for the first four years, then climbed to 40 mg/L in year eight.

### Submerged Aquatic Vegetation (SAV)

It is well known that SAV reduces resuspension in lake environments (James and Barko, 2000; James *et al*., 2001; 2002; Horppila and Nurminen, 2003). In shallower wetland environments, SAV would presumably serve that same function, and would provide the additional benefits ascribable to the submerged portions of emergent vegetation. Consequently, TSS will be generated in SAV systems.

Very few studies of TSS behavior in SAV beds have been reported. DB Environmental (DBE, 1999) measured 2 mg/L in, and 3 mg/L out, of SAV mesocosms treating agricultural runoff. However, the annual accretion rate of new sediments was 1.0 cm/yr, again indicating that internal generation was a dominant mechanism. Toet (2003) measured turbidity in a set of nine side-by-side wetlands receiving highly treated municipal effluent. The front halves were vegetated with *Typha* (4) and *Phragmites* (4). The back halves were vegetated with SAV (*Elodea*, *Ceratophyllum*, and *Potamogeton*), and there was an open water control. The front sections increased turbidity from 3 to 6 NTU, and the back sections provided a slight further increase to 7 NTU.

Based upon this limited wetland information, it appears that SAV beds have about the same background TSS as emergent wetlands; but upon lake information, it is expected that SAV will help prevent resuspension. Overall, the current expectation is that SAV beds will behave approximately like emergent systems for TSS processing.

## 7.4 PARTICULATE PROCESSES IN HSSF WETLANDS

Although HSSF wetlands are configured very differently than FWS wetlands, the same physical processes apply to different degrees and to different magnitudes. Processes that affect the removal and generation of particulate matter in HSSF wetlands are discussed in this section. Like FWS wetlands, HSSF wetlands are very effective in trapping and retaining TSS associated with the inlet flows. Unlike FWS wetlands, this accumulated TSS material reduces the hydraulic conductivity of the wetland, often to a significant degree. Bed clogging that occurs in HSSF wetlands as a result of TSS accumulation has often led to hydraulic failure and associated flooding of the wetland bed, which remains a significant operation and maintenance challenge to this day.

### Particulate Settling

Like FWS wetlands, HSSF wetlands are very effective at removing TSS associated with the inlet flow. One of the primary mechanisms is gravitationally driven particulate settling. This has already been discussed in detail for FWS wetlands (Equations 7.1–7.7). Because the bed porosity in HSSF wetlands is low ($\varepsilon = 0.30–0.40$) relative to FWS wetlands, it is useful to consider gravitational settling in terms of the actual flow velocity ($v$) rather than the superficial flow velocity ($u$). Thus, Equation 7.6 can be rewritten as

$$t_{travel} = \frac{L}{v} \tag{7.25}$$

where

$L$ = wetland length, m
$t_{travel}$ = time to traverse wetland, s
$v$ = actual flow velocity, m/s ($v = u/\varepsilon$)
$u$ = superficial flow velocity, m/s
$\varepsilon$ = bed porosity, dimensionless

Theoretically, all particles of a size corresponding to a given fall velocity will be removed by settling if the travel time exceeds the settling time. In FWS wetlands, the *fall distance* is approximated as the overall water depth within the wetland. In HSSF systems, the wetland is filled with a granular

bed. The porosity of this bed increases the flow velocity ($v > u$), but decreases the fall distance, because the particle only has to fall the distance of the average pore space before hitting an intercepting surface, not the entire depth of the wetland bed. In most instances, the pore size within a HSSF wetland bed can be approximated by the $d_{10}$ of the bed media (90% of the particles within the bed are larger than the $d_{10}$). Thus, Equation 7.7 can be rewritten as

when

$$\frac{L}{v} > \frac{d_{10}}{w}$$

$$N_{\text{fall}} = \frac{Lw}{vd_{10}} > 1 \tag{7.26}$$

where
$L$ = wetland length, m
$v$ = actual flow velocity, m/s
$d_{10}$ = particle size representing the smallest 10% of the bed media
$w$ = terminal solids settling velocity, m/s
$N_{\text{fall}}$ = particle falling number, dimensionless

As a practical matter, generally the falling rate ($w$) is much greater than the actual flow velocity ($v$), ($w >> v$). As a result, virtually all the particles associated with the influent wastewater are settled out, generally within the first 5% of the wetland bed (Puigagut *et al*., 2006).

### Filtration and Interception

As discussed for FWS wetlands, the principal mechanisms of granular bed filtration are well known and documented in handbooks (see, e.g., Metcalf and Eddy Inc., 1991; Crites and Tchobanoglous, 1998). These include:

1. Inertial deposition, or impaction—particles moving fast enough that they impact bed particles rather than being swept past by the flowing water.
2. Diffusional deposition—random processes at either microscale (Brownian motion) or macroscale (bioturbation) which move a particle to an immersed surface.
3. Flow line interception—particles moving with the water and avoiding head-on collisions, but passing close enough to graze the stem and its biofilm, and sticking.

Media size in HSSF wetlands around the world ranges from soils ($d_{10} < 0.1$ mm) up to coarse gravels ($d_{10} > 4$ mm). This size range in bed media spans the dominant scale factors of Mechanisms 1–3 listed above. For fine-grained bed media, Mechanisms 1 and 2 will predominate. For gravel media, Mechanism 3 will be the most important.

As a practical matter, these mechanisms all combine to preferentially remove incoming TSS in the inlet region of the HSSF bed. For fine-grained media, Mechanisms 1 and 2 remove particles almost immediately. In coarser bed (gravel) systems, Mechanism 3 will predominate, and will work in conjunction with the particulate settling mechanisms just described.

### Resuspension

In contrast to FWS wetlands, resuspension mechanisms are strongly minimized in HSSF wetlands due to the physical configuration of the HSSF reactor. Flow velocities within the HSSF bed are low, and generally do not generate shear stresses sufficient to scour particulate matter. As flow in HSSF wetlands occurs below the top of the bed, resuspension mechanisms such as wind mixing and turbulence are not factors. Similarly, bioturbation (from burrowing rodents) and gas lift, although theoretically possible, occur at such small localized scales, that their effect on the overall wetland is nil. As a result of these factors, resuspension is generally not a significant phenomenon in HSSF wetlands.

### Chemical Precipitation

Reaction chemistry as noted previously for FWS wetlands can also occur in HSSF wetlands. One use of HSSF wetlands has been as sulfate-reducing systems to induce the precipitation of copper, nickel, and other metals (Eger, 1992). Many metals form highly insoluble sulfide precipitates (Palmer *et al*., 1988), as discussed in Chapter 11. A peat-bed HSSF wetland has been used since 1986 to remove copper and nickel from mine drainage at the LTV Dunka Mine near Hoyt Lakes, Minnesota (Eger and Lapakko, 1989; Frostman, 1993).

Other than HSSF wetlands treating mine wastes (Younger *et al*., 2002), accumulation of chemical precipitates generally does not occur at a rate significant enough to impact the hydraulic conductivity of the HSSF wetland bed.

### Production of Biological Solids

Although HSSF wetlands are effective in removing influent suspended solids through settling, interception, and filtration, and may generate small amounts of solids through chemical precipitation, the majority of the particulate matter present in a HSSF bed treating primary or secondary domestic wastewater consists of biological solids that are generated internally within the system. These consist of

1. Plant detrital material (including associated microbial and fungal networks)
2. Microbial films present on bed media particles

#### Plant Contributions

Cumulative experience with HSSF wetlands indicates that deeper gravel beds (>40 cm) will contain an upper zone

that contains essentially all the plant roots and a lower zone without roots. The presence of root blockage is an important factor: the root zone impedes flow more than the relatively clean media below it. Several tracer studies have documented this phenomenon (Fisher, 1990; Pilgrim *et al*., 1992; Tanner and Sukias, 1995; Marsteiner *et al*., 1996; Tanner *et al*., 1998a; Drizo *et al*., 2000; García, 2003).

Apart from the living root and rhizome material, the upper layer of an HSSF wetland may contain significant amounts of organic matter associated with the plants. For example, Tanner and Sukias (1995) found that planted wetlands developed at least twice as much organic matter in the top 10 cm, compared to unplanted replicates, over a 22-month period. It was not determined whether this material was generated by above- or belowground plant activity.

### Microbial Contributions

The solids in HSSF wetlands originate from particulates (filtration) and from living and dead microbial biomass (biosolids = sludge). Microbial biomass forms in response to both particulate and soluble organic loading rates. These biofilms further entrap both organic and inorganic solids (Winter and Goetz, 2003), forming a composite material. In soil absorption systems, this material is contained in a layer commonly termed a *biomat* (Crites and Tchobanoglous, 1998; Beal *et al*., 2004). Others have designated it as *sludge* (Cooper *et al*., 2006a) or *biosolids* (Ragusa *et al*., 2004). Internal solids accumulation can also be affected by chemical phenomena such as sulfide precipitation (Liebowitz *et al*., 2000), and varies in different applications depending on the nature of the waste being treated. Acknowledging that wetlands internal solids are often mostly organic, and are spatially distributed in at least two dimensions, we opt for calling these internal bed materials *biosolids*.

Biosolids formation is greatest at the inlet end of the wetland where the organic loading is highest (Ragusa *et al*., 2004). The loss of pore volume due to biomat formation reduces the hydraulic conductivity in this inlet zone (Zhao *et al*., 2004) (see Chapter 2). Organic matter is removed as wastewater flows through the wetland, resulting in declining biosolids growth. At the outlet, where only small quantities of soluble organic matter are available to the microbes and fungi, biosolids formation is minimal. The nonuniform distribution of internal biosolids along the length of the bed results in a nonuniform distribution of hydraulic conductivity throughout the bed, as discussed in Chapter 2.

### Accretion and Bed Clogging

The combined effects of particulate settling, filtration, and interception result in highly efficient trapping of TSS within the inlet region of the HSSF bed. Wetland plants root preferentially within the upper regions of the HSSF bed, obstructing flow in this region. The loading of organic matter, for systems treating domestic wastewater, in both soluble and particulate forms, results in the preferential development of microbial biomats in the inlet region of the HSSF wetland bed. The net result of these mechanisms is a highly nonuniform distribution of solids, plant roots, microbial activity, and associated reductions in hydraulic conductivity, as discussed in Chapter 2. Eventually, this inlet zone may become clogged, and the bed will develop overland flow in this region.

Clogging can occur just from deposited particulate (mineral + organic) material. In a laboratory experiment, Sun (1998) was able to demonstrate that when enough sawdust was added to a flume containing pea rock (effectively reducing the porosity from 39.5% to 33.4%), the resulting head loss was controlled by the particulate matter, not the bed media (Sun *et al*., 1998). Porosity reduction due to particle trapping provides reasonable estimates of the time to clogging (Blazejewski and Murat–Blazejewska, 1997).

Most organic matter is removed in the inlet zone of the HSSF wetland bed. This is the zone of heaviest biosolids accumulation, where the greatest reductions in hydraulic conductivity occur. This zone can be termed the *biosolids clogging distance* and is analogous to the clogging mat that develops in soil infiltration systems treating septic tank effluent (U.S. EPA, 2002c). A schematic of the clogging phenomenon is shown in Figure 7.24.

In fine-grained materials, there is greater bed particle surface area available per unit length of flow path. As a result, more microbial biofilm can form in response to the organic loading. Because the pore size is smaller, the biosolids are more effective in entrapping organic and inorganic solids (as discussed under the Filtration and Interception section above). If the resulting accumulation completely fills the pore spaces, the hydraulic conductivity is controlled (reduced) by the characteristics of the biosolids and not by the characteristics of the media. In this case, the wastewater will likely surface. Consequently, fine-grained media such as HSSF soil filters are unlikely to avoid clogging and the associated flooding, and overland flow.

With coarse bed materials, there is less surface area available for biofilm formation per unit length of flow path. Due to the larger pore spaces, the biosolids cannot completely fill the pore volume, and effective flow paths through the media still exist. The net effect lengthens the biosolids penetration distance but decreases the potential for plugging (Zhao *et al*., 2004). This concept is illustrated in Figure 7.24.

Progressive accumulations of biosolids can lead to a progressive clogging failure of the wetland bed, and the HSSF wetland will end up functioning as an overland flow treatment system, as illustrated in Figure 7.25. This mode of hydraulic failure has occurred in many HSSF wetlands (see Figure 2.28).

## 7.5 TSS REMOVAL IN HSSF WETLANDS

The fate and transport of TSS in HSSF wetlands are under the same physical principles as in FWS wetlands. Consequently, Equations 7.15–7.21 can be applied to HSSF wetlands, with the exception that the actual flow velocity ($v$) is used instead of the superficial flow velocity ($u$). *Note,* $v = u/\varepsilon$, *where* $\varepsilon$

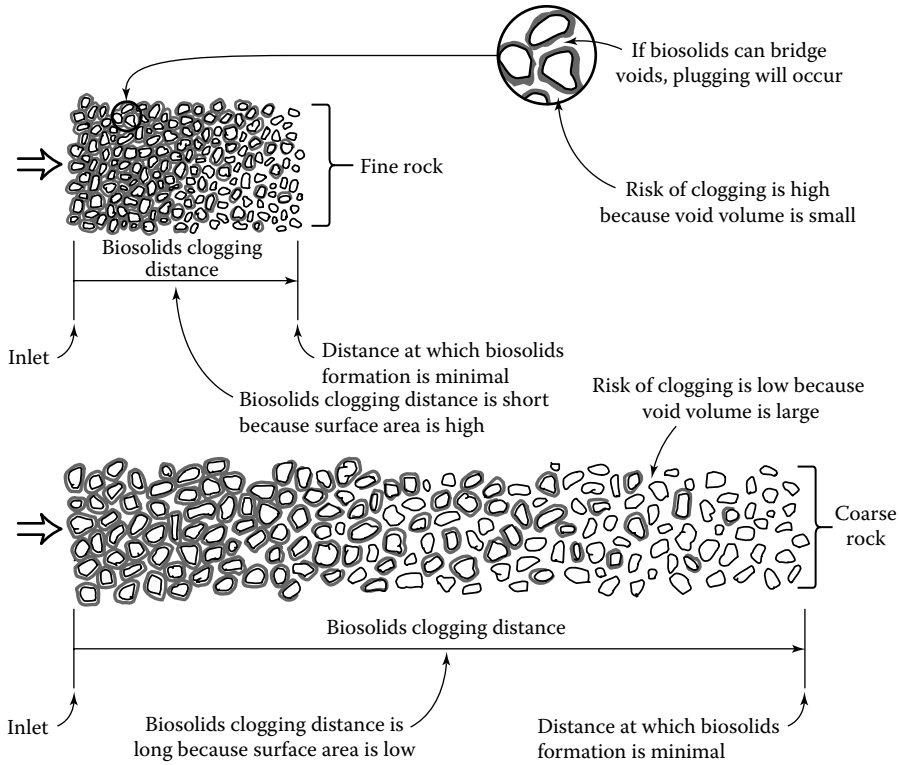



**FIGURE 7.24** Biosolids clogging distance as a function of media size. (From Wallace and Knight (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements*. Final Report, Project 01-CTS-5, Water Environment Research Foundation (WERF): Alexandria, Virginia. Reprinted with permission.)

*represents the HSSF bed porosity* (ε *is typically between 0.3 and 0.4*).

Application of these equations is of limited utility, as in many instances the TSS entering the wetland is removed very rapidly, and the effluent TSS leaving the wetland is determined by internal biological processes, but not by the removal efficiency for incoming TSS, as indicated in Figure 7.26. However, if HSSF wetland detention times are small, settling may not be complete, and the *w-C** model (Equations 7.20 and 7.21) provides a reasonable description (Figure 7.27). As a result, the solids leaving a HSSF wetland are typically not related to the solids entering the system, but are produced by the decomposition and resuspension of biomat particulates within the HSSF bed.

### Seasonal and Stochastic Effects

Because HSSF wetland effluent TSS is a function of internal ecosystem processes, random physical and biological events result in short-term effects on effluent concentrations. Additionally, season and temperature are modifiers of the processes that generate and cycle solids. These effects typically follow a mild annual cycle with superimposed variability.

The trend may be determined most accurately if there are data spanning many annual cycles, which may then be "folded" into one multiyear display and averaged. TSS data time series often display some degree of sinusoidal behavior through the course of a calendar year, as described by Equation 7.22. An example of this is shown in Figure 7.28.

Examples of sinusoidal fitting of seasonal behavior in HSSF wetlands are summarized in Table 7.9 for three tertiary and four secondary treatment wetlands in England. The scatter of TSS data is large, and the seasonal trend typically accounts for a small percentage of the variability. For the tertiary HSSF wetlands listed in Table 7.9, effluent $R^2$-values range from 0.02 to 0.12, implying that 2%–12% of the effluent TSS variability could be attributed to seasonal effects. For the secondary systems listed in Table 7.9, effluent $R^2$-values range from 0.02 to 0.20. In general, these $R^2$-values are lower than for FWS wetlands.

HSSF wetlands typically display a peak TSS concentration in spring or summer ($t_{max}$ between 112 and 198 days for the Northern Hemisphere), as shown in Table 7.9, but these seasonal peaks account for only a small fraction of the effluent variability in TSS.

Because TSS concentrations in HSSF wetlands are only weakly seasonal, it is possible to ignore seasonal effects, and combine this into the overall variability of the system. This has been done in the treatment wetland literature (Wallace and Knight, 2006) for combined wetland data sets. Figure 7.29 shows an example of this approach.

Based on the probability distribution of effluent TSS concentrations, it is possible to determine a multiplier ($C_o/C_{median}$) associated with a given percentile of the effluent distribution.

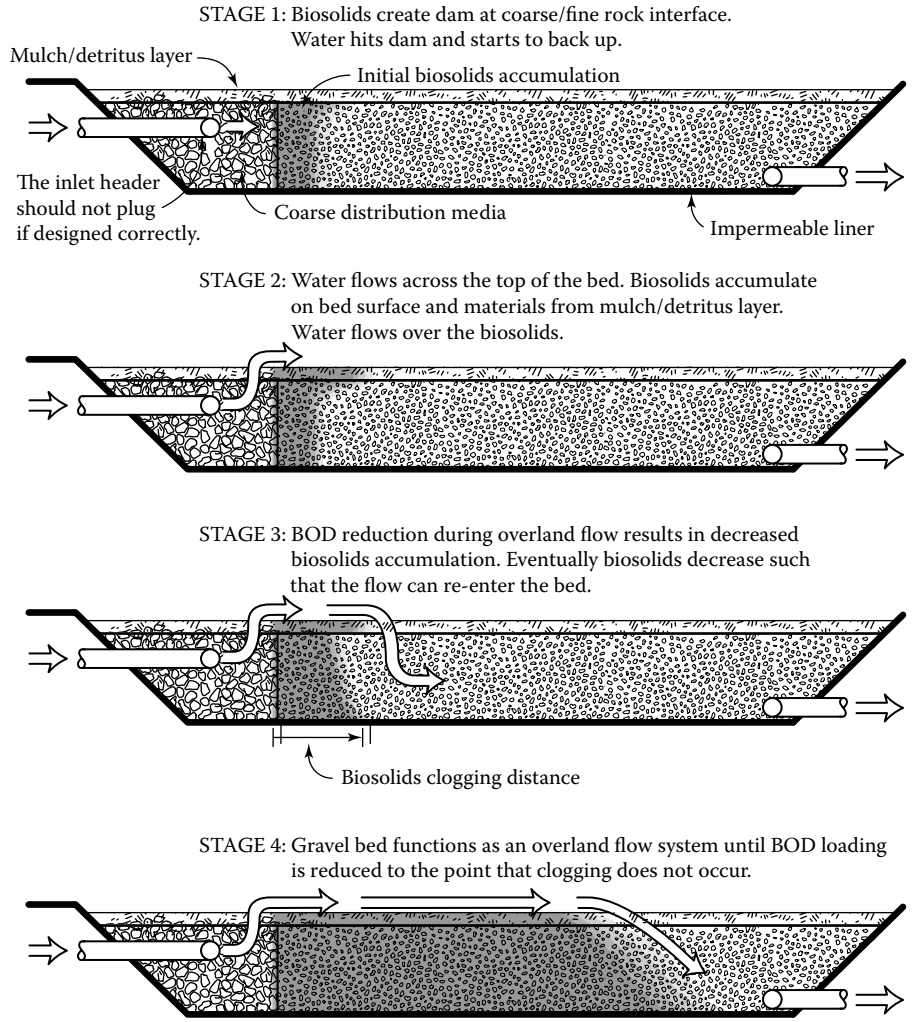



**FIGURE 7.25** Stages of bed clogging in HSSF wetlands. (Adapted from Wallace and Knight (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements*. Final Report, Project 01-CTS-5, Water Environment Research Foundation (WERF): Alexandria, Virginia; and D.J. Cooper *et al*. (2006b) *Factors affecting the longevity of subsurface horizontal flow systems operating as tertiary treatment for sewage effluent*: *Part II*. Kröpfelová (Ed.), Paper presented at the 6th International Workshop on Nutrient Cycling and Retention in Natural and Constructed Wetlands, 31 May–4 June 2006; Trebon, Czech Republic.)

These multipliers are often useful because they may well be involved in the permitting or licensing of a treatment wetland. Examples of these effluent multipliers are shown in Table 7.10 for seven HSSF wetlands (three tertiary and four secondary) in England. Lightly loaded (tertiary) HSSF wetlands often return a median effluent concentration close to detection limits. These systems display greater effluent variability than more heavily loaded systems with higher effluent TSS values. This greater variability can be attributed to stochastic effects, especially sampling error, that result in isolated high instances of effluent TSS. These instances still occur in secondary HSSF treatment wetlands, but the impact is not as great because these wetlands return, on average, a higher effluent TSS to begin with.

## Input–Output Relations

Suspended solids have been measured at inlets and outlets for a large number of HSSF wetlands. As the TSS leaving the wetland is a function of internal biological processes, there is a nonzero background concentration ($C^*$) for effluent TSS. HSSF data sets can be explored graphically; the most commonly used relationship is the outlet concentration ($C_o$) versus inlet loading graph (Figure 7.30).

As has been previously discussed, outlet concentrations ($C_o$) are generally not related to inlet concentration ($C_i$) (Figure 7.26). Also, outlet TSS concentrations have only minor seasonal variations (Figure 7.28). Therefore, there is little apparent relationship between TSS loading and effluent TSS concentrations. The data presented in Figure 7.30 indicate that HSSF wetlands return an average effluent concentration of 22.5 mg/L over a wide range of influent loadings, with a 90th percentile limit of 42 mg/L. This is broadly consistent with other statistical analyses of TSS removal in HSSF wetlands (Wallace and Knight, 2006). Performance criteria postulated by the U.S. Environmental Protection Agency (U.S. EPA, 2000a) also fall within the 50th and 90th percentile bands shown in Figure 7.30.

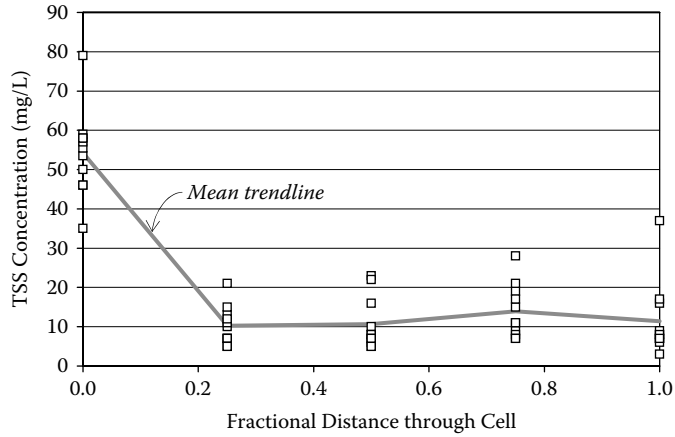



**FIGURE 7.26** Internal profile of TSS for the NERCC wetland near Duluth, Minnesota. Data represents eleven transects taken down the length of the HSSF wetland bed between May and October 1998. The hydraulic loading was small, 1.1–1.3 cm/d, and hence nominal detention times were about 14 days. Essentially all the inlet TSS is removed prior to the first sampling port (25% down the length of the bed). (From unpublished data.)

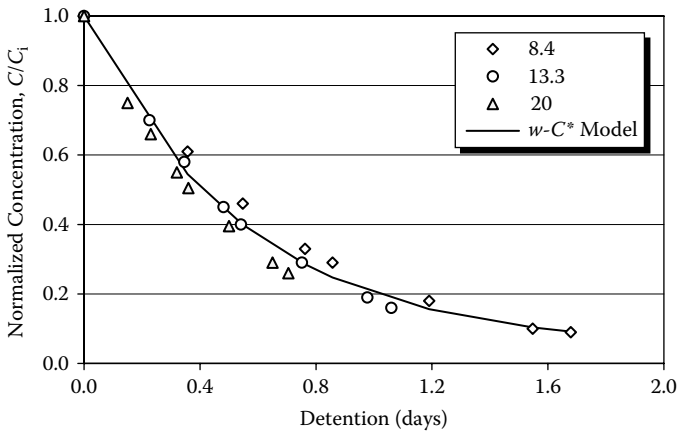



**FIGURE 7.27** Turbidity reduction for a HSSF wetland at Richmond, New South Wales, Australia, at different superficial velocities (8.4–20 m/d). The settling rate is $w = 0.29$ m/d, and the background is $C^*/C_i = 0.046$ ($R^2 = 0.987$). (Data from Sapkota and Bavor (1994) *Water Science and Technology* 29(4): 55–66.)

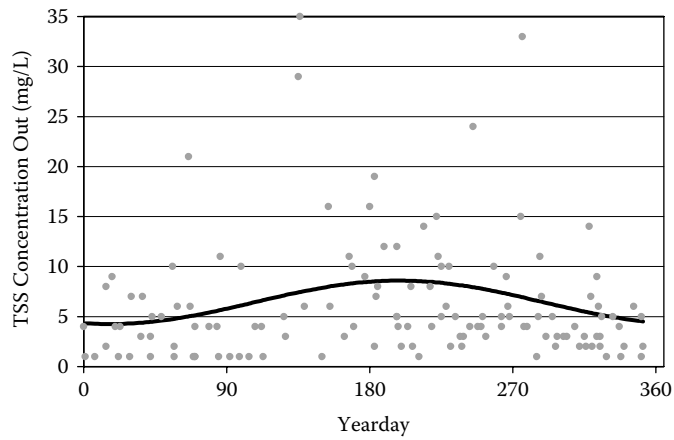



**FIGURE 7.28** TSS effluent concentrations for a HSSF treatment wetland in Staffordshire, England. Four years of data are represented; sampling frequency was every two weeks. (Data from CWA database (2006) *Constructed Wetlands Interactive Database*, *Version 9.02*. Compiled by G.D. Job and P.F. Cooper. United Kingdom Constructed Wetland Association (CWA): Gloucestershire, United Kingdom.)

**TABLE 7.9**
**Annual Trends for Influent and Effluent TSS for Seven HSSF Wetland Systems in England**

| System Location | Waste Stream | $C_{mean}$ (mg/L) | Amplitude, $A$ | $t_{max}$ (Julian day) | $R^2$ |
|---|---|---|---|---|---|
| **Tertiary Systems** | | | | | |
| Leicestershire | Influent | 19 | 0.26 | 361 | 0.07 |
| | Effluent | 4 | 0.15 | 338 | 0.02 |
| Solihull | Influent | 19 | 0.24 | 124 | 0.11 |
| | Effluent | 5 | 0.32 | 165 | 0.12 |
| Staffordshire | Influent | 41 | 0.23 | 237 | 0.04 |
| | Effluent | 6 | 0.34 | 198 | 0.02 |
| **Secondary Systems** | | | | | |
| Yorkshire | Influent | 162 | 0.10 | 115 | 0.09 |
| | Effluent | 24 | 0.13 | 202 | 0.06 |
| North Yorkshire 1 | Influent | 106 | 0.21 | 197 | 0.01 |
| | Effluent | 35 | 0.21 | 215 | 0.19 |
| North Yorkshire 2 | Influent | 144 | 0.04 | 261 | 0.01 |
| | Effluent | 32 | 0.35 | 119 | 0.20 |
| North Yorkshire 3 | Influent | 144 | 0.04 | 261 | — |
| | Effluent | 34 | 0.32 | 112 | 0.05 |

*Note:* Site names for U.K. systems are approximate. The trend in each time series is presumed to be sinusoidal (see text for notation). $C = C_{mean}\,(1 + A\cos[\omega\,(t - t_{max})]) + E$

*Source:* CWA database (2006) *Constructed Wetlands Interactive Database*, *Version 9.02*. Compiled by G.D. Job and P.F. Cooper. United Kingdom Constructed Wetland Association (CWA): Gloucestershire, United Kingdom.

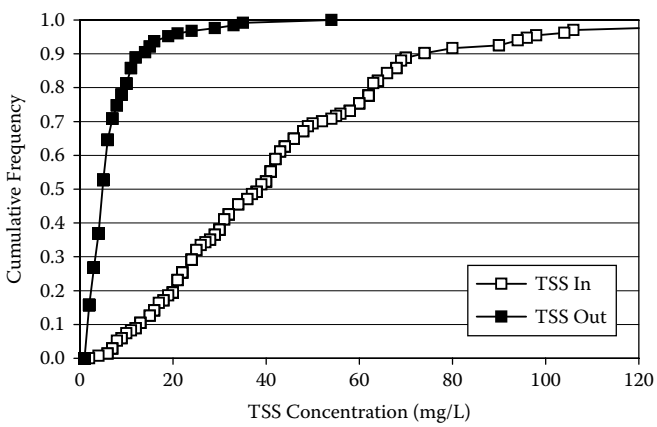



**FIGURE 7.29** Frequency percentiles for inlet and outlet TSS for a HSSF wetland in Staffordshire, England. The median inlet TSS was 38 mg/L; the median outlet TSS was 5 mg/L. Four years of data are represented; sample frequency was every two weeks. (Data from CWA database (2006) *Constructed Wetlands Interactive Database*, *Version 9.02*. Compiled by G.D. Job and P.F. Cooper. United Kingdom Constructed Wetland Association (CWA): Gloucestershire, United Kingdom.)

## 7.6 TSS REMOVAL IN VF WETLANDS

VF wetlands span a wide variety of operating regimes. To ensure adequate influent distribution, VF wetlands are often operated on a pulse-load (surface flooding) regime, tidal-flow (fill-and-drain) regime, or with networks of perforated distribution pipes. The fate, transport, and generation of TSS in vertical flow wetlands depends on several factors, including:

- Flow mode (continuous or intermittent; saturated or unsaturated)
- Inlet organic loading (particulate + soluble)
- Loading of inert (mineral TSS)
- Size of the wetland bed media

The wide range of flow modes and media sizes presents considerable difficulty when generalizing TSS fate, generation and transport regarding VF wetlands. Instead, this section focuses on the most common VF wetland mode, intermittent downflow beds. Other types of VF wetland systems are discussed in more detail in Part II of this book.

### Intermittent Downflow Beds

As discussed in Chapters 1 and 2, these types of VF wetlands are pulse-loaded, and operate on principles of unsaturated flow. Typically, these systems use a relatively fine, sand bed media ($d_{10} \sim 0.25$ mm) (Gesellschaft zur Förderung der Abwassertechnik d.V (GFA), 1998; Brix and Johansen,

**TABLE 7.10**
**Trend Multipliers for Effluent TSS in Tertiary (N = 3) and Secondary (N = 4) HSSF Wetlands in England**

| Location | Median Inlet (mg/L) | Median Outlet (mg/L) | Excursion Frequency 80% | 90% | 95% |
|---|---|---|---|---|---|
| Leicestershire | 14 | 4 | 1.30 | 2.00 | 2.50 |
| Solihull | 18 | 4 | 1.75 | 2.00 | 2.25 |
| Staffordshire | 38 | 4 | 2.25 | 3.10 | 4.45 |
| **Mean for Tertiary Systems** | | | **1.77** | **2.37** | **3.07** |
| Yorkshire | 154 | 22 | 1.35 | 1.81 | 2.39 |
| North Yorkshire 1 | 100 | 34 | 1.55 | 1.91 | 2.10 |
| North Yorkshire 2 | 144 | 29 | 1.50 | 1.78 | 2.05 |
| North Yorkshire 3 | 144 | 30 | 1.47 | 1.85 | 2.25 |
| **Mean for Secondary Systems** | | | **1.47** | **1.84** | **2.20** |

*Note:* Site names for U.K. systems are approximate.

*Source:* CWA database (2006) *Constructed Wetlands Interactive Database*, *Version 9.02*. Compiled by G.D. Job and P.F. Cooper. United Kingdom Constructed Wetland Association (CWA): Gloucestershire, United Kingdom. Trend multiplier is $(1 + \Psi)$; see Equation 6.61.

2006). With this bed media, deposition and filtration of particulates, especially near the surface of the bed, is a dominant removal mechanism of incoming TSS (Woodward and Ta, 1988; Platzer and Mauch, 1997).

In addition to the accumulation of incoming TSS at or near the surface of the wetland, a surficial biomat may develop on top of the VF bed in response to the overall organic loading (Figure 7.31). The mechanisms of biomat formation are essentially similar to those of biosolids accumulation encountered internally in HSSF wetlands. However, there is one important difference in the biomat formation in HSSF and VF wetlands. In HSSF wetlands, the loading is typically continuous, in a saturated flow environment. In VF wetlands, the loading is intermittent, allowing for "resting" periods of no biomat formation. For VF wetlands that are unsaturated during the resting phase, the high availability of atmospheric oxygen (21%) aids in aerobic decomposition of accumulated biomat material. Further,

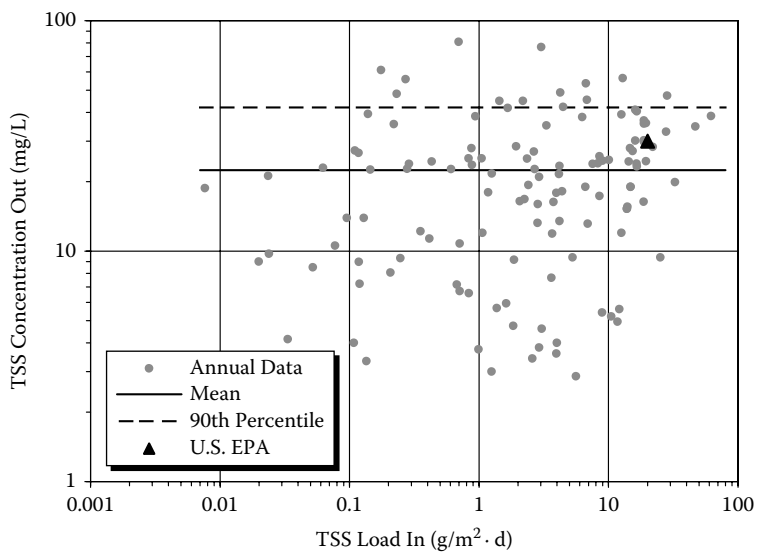



**FIGURE 7.30** TSS in the effluent of HSSF wetlands is not a function of inlet TSS loading. Chart represents 26 systems and 130 system-years of performance data from the CWA (2006) and WERF (2006) databases. The mean is 22.5 mg/L and the 90th percentile is 42 mg/L.



**FIGURE 7.31** Accumulated particulate organic matter on top of an intermittent downflow VF wetland bed in Roussillon, France.

the material is more accessible, because of its top location. Because of this configuration, VF wetlands are much more amenable to a "load and rest" operational regime to mitigate TSS clogging.

## Input–Output Relations

Removal of TSS is primarily through the mechanisms of filtration and interception, as is readily seen in Figure 7.31. This is broadly related to TSS loading, as indicated in Figure 7.32, load-in–concentration-out plot of intermittent downflow beds.

Vertical flow wetlands are highly effective in TSS removal, provided they are managed in a way to avoid bed clogging problems (typically accomplished through a load and rest operation regime). Data presented in Figure 7.32 represent 71 system-years of data from 31 vertical flow wetlands (intermittent downflow beds). The median inlet TSS concentration ($C_i$) was 90 mg/L; the median outlet TSS concentration ($C_o$) was 12 mg/L (87% concentration reduction).

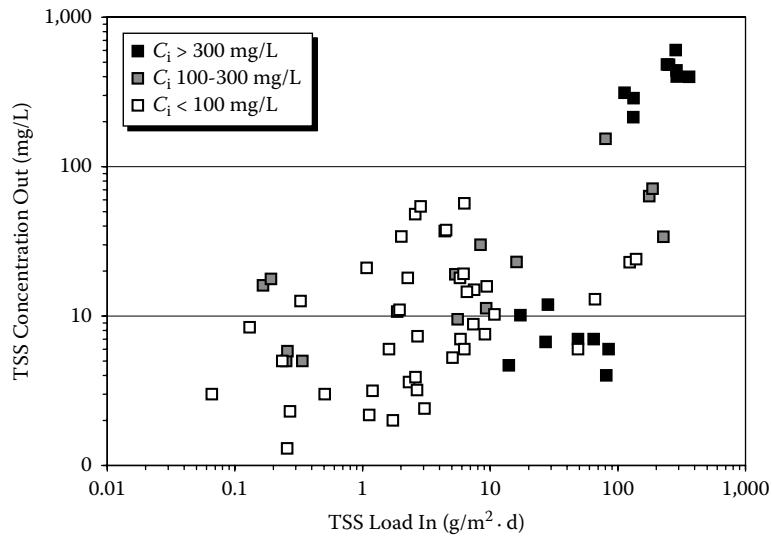



**FIGURE 7.32** Inlet TSS loading versus effluent TSS concentrations for VF wetlands. Data show 71 system-years of data from 31 intermittent downflow wetlands.

## SUMMARY

Treatment wetlands are consistently effective at reducing elevated concentrations of suspended solids. As most treatment wetlands are overdesigned in terms of TSS reduction, existing data are primarily useful for estimating background TSS outflow concentration variability, and are not helpful in estimating an area-based reaction rate constant for this parameter. If there is a need to specifically target TSS removal with more accuracy, procedures should rely on measured settling rates for the specific wastewater, while taking due caution to recognize the inevitable internal wetland processes that will result in irreducible background TSS concentrations and stochastic variability in response to factors outside the treatment wetland operator's control.

Suspended solids removal in FWS wetlands occurs through sedimentation, aggregation, and filtration/interception. FWS wetlands are generally very efficient in removing suspended solids, but foraging wildlife and wind- and wave-induced mixing can resuspend particulate matter. Phytoplankton production can also increase the concentration of suspended sediments in FWS wetlands. Suspended solids in HSSF and VF wetland systems generally do not exhibit particle resuspension because wind- and animal-induced mixing of the water column does not occur. Most suspended solids in HSSF and VF wetlands are removed through sedimentation and filtration or interception. High influent loading of suspended solids can lead to excessive biological growth which may clog sand or gravel media, causing head loss through the system that may lead to overland flow in HSSF wetlands and a complete failure of VF systems. Design approaches and operational responses to bed clogging in SSF wetlands are discussed in Part II of this book.

# 8 Carbon and Biochemical Oxygen Demand

Carbon compounds interact strongly with wetland ecosystems. The carbon cycle in wetlands is vigorous and typically provides carbon exports from the wetland to receiving ecosystems. Many internal wetland processes are fueled by carbon imports and by the carbon formed from decomposition processes.

Treatment wetlands frequently receive large external supplies of carbon in the added wastewater. Any of several measures of carbon content may be made, with biochemical oxygen demand (BOD) being the most frequent in the treatment of municipal wastewater. Degradable carbon compounds are rapidly utilized in wetland carbon processes. At the same time, a variety of wetland decomposition processes produce available carbon. The balance between uptake and production provides the carbon exports. In general, the amounts of carbon cycled in the wetland are comparable to the quantities added in domestic wastewater.

The growth of wetland plants requires carbon dioxide ($CO_2$) for photosynthesis. A variety of organisms release $CO_2$ as a product of respiration. Many pathways lead to the microbial production of $CO_2$, as well as methane ($CH_4$). Both gases dissolve in water to a limited extent; so there are active transfers of carbon to and from the atmosphere.

In terms of treatment, it is therefore not surprising to find good carbon reductions for the added wastewater, accompanied by nonzero background levels of various carbon compounds and the related BOD. For purposes of wetland design for BOD removal, the challenge is to find relatively simple design tools despite the enormously complex set of wetland functions.

## 8.1 WETLAND CARBON SPECIATION AND PROCESSING

A wide spectrum of carbon compounds exists in either dissolved or particulate forms in aquatic systems. The usual dividing line is a 0.45-μm filter. The following distinctions are made as a result of analytical methods:

- TC = total carbon (includes all dissolved and suspended forms)
- PC = particulate carbon (includes organic and inorganic forms)
- DC = dissolved carbon (includes organic and inorganic forms)
- IC = inorganic carbon (includes all dissolved and suspended forms)
- DIC = dissolved inorganic carbon (usually comprises $CO_2$, carbonate, and bicarbonate)
- TOC = total organic carbon (includes all dissolved and suspended forms)
- DOC = dissolved organic carbon
- NDOC = nondissolved organic carbon
- VOC = volatile organic carbon (compounds)

In soils or biomass, samples are subjected to combustion and dissolution, followed by analysis for total carbon.

### BOD, COD, AND TOC

Different analytical techniques are used to measure the amount of organic material in the wastewater. BOD is a measure of the oxygen consumption of microorganisms in the oxidation of organic matter. It is measured as the oxygen consumption in an airtight incubation of the sample. This test normally runs for five days, and the result is then more properly designated as $BOD_5$. Some oxygen may be used in nitrification if the necessary organisms are present in the sample. If this potential nitrogenous oxygen demand is inhibited chemically during the test, the result is carbonaceous biochemical oxygen demand ($CBOD_5$).

Chemical oxygen demand (COD) is the amount of a chemical oxidant, usually potassium dichromate, required to oxidize the organic matter. This measure is larger than BOD, because the strong oxidant attacks a larger group of compounds. However, nitrogenous compounds, such as ammonia, are not oxidized by the COD test. Oxygen or oxidant consumption may be measured before or after filtration, leading to measures of total and soluble BOD and COD. In the wetland environment, the presence of humic materials leads to COD values that are much larger than BOD values. In a northern peatland, the ratio was approximately 0.05 ($BOD_5$ = 5 mg/L:COD = 100 mg/L) (unpublished data from the Houghton Lake, Michigan, peatland). At Tres Rios, Arizona, wetlands treating nitrified secondary effluent, four wetlands gave ratios of 0.055 ± 0.004, averaged over seven years. In municipal wastewaters, the ratio is typically 0.4–0.8 (Metcalf and Eddy, Inc., 1991). Industrial wastewaters may have lower ratios.

Total organic carbon (TOC) is measured by chemical oxidation followed by analysis for $CO_2$. In a northern peatland, the ratio $BOD_5$:TOC was approximately 0.2 ($BOD_5$ = 5 mg/L:TOC = 25 mg/L) (unpublished data from the Houghton Lake peatland), and was 0.28 at Estevan, Saskatchewan. At Tres Rios wetlands treating nitrified secondary effluent, four wetlands gave ratios of $CBOD_5$:TOC = 0.25 ± 0.08, averaged over

**TABLE 8.1**
**Comparison of Oxygen Consumption Parameters for Various Waters**

| | $BOD_5$/COD | $CBOD_5$/COD | $BOD_5$/TOC | $CBOD_5$/TOC |
|---|---|---|---|---|
| **From Crites and Tchobanoglous** | | | | |
| Untreated wastewater | 0.3–0.8 | — | 1.2–2.0 | — |
| After primary settling | 0.4–0.6 | — | 0.8–1.2 | — |
| Final effluent | — | 0.1–0.3 | — | 0.2–0.5 |
| **From Metcalf and Eddy** | | | | |
| Untreated wastewater | 0.4–0.8 | — | 1.0–1.6 | — |
| **FWS Wetland Effluents** | | | | |
| Columbia, Missouri | 0.21–0.23 | 0.11–0.13 | — | — |
| Tres Rios, Arizona | 0.05–0.06 | — | — | 0.17–0.33 |
| Estevan, Saskatchewan | — | — | 0.28 | — |
| Houghton Lake, Michigan | 0.05 | — | 0.2 | — |
| Orlando Easterly, Florida | — | — | 0.09–0.13 | — |

*Source:* WWTP values from Crites and Tchobanoglous (1998) *Small and Decentralized Wastewater Management Systems.* McGraw-Hill, New York; Metcalf and Eddy Inc. (1991) *Wastewater Engineering, Treatment, Disposal, and Reuse.* Tchobanoglous and Burton (Eds.), Third Edition, McGraw-Hill, New York.

seven years. In municipal wastewaters, the ratio is 1.0:1.6 (Metcalf and Eddy Inc., 1991).

The interrelation among the various measures of carbon and oxygen demand are given in Table 8.1. The interpretation of these ratios is that natural wetlands cycle at low levels of biologically usable carbon compounds, whereas municipal wastewaters are rich in usable carbon compounds.

Wetlands are efficient users of external carbon sources, manifested by excellent reductions in $BOD_5$ and COD. However, wetlands possess nonzero background levels of both BOD and COD, which depend on the type and status of the wetland. Typical ranges for background concentrations are 1–10 mg/L for $BOD_5$ and 10–100 mg/L for COD.

## Wetland Chemistry of Carbon

### Dissolved Inorganic Carbon

Of the hundreds of carbon compounds that may occur in the wetland environment, relatively few are inorganic. Dissolved inorganic carbon consists primarily of $CO_2$, carbonate, and bicarbonate.

In pure water solution, the principal carbonate species are related to atmospheric $CO_2$ by the temperature and pH-dependent dissolution and dissociation series:
Henry's Law:

$$H_2CO_3^* \rightleftharpoons H_2O + CO_{2(g)} \qquad K_H = \frac{P_{CO_2}}{[H_2CO_3^*]} \tag{8.1}$$

where

$$[H_2CO_3^*] = [H_2CO_3] + [CO_2] \tag{8.2}$$

Hydration:

$$H_2CO_3 \rightleftharpoons H_2O + CO_2 \qquad K = \frac{[CO_2]}{[H_2CO_3]} \tag{8.3}$$

First Dissociation:

$$H_2CO_3 \rightleftharpoons HCO_3^- + H^+ \qquad K_{H_2CO_3} = \frac{[HCO_3^-][H^+]}{[H_2CO_3]} \tag{8.4}$$

Second Dissociation:

$$HCO_3^- \rightleftharpoons CO_3^{2-} + H^+ \qquad K_2 = \frac{\left[CO_3^{2-}\right][H^+]}{[HCO_3^-]} \tag{8.5}$$

and where, as a result of Equation 8.2,

$$K_1 = \frac{K_{H_2CO_3}}{K+1} \tag{8.6}$$

the notation of Pankow (1991) has been adopted. Brackets indicate the concentration of the chemical species, in molarity; and all are in water except for atmospheric $CO_2$. The value of the equilibrium constant $K \approx 650$, and hence most of the dissolved carbon is present as $CO_2$. Equations 8.1–8.6 may be solved for concentrations, given the partial pressure of $CO_2$ and the various equilibrium constants.

$$[H_2CO_3^*] = K_H P_{CO_2} \tag{8.7}$$

$$[HCO_3^-] = \frac{K_1}{[H^+]} K_H P_{CO_2} \tag{8.8}$$

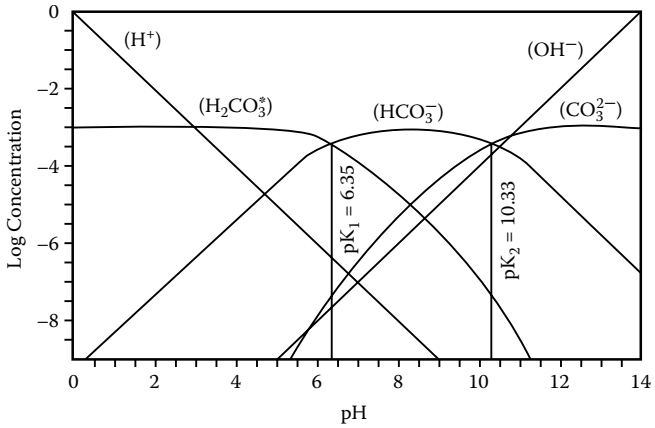



**FIGURE 8.1** Distribution of carbonate species in water at 25°C. The partial pressure of $CO_2$ in the air is taken as $3.16 \times 10^{-4}$ atm. (From Metcalf and Eddy Inc. (1998) *Wastewater Engineering, Treatment, Disposal, and Reuse,* Tchobanoglous *et al.* (Eds.), Fourth Edition, McGraw-Hill, New York. Reprinted with permission.)

$$\left[CO_3^{2-}\right] = \frac{K_1 K_2}{[H^+]^2} K_H P_{CO_2} \tag{8.9}$$

The equilibrium constants, and hence the various concentrations, are all pH- and temperature-dependent. These forms are distributed in water at 25°C as shown in Figure 8.1 (Pankow, 1991). However, it must be noted that wetland waters are more complex than the pure water system and therefore will not follow such idealized chemistry precisely. Modifications of the calculation (APHA, 1992) deal with expected deviations due to dissolved solids, but not the full suite of biological variations that may be expected in wetlands. Production and consumption of $CO_2$ in the wetland may significantly alter the chemical balance in the water.

An important feature of the carbonate system is its influence on pH under mediation by algae. Algal consumption of $CO_2$ drives pH upward, and may give rise to $9 < pH < 10$ in unshaded wetland environments or ponds.

### Precipitates

A variety of cations can precipitate carbonates under certain conditions. The most important is calcium carbonate, $CaCO_3$. A major process in periphyton-dominated wetlands is chemical precipitation of $CaCO_3$ under conditions of high pH created by the algae (Gleason, 1972). Similarly, in beds of submerged aquatic vegetation, $CO_2$ and bicarbonate are consumed during photosynthesis, thereby raising the water column pH and promoting $CaCO_3$ precipitation (Dierberg *et al*., 2002).

A variety of cations can precipitate carbonate under certain conditions. Some important mineral precipitates in the wetland environment are:

| | |
|---|---|
| Calcite: | $CaCO_3$ |
| Aragonite: | $CaCO_3$ |
| Magnesite: | $MgCO_3$ |
| Dolomite: | $CaMg(CO_3)_2$ |

Calcium carbonate saturation indices may be calculated in a number of ways (APHA, 1992). However, overall carbon mineral chemistry is very complex; consequently, accurate calculations of solubilities are generally not possible, especially in wetland environments.

## Organic Carbon

### Biomass: Growth, Death, Decomposition

The wetland cycle of growth, death, and partial decomposition uses atmospheric carbon, and produces gases, dissolved organics, and solids (Figure 8.2). Decomposition involves the sugars, starches, and low molecular weight celluloses in the dead plant material. Gaseous products include methane and regenerated $CO_2$. A spectrum of soluble large organic molecules, collectively termed *humic substances*, are released into the water. The solid residual of plant decomposition is peat or organic sediment, which originated as celluloses and lignins in the plants. These wetland soil organics are broadly classified as fulvic material, humic material, and humin, based upon whether they are acid soluble, base soluble, or insoluble (NRCC, 1979).

The sediments, soils, and biomass in a wetland contain major proportions of carbon. The carbon content of 28 species of wetland plants has been reported by Boyd (1978) as 41.1% ± 0.7% (dry weight, mean ± SE). *Typha latifolia* values from 30 sites ranged from 43.3% to 47.2% (Boyd and Hess, 1970). Reddy *et al.* (1991) reported 44.0% ± 2.5% for peats in the upper 30 cm of the soil column. Soil scientists sometimes use a concentration of 58% for the carbon content of soil organic matter (the Van Bemmelen factor; Collins and Kuehl, 2001). Thus nearly half of the dry wetland plant and soil material is carbon.

The internal wetland carbon cycle is large. A general idea of the magnitudes of the various carbon transfers in a northern treatment marsh may be gained from considering the annual growth and decomposition patterns (see Chapter 3). A eutrophic treatment marsh grows about 3,000 dry g/m$^2$ of aboveground biomass each year, with a carbon content of about 43%. This translates to an annual average requirement for 35 kg/ha·d of carbon. In northern climates, this requirement is utilized during a growing season of approximately four months. In the case of emergent macrophytes, some of this carbon may be withdrawn from the atmosphere. However, submerged vegetation draws carbon from the aquatic carbonate system.

Decomposition of the resultant litter returns a significant portion of that carbon to the atmosphere and to wetland waters, but in treatment wetlands, a small fraction, on the order of 15% or 20%, is stored in accreted soil and sediments. That storage (burial) fraction therefore amounts to about 5 kg/ha·d as an annual average for the eutrophic marsh example. The balance, about 30 kg/ha·d, is processed via one or more mechanisms involving a variety of electron acceptors (oxidants), or via anaerobic digestion which generates methane.

The oxygen consumed by aerobic decomposition of sediments and litter is termed the sediment oxygen demand (SOD). In stream environments with large wastewater influences, the rate of consumption of oxygen by the stream sediments may be estimated as 20–100 kg/ha·d (Metcalf and

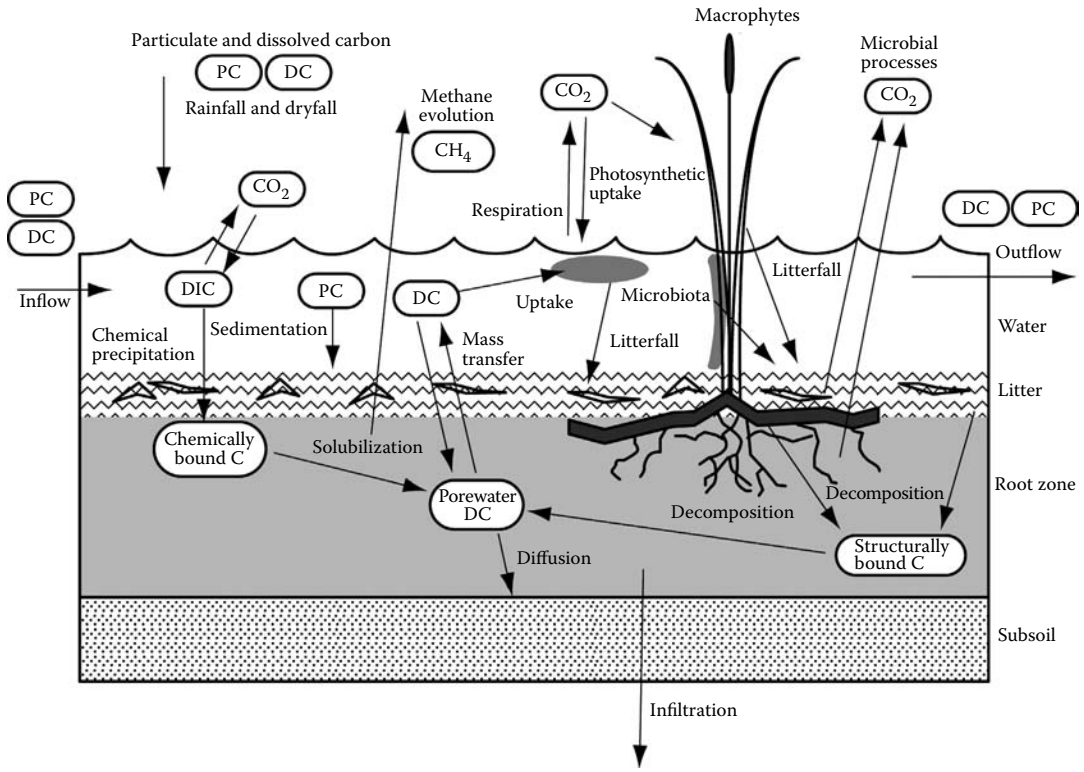



**FIGURE 8.2** Carbon storages and transfers in the wetland environment. DC = dissolved carbon; PC = particulate carbon; DIC = dissolved inorganic carbon; DOC = dissolved organic carbon; $CH_4$ = methane; $CO_2$ = carbon dioxide. Biomass carbon consists of living and dead biomass, as well as organic decomposition products. (From Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

Eddy Inc., 1991). In the eutrophic marsh example, if all the decomposition were to proceed via oxidation with dissolved oxygen as the electron acceptor, and $CO_2$ as the product, the equivalent SOD loading would be $(32/12) \times 30 = 80$ kg/ha·d. As will be subsequently shown, this potential SOD loading is at the upper end of the range of external BOD loadings (BLI) for treatment wetlands.

The wetland environment is more complicated than the stream environment. Some of the carbon is processed above-water, as standing dead material oxidizes. Some of the submerged sediments and litter are processed into soluble organic compounds that contribute to CBOD in the water, thus creating a nonzero background CBOD in a wetland environment. Starches, sugars, and cellulose are degraded to amino acids and fatty acids (Reddy and Graetz, 1988). In addition to dissolved oxygen, a variety of electron acceptors may be involved in decomposition.

## Carbon Processing in Wetland Necromass and Soils

A rough representation of the various decomposition "reactions" may be set down (Mitsch and Gosselink, 1993). These occur in different horizons in the wetland, as indicated in Figure 8.3.

*Respiration* occurs in aerobic zones:

$$\underset{\text{carbohydrates}}{C_6H_{12}O_6} + 6\,O_2 \rightarrow 6\,CO_2 + 6\,H_2O \tag{8.10}$$

*Fermentation* occurs in anoxic or anaerobic zones:

$$\underset{\text{carbohydrates}}{C_6H_{12}O_6} \rightarrow 2\,\underset{\text{lactic acid}}{CH_3CHOHCOOH} \tag{8.11}$$

$$\underset{\text{carbohydrates}}{C_6H_{12}O_6} \rightarrow 2\,\underset{\text{ethanol}}{CH_3CH_2OH} + 2\,CO_2 \tag{8.12}$$

*Nitrate Reduction* (denitrification) occurs in anoxic or anaerobic zones:

$$\underset{\text{carbohydrates}}{C_6H_{12}O_6} + 4\,NO_3^- \rightarrow 6\,CO_2 + 6\,H_2O + 2\,N_2 + 4\,e^- \tag{8.13}$$

*Iron Reduction* occurs in anoxic or anaerobic zones:

$$\underset{\text{acetate}}{CH_3COO^-} + 8\,Fe^{3+} + 3\,H_2O \rightarrow 8\,Fe^{2+} + CO_2 + HCO_3^- + 2\,H_2O + 8\,H^+ \tag{8.14}$$

*Sulfate Reduction* occurs in anaerobic zones:

$$2\,\underset{\text{lactate}}{CH_3CHOHCOO^-} + SO_4^{2-} + H^+ \rightarrow 2\,\underset{\text{acetate}}{CH_3COO^-} + 2\,CO_2 + 2\,H_2O + HS^- \tag{8.15}$$

$$\underset{\text{acetate}}{CH_3COO^-} + SO_4^{2-} + 2\,H^+ \rightarrow 2\,CO_2 + 2\,H_2O + HS^- \tag{8.16}$$

*Methanogenesis* occurs in anaerobic zones:

$$4H_2 + CO_2 \rightarrow CH_4 + 2\,H_2O \tag{8.17}$$

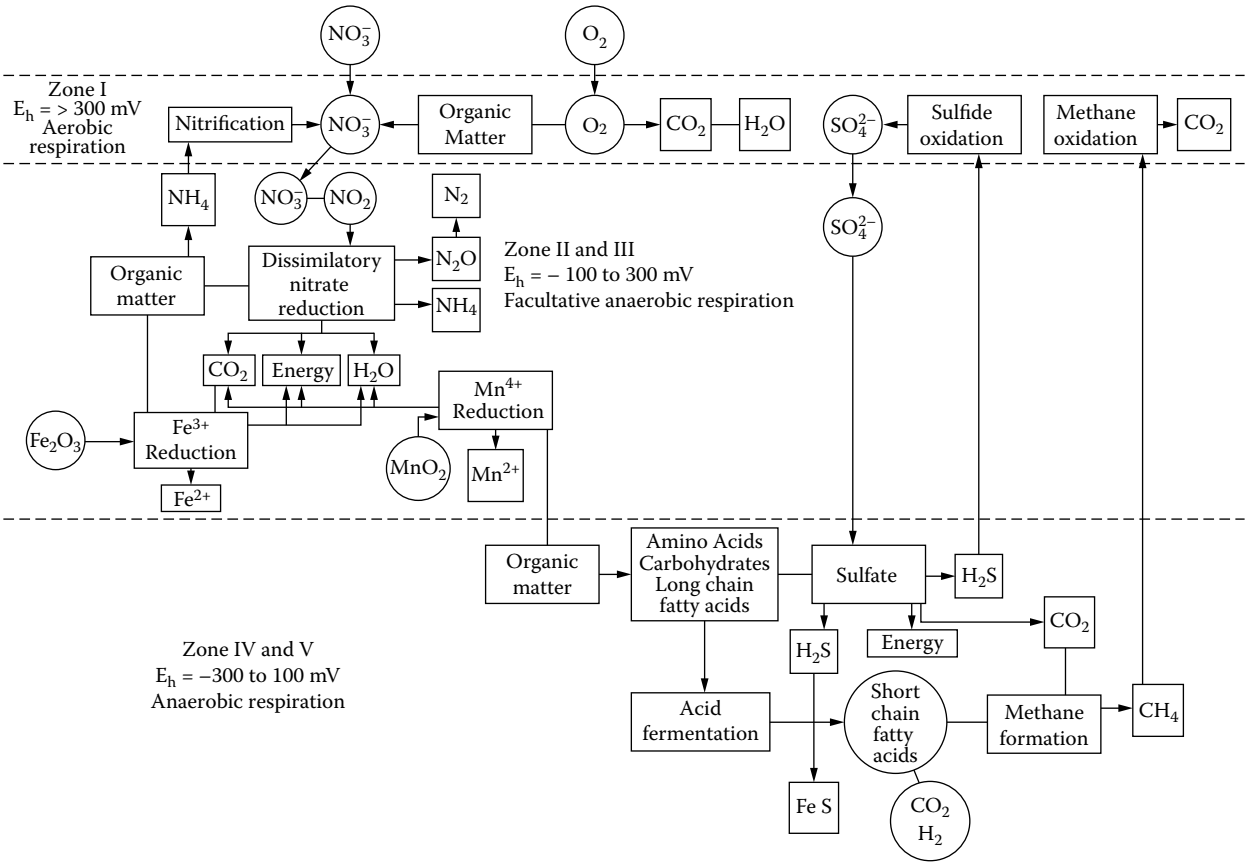



**FIGURE 8.3** Pathways of organic carbon decomposition in wetland soils. Aerobic, facultative anaerobic, and obligate anaerobic processes are all typically present at different depths in the soil. (From Reddy and Graetz (1988) In *The Ecology and Management of Wetlands*. Hook (Ed.), Croom Helm, London, United Kingdom, pp. 307–318. Reprinted with permission.)

$$\underset{acetate}{CH_3COO^-} + 4\,H_2 \rightarrow 2\,CH_4 + H_2O + OH^- \qquad (8.18)$$

The relative percentages of these reactions were investigated in controlled SSF wetland microcosms by Burgoon (1993), using acetate as the carbon source. His results demonstrated that all routes can be important, depending upon physical and chemical conditions.

It is apparent that the wetland provides a spectrum of potential pathways for the utilization of organic carbon compounds. Sufficient information is not available to quantify both the complex chemistry and the spatial distribution of chemical compounds. Therefore, the interactions must be described via correlations and rate equations, which are supportable by wetland performance data.

## 8.2 BOD REMOVAL IN FWS WETLANDS

A large amount of BOD data now exists for FWS wetlands treating a variety of wastewaters. There are a number of ways to summarize this information, including removal rate models and graphical summaries. When waters with moderate to large concentrations of BOD flow through a wetland, a decrease in concentration to a nonzero plateau is typically observed. This behavior is illustrated in Figure 8.4 for one of the continuous flow Sacramento, California, wetlands (Nolte and Associates, 1997). Samples were taken along the wetland

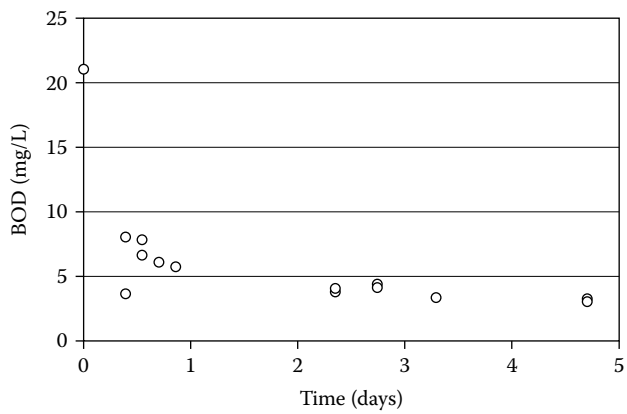



**FIGURE 8.4** Profiles of BOD concentration in Cell 7B of the Sacramento, California, treatment wetlands on May 3 and May 4, 1995. The plateau is at 3.1 mg/L. (Data from Nolte and Associates (1997) *Sacramento Regional Wastewater Treatment Plant Demonstration Wetlands Project*. 1996 Annual Report to Sacramento Regional County Sanitation District, Nolte and Associates: Sacramento, California.)

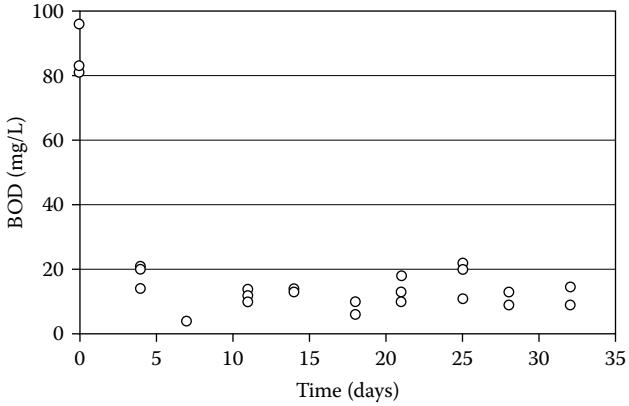



**FIGURE 8.5** The progression of BOD concentrations in three wetlands operated in the batch mode. The plateau is at 11.3 mg/L (Data from Lakhsman (1981) *A Demonstration Project at Humboldt to Provide Tertiary Treatment to the Municipal Effluent Using Aquatic Plants*. SRC Publication No. E-820-4-E-81. 74 pp. Saskatchewan Research Council.)

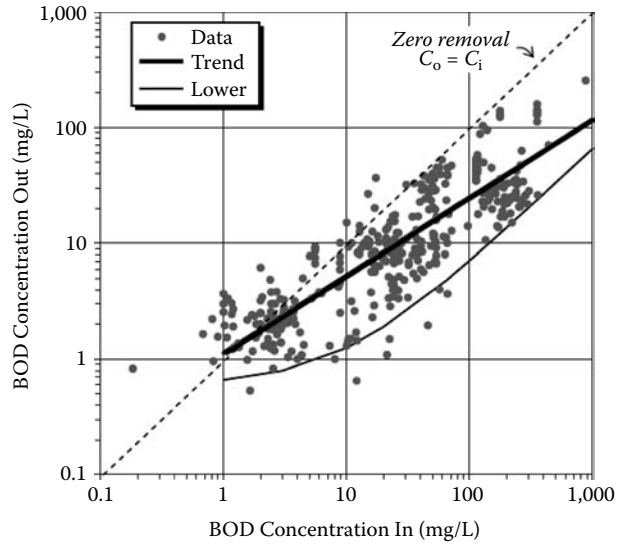



**FIGURE 8.6** Input–output concentration for BOD in FWS wetlands. Each point represents an annual average for one wetland. There are 385 wetland·years of data for 138 wetlands. The trend line is $y = 1.13\,x^{0.67}$ ($R^2 = 0.75$ logarithmic). The lower bound line is $y = 0.6 + 0.065\,x$, and includes 98% of the annual averages.

length, at positions corresponding to increasing nominal detention time. The same sort of response is seen in the results of Lakhsman (1981) for batch wetland treatment of lagoon effluents. A set of wetlands were charged with wastewater, then closed in, with no water additions or withdrawals. Typical response data showed a sharp decrease in $BOD_5$ to a nonzero, fluctuating background (Figure 8.5). The decrease is steep—perhaps exponential—but to a nonzero background $BOD_5$.

### Annual Input–Output Concentration Relations

The concentration of carbonaceous compounds is reduced in FWS wetlands for incoming concentrations above background. If, however, incoming BOD is below background, concentrations may increase upon passage through the system. As inlet concentrations increase, outlet concentrations increase, in a log-linear progression (Figure 8.6). There is considerable intersystem variability, but the data exhibit a lower bound, which may be interpreted as the lowest background concentration corresponding to a given inlet concentration. This curve is approximated by

$$C^{**} = 0.6 + 0.065\,C_i \tag{8.19}$$

where

$C_i$ = inlet BOD concentration, mg/L
$C^{**}$ = lower limit background BOD concentration, mg/L

Depending on hydraulic conditions, and the character of the incoming BOD, individual wetlands will typically exhibit different $C^*$-values as model calibration parameters, which may be larger than $C^{**}$.

### First-Order Modeling

The *P-k-C** first-order model can readily account for observations, for appropriate values of parameters (see Chapter 6). However, parameter values are known to depend on system hydraulics (Kadlec, 2000), as well as on speciation of the BOD (Crites and Tchobanoglous, 1998; Kadlec, 2003a).

BOD and COD are water quality parameters measured by procedures that lump individual chemical compounds into an overall, or total, concentration for that class of materials. It is clear that the individual components of such mixtures may be degraded or removed at different rates, and that there is a corresponding difference in removal rate constants (Crites and Tchobanoglous, 1998; Tchobanoglous *et al*., 2000; Kadlec, 2003a). There is therefore a distribution of rate constants across the various mass fractions of the mixture. As water containing such a mixture passes through the wetland, its composition changes because different fractions of the mixture are reduced at different rates. The mixture becomes *weathered*, a term coined to describe the selective stripping of light volatile materials upon exposure to outdoor environments. In the case of BOD and COD, the easy-to-degrade substances are lost first; more recalcitrant compounds persist for longer times.

The BOD test itself reflects only a fraction of the carbonaceous mixture, because it is terminated before all components are oxidized. For municipal wastewater, the five-day BOD test typically measures about two thirds of the ultimate BOD (UOD) (Metcalf and Eddy, Inc., 1991; Crites and Tchobanoglous, 1998).

### Effects of Lumping on Removal Models

The potential effects of speciation in lumped contaminant measures, particularly BOD, as manifested in changing rates, have been known for several years (Tchobanoglous, 1969; Crites and Tchobanoglous, 1998; Shepherd *et al*., 2001).

Crites and Tchobanoglous (1998) set forth a formulation for a "retarded rate expression." However, Kadlec (2003a) demonstrated that this concept was subsumed by a relaxed tanks-in-series (TIS) model. The *P-k-C** model is here defined to be (see Chapter 6):

$$\frac{C_o - C^*}{C_i - C^*} = \frac{1}{(1 + k/Pq)^P} \tag{8.20}$$

where

$C_i$ = inlet BOD concentration, mg/L
$C_o$ = outlet BOD concentration, mg/L
$C^*$ = background BOD concentration, mg/L
$k$ = apparent TIS rate constant, m/yr
$P$ = apparent number of TIS for BOD reduction
$q$ = hydraulic loading rate, m/yr

The parameter $P$ accounts for two effects: the detention time distribution (DTD) and the $k$-value distribution ($k$VD) (see Chapter 6). The value of $P$ is always less than the number of tanks determined from a tracer test. For broad distributions of $k$-values, such as may occur for BOD, a hydraulic TIS number of four (see Table 6.3) will be reduced to a $P$-value of one or two. However, the $C^*$-value in Equation 8.20 reflects several possible different causes. There may be a real irreducible component of BOD (hard to imagine, because it all disappears in the lab test), or there may be wetland ecosystem feedback of BOD constituents. But in addition, DTDs and $k$VDs may create an apparent $C^*$ as an artifact of model parameter fitting. These may be considered "bypassing $C^*$" and "weathering $C^*$", respectively.

Reasonable data fits may be obtained for specific wetlands or specific sites. Seven Gustine, California, wetlands were operated at different hydraulic loadings (different detention times) for a calendar year (Walker and Walker, 1990). The *P-k-C** model parameters determined from that input–output data were: $P = 1$, $k = 63$ m/yr, and $C^* = 9.7$ mg/L ($R^2 = 0.60$). Those parameters also provided a reasonable fit to transect data (Figure 8.7, $R^2 = 0.59$). However, it is uncommon to have multiple wetlands and multiple loadings from which to derive these types of calibrations.

## Concentration Profiles and Modeling Pitfalls

Difficulties with the *P-k-C** first-order model are compounded by the problem that data sets are very often poorly conditioned to produce good estimates of both $k$ and $C^*$ by any of the several methods of parameter estimation. This is easily visualized from Figures 8.4, 8.5, and 8.8, which contain examples of the early exponential decline (governed by $k$), together with the late plateau (governed by $C^*$). There are insufficient data in the exponential region for Sacramento and Humboldt to get a good estimate of $k$, but plenty of data to define $C^*$. Conversely, the Arcata pilot, Benton, and Gustine data sets never reach a plateau; all the data is concentrated in the exponential decline region. Thus, for these wetlands, transect data will provide a good estimate of $k$, but a very poor estimate of $C^*$. Input–output data for these sites may nonetheless be fitted to the model. In addition to the Gustine results given above, Benton input–output data over a two-year span resulted in $P = 1$, $k = 260$ m/yr, and $C^* = 5$ mg/L. At the Arcata pilot, input–output data over a two-year span resulted in $P = 1$, $k = 53$ m/yr, and $C^* = 4$ mg/L.

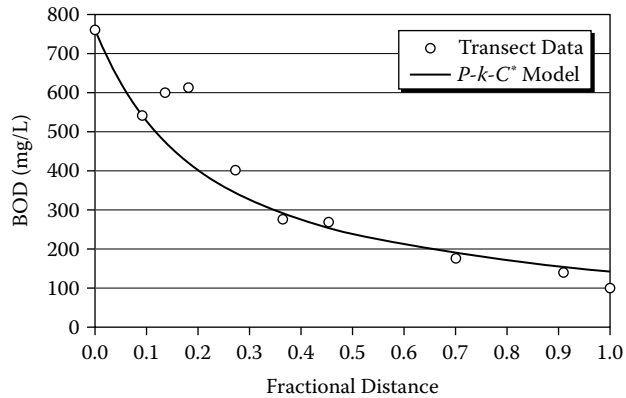



**FIGURE 8.7** BOD profile in the flow direction for wetland 1D at Gustine, California. The model curve was derived from independent input–output data for seven wetlands over a calendar year. (From Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

It is tempting to arbitrarily pick some low concentration to represent $C^*$, but that is counter-indicated by the importance of $C^*$ in wetland sizing, as shall be seen in the following sections. There is not an existing method to make such an estimate with confidence. One need look no further than data from two wetlands in the same geographical region: Humboldt, Saskatchewan, shows $C^* = 11.3$, but not far away, Oak Hammock, Manitoba, shows $C^* = 2.4$. Both are batch systems treating domestic lagoon effluent. We shall also see that $k$-values are widely variable, both across years for one wetland (interannual variability) and across wetlands (intersystem variability). Thus, to the dismay of researchers seeking to do THE definitive design model calibration study, no such study can be trusted in and of itself.

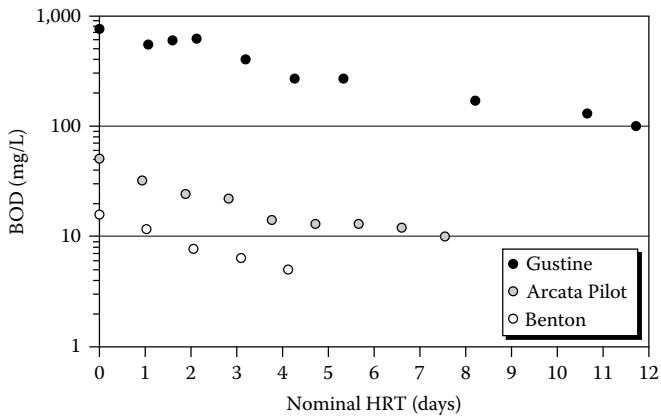



**FIGURE 8.8** Initial exponential declines in BOD for FWS wetlands. These systems did not achieve any apparent plateau.

### Distribution of *k*-Values

It is instructive to examine multiple data sets that provide a distribution of *k*-values and *C**-values. If all data are considered together, the inter- and intrasystem effects are compounded by a shift in the probable mechanisms of BOD reduction, as detailed in Equations 8.10–8.18. As loadings increase, aerobic processes become less of a probable factor, and are replaced by anoxic processes. Therefore, four levels of inlet concentration are considered: tertiary ($0 < C_i < 30$ mg/L); secondary ($30 < C_i < 100$ mg/L); primary ($100 < C_i < 200$ mg/L); and "super" ($C_i > 200$ mg/L). The effect of BOD weathering, which produces lower *k*-values as reaction proceeds, is quite strong for BOD. Data fits are better for *P*-values that are considerably lower than the tracer-determined number of tanks-in-series (*N*TIS) values. In general, data fits are best at $P = 1$, as noted earlier for Gustine, Benton, and Arcata. If the annual performance database is used for calibration, a value of *P* somewhat less than 1 is found, and therefore analysis has been performed using $P = 1$. For purposes of uniformity, the presumptive *C**-values are taken to be those of Equation 8.20, leading to $C^* = 2, 5, 10$, and 20 mg/L for the four categories, respectively.

The resultant annual average *k*-values are given in Table 8.2. The median values are not much different for tertiary, secondary, and primary applications (median = 37 ± 4 m/yr), but increases for the stronger influents (super) to 189 m/yr. The spread of these distributions is quite large, implying that the characteristics of individual wetlands, or individual years in the period of record, can have strong influences on performance.

### Annual Loading Relations

The BOD concentration produced in treatment wetland depends upon three primary variables (area, water flow, and inlet concentration), as well as numerous secondary variables (vegetation type, internal hydraulics, depth, event patterns, and others). It is presumed that the area effect may be combined with flow as the hydraulic loading rate (flow per unit area), because two side-by-side wetlands with double the flow should produce the same result as one at nominal flow. Therefore, two primary variables are often considered: hydraulic loading rate ($q$ = HLR) and inlet concentration ($C_i$). Previous performance analyses have been based upon these two variables (Kadlec and Knight, 1996).

An equivalent approach is to rearrange the primary variables, without loss of generality, by using BLI rate ($q \cdot C_i$) and concentration ($C_i$). Thus it is expected that the outlet concentration produced ($C_o$) will depend upon BLI and $C_i$. A graphical display has often been adopted in the literature (Kadlec and Knight, 1996; U.S. EPA, 2000a; Wallace and Knight, 2006). In the broad context, multiple data sets are represented by a trend that shows decreasing $C_o$ with decreasing BLI (Figure 8.9). Scatter is presumably due to secondary variable differences, such as the relative proportions of different vegetation types, hydraulic efficiencies, and other factors. The points at lowest loadings are for systems receiving very low BOD.

Each point in Figure 8.9 represents the average of one year's data for a given FWS wetland. Both BOD and CBOD data are represented; therefore, it is understood that some of the scatter is due to the difference between these two measures. The use of annual averages removes seasonal variability, if any, and precludes the effects of synoptic error (see Chapter 6).

### Model Curves

The data cloud in Figure 8.9 has been reproduced in Figure 8.10, together with the *P-k-C** model results for various parameter values. The hydraulic loading is also an

**TABLE 8.2**
**Distribution of Annual Areal Rate Coefficients $k_A$ (m/yr) for BOD in FWS Wetlands**

| | Tertiary | Secondary | Primary | Super |
|---|---|---|---|---|
| $C_i$ (mg/L) | 0–30 | 30–100 | 100–200 | >200 |
| $C^*$ (mg/L) | 2 | 5 | 10 | 20 |
| N | 203 | 77 | 63 | 43 |
| **Percentile** | | | | |
| 0.05 | 2 | 2 | 9 | 24 |
| 0.1 | 7 | 4 | 12 | 26 |
| 0.2 | 13 | 11 | 19 | 35 |
| 0.3 | 16 | 16 | 23 | 54 |
| 0.4 | 22 | 30 | 31 | 130 |
| 0.5 | 33 | 41 | 36 | 189 |
| 0.6 | 62 | 49 | 48 | 271 |
| 0.7 | 79 | 67 | 112 | 439 |
| 0.8 | 175 | 103 | 217 | 576 |
| 0.9 | 195 | 295 | 411 | 827 |

*Source:* The *C**-values range according to Equation 8.20, as indicated, and the value of $P = 1$.

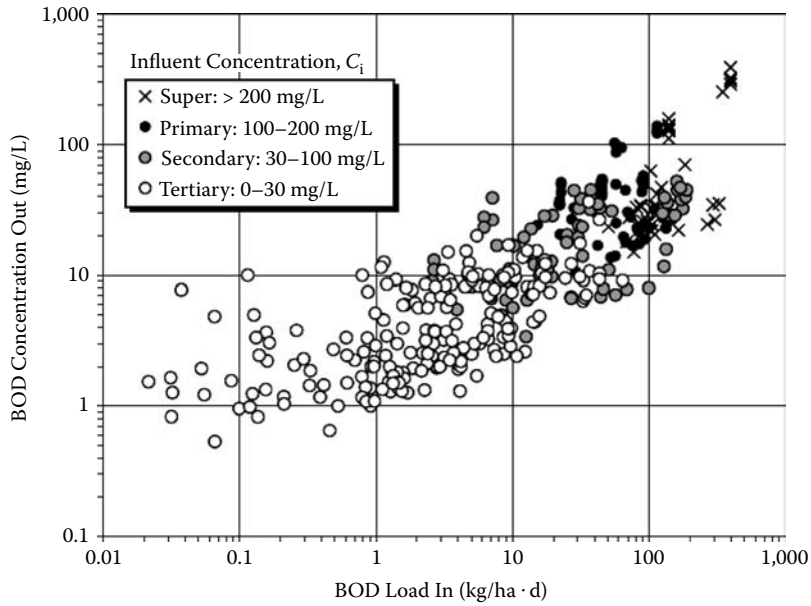



**FIGURE 8.9** Outlet BOD concentration versus BOD loading for FWS wetlands. Each of the 383 points represents an annual average for one of 136 wetlands. Data groups are for tertiary ($0 < C_i < 30$ mg/L); secondary ($30 < C_i < 100$ mg/L); primary ($100 < C_i < 200$ mg/L); and "super" ($C_i > 200$ mg/L).

independent parameter in that model. It is seen that the data are bounded by Line 1, which represents high $C^*$ and low HLR and $k$; and Line 2, which conversely represents low $C^*$ and high HLR and $k$. These correspond to a very wide range of potential $k$ and $C^*$-values; in fact, so wide that there is little resolution of the data by the model. Lines 3 and 4 represent a central tendency of the data, but do not entirely resolve either the $k$ or $C^*$ variability. Thus it is seen that the intersystem data

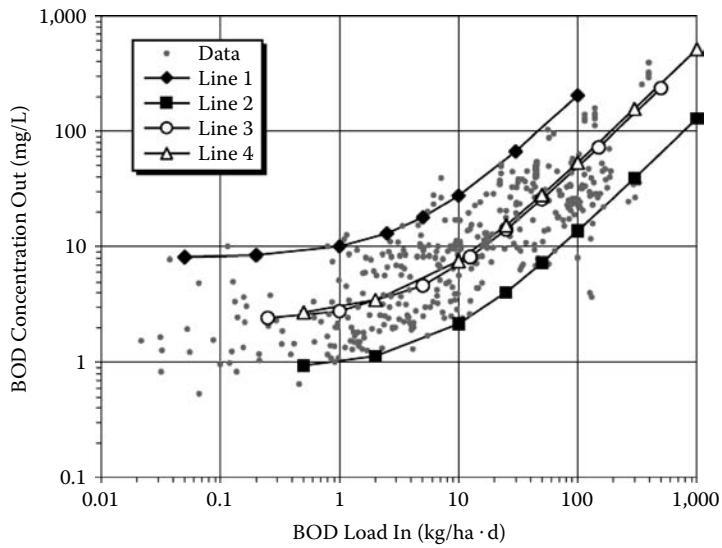



**FIGURE 8.10** Selected results for the $P$-$k$-$C^*$ model compared to annual data for BOD in FWS wetlands. The value $P = 1$ has been selected.

| Line | $C^*$ (mg/L) | $k_A$ (m/yr) | HLR (cm/d) |
|---|---|---|---|
| 1 | 10 | 15 | 1 |
| 2 | 1 | 250 | 10 |
| 3 | 3 | 60 | 5 |
| 4 | 5 | 35 | 10 |

does not aid in pinpointing narrow ranges of model parameters. In semiquantitative terms, the ranges that span the data are:

$$15 < k < 250 \text{ m/yr}$$

$$2 < C^* < 20 \text{ mg/L}$$

$$1 < P < 2$$

It is noteworthy that the central tendency reported by Kadlec and Knight (1996), i.e., $k$ = 34 m/yr and $C^* \approx 3.5$ mg/L for $P = \infty$, is still a good central estimate for the much larger data set now available.

### Variability in Annual Performances

Interestingly, the intrasystem interannual variability (year-to-year variability for one wetland with several years' data) is not necessarily much smaller than the intersystem variability (variability among several wetlands). Some single wetlands span the data cloud from one extreme to the other for different years of operation. As examples, the annual values of a few wetlands have been identified in Figure 8.11. For some, such as Poinciana, Arcata Enhancement, and Cannon Beach, the interannual variation is a significant fraction of the intersystem variation at the same loading (about 80%). Other wetlands have less interannual variability, such as Reedy Creek and Dove Creek, but still about half of the intersystem variation.

In terms of model parameters, the result is a large spread in $k$-values. This may be illustrated by examining the spread of $k$-values (for $P = 1$ and $C^* = 2$) for the various years and systems at Arcata, all working at the same site (Figure 8.12).

Out of this modeling effort, the central messages are that (1) the $P$-$k$-$C^*$ model spans the intersystem data (as it should), but that (2) there is no resolution of the wide range of parameter values that might be selected. Consequently, the $P$-$k$-$C^*$ model

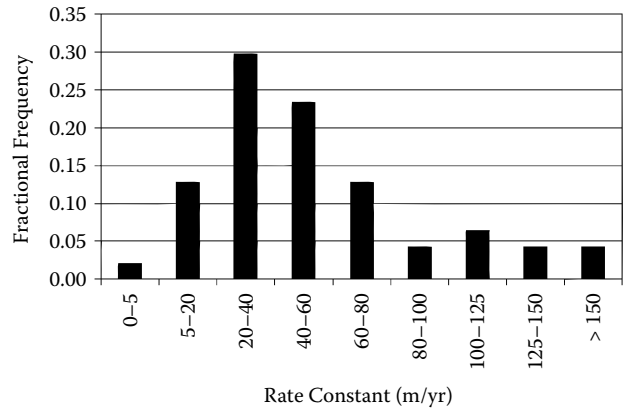



**FIGURE 8.12** Rate constants for BOD removal for the aggregate of Arcata, California, data sets. The basis is $C^* = 2$ mg/L and $P = 1$. There are 23 annual average points for the pilot cells (12 cells over two years), 12 years for the combined treatment marsh cells, and 12 years for the combined enhancement marsh cells. The site $k = 54 \pm 39$ m/yr (mean ± SD).

by itself is insufficient for wetland design. This simple model can be fit to a single profile or input–output data set, and represent it very well; but inherent variabilities remain quite large. It is not possible to say with certainty what next year's $k$-value will be, nor what the next wetland's $k$-value will be. Unfortunately, this is also true for $C^*$-values. It is informative to seek further understanding of the factors that may control performance.

### Effects of Design and Operating Conditions

#### Water Depth

In Chapter 6, it was indicated that one of two assumptions were possible as limiting cases of first-order removal models:

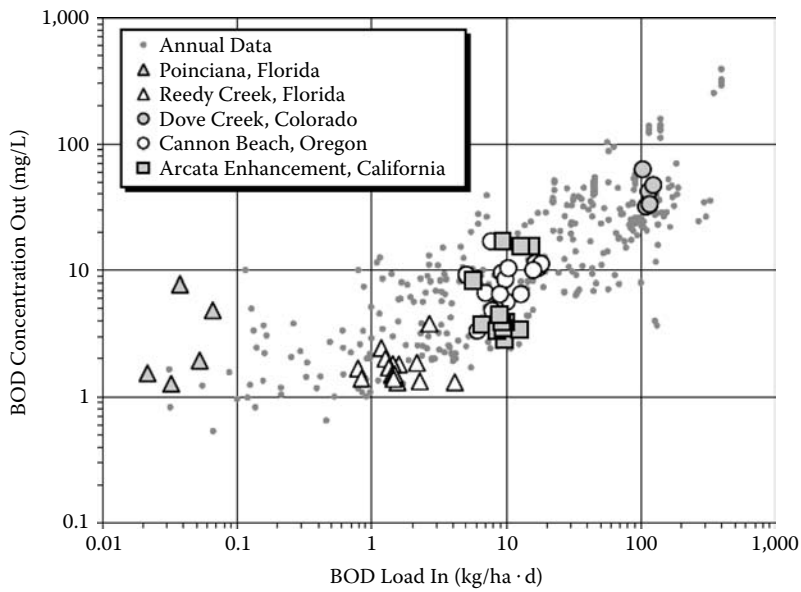



**FIGURE 8.11** Single system performance within the general milieu of annual data.

either (1) the contaminant was processed everywhere within the water column, in proportion to the water volume; or (2) the contaminant was processed in proportion to the wetland planar area. In terms of model equations, the influence is exerted through the depth dependence of removal:

$$\frac{C - C^*}{C_i - C^*} = \frac{1}{(1 + k_A / Pq)^P} = \frac{1}{(1 + k_V \tau_n / P)^P} \qquad (8.21)$$

from which it follows that

$$k_V = \frac{k_A}{h_n} \qquad (8.22)$$

where

$h_n$ = nominal wetland water depth, m
$\tau_n$ = nominal detention time, d
$k_A$ = areal rate coefficient, m/d (= m/yr ÷ 365)
$k_V$ = volumetric rate coefficient, 1/d

The question arises whether $k_A$ is constant, or whether $k_V$ is constant. In the former case, the extra detention time created by deeper operation is of no benefit, because $k_V$ is reduced as depth increases; in the latter case, increased depth creates no penalty in decreased $k_V$-values, and performance can be increased by increasing the water depth.

As one test of the two possibilities, operational data from a wetland with sequentially varied depths may be examined. The Listowel wetlands were operated at various depths over a four-year period, with the resulting ability to examine Equation 8.22. There is a strong increase in $k_V$-values with ($1/h_n$) for depths above about 5 cm (Figure 8.13), indicating that $k_A$ is more nearly constant than $k_V$. It is possible that the drop in $k_V$ for depths less than 5 cm is due to the incomplete wetting of the wetland surface.

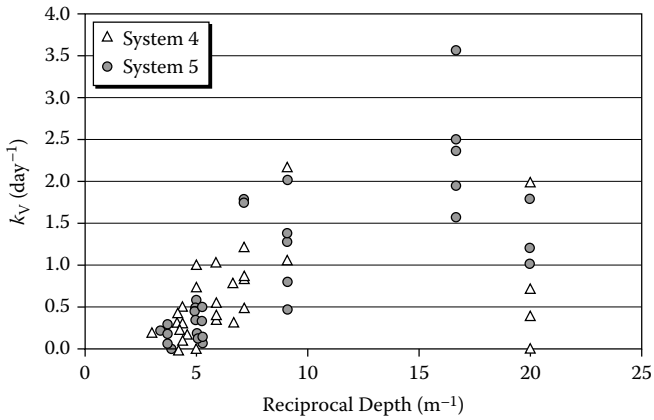



**FIGURE 8.13** Variation of the volumetric rate constant for BOD removal for Listowel, Ontario, Systems 4 and 5. The parameters $P$ = 2 and $C^*$ = 2 mg/L have been chosen.

A second test is to compare side-by-side wetlands operated at different depths. The Arcata pilot wetlands were operated in that fashion for two years. Each of three hydraulic loadings was replicated at two depths. For each loading, the value of $k_V$ was lower at the larger depth (Table 8.3). Over the entire suite of experiments, a 35% depth increase resulted in a 35% $k_V$ decrease. This also indicates that $k_A$ is more nearly constant than $k_V$.

Either $k_A$ or $k_V$ can be used to represent a data set or be used in design. However, the use of $k_V$ requires the accompanying information on water depth ($h$) because of the depth dependence indicated in Equation 8.22. This depth dependence also means that more detention time created by deeper water is counteracted by a decrease in the volumetric rate constant. The hydraulic loading rate is not depth-dependent,

**TABLE 8.3**
**Depth Effects on Rate Constants for the Arcata, California, Pilot Project**

| HLR (cm/d) | Depth (cm) | Percent Depth Increase | $k_V$ (1/day) | Percent $k_V$ Decrease |
|---|---|---|---|---|
| 0.230 ± 0.010 | 36 ± 2.0 | — | 0.71 | — |
| 0.215 ± 0.025 | 52 ± 0.5 | 31% | 0.58 | 18% |
| 0.110 ± 0.005 | 27 ± 0.5 | — | 0.52 | — |
| 0.113 ± 0.003 | 46 ± 2.0 | 41% | 0.26 | 50% |
| 0.065 ± 0.005 | 30 ± 2.0 | — | 0.39 | — |
| 0.065 ± 0.005 | 46 ± 0.5 | 35% | 0.25 | 36% |
| **Mean** | | **36%** | | **35%** |

*Note:* Twelve pilot cells were operated as duplicates at two depths and three hydraulic loading rates, over a period of two years, beginning one year after start-up. The *P-k-C** model parameters were fixed at $P$ = 1 and $C^*$ = 2 mg/L.

*Source:* From analysis of data in Gearheart *et al.* (1989). In *Constructed Wetlands for Wastewater Treatment: Municipal, Industrial, and Agricultural*. Hammer (Ed.), Lewis Publishers, Chelsea, Michigan, pp. 121–137.

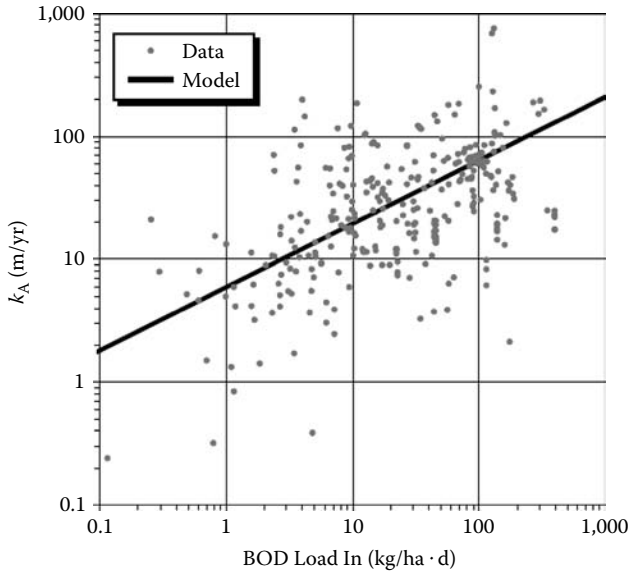



**FIGURE 8.14** Dependence of rate constants on BOD loading.

and the same data indicate that $k_A$ is nearly independent of depth. The use of areal coefficients does not require depth. For many FWS wetlands, especially large ones, depth is not known to a reasonable degree of accuracy (see Chapter 2). For these reasons, the parameter $k$ is used herein.

## Loading Effect on *k*-Values

Importantly, both $k_V$ and $k$ depend to some degree upon BLI rate. This is the observed trend of the data from a large number of free water surface wetlands (Figure 8.14). The selected parameters were $P = 2$ and $C^* = 2$ mg/L. Although the correlation depends to some extent upon the values of $P$ and $C^*$, there is no selection of these parameters that removes the dependence of $k_V$ and $k$ on the BLI rate.

The first-order model has increased sensitivity to loading if the value of $C^*$ is chosen to be zero (Kadlec, 2000). Under that assumption, the values of $k_{V1}$ are nearly proportional to BLI, or inversely proportional to the detention time, for low hydraulic loadings. The additional subscript "1" indicates that the model contains only one parameter, the $k$-value, as opposed to two ($k$ and $C^*$). This sensitivity is exacerbated if the plug flow model is used, i.e., $P = \infty$. The near-proportionality of $k_{V1PF}$ to BLI has been repeatedly recognized (Reed *et al*., 1995; Kadlec, 2000; Water Environment Federation, 2001; Ran *et al*., 2004). WEF (2001) report the following relation:

$$k_{V1PF} = 0.030 + 0.00648 \cdot \text{BLI} \qquad (8.23)$$

where

BLI = BOD loading in, kg/ha·d

$k_{V1PF}$ = plug flow rate constant with $C^* = 0$, $d^{-1}$

This dependence leads to a design paradox. The required wetland area is inversely proportional to the $k$-value, whereas the inlet BLI is inversely proportional to wetland area. Suppose a BLI has been chosen as a first estimate, and the corresponding $k$-value determined (e.g., from Equation 8.23); and the predicted outlet BOD is too high. The obvious correction is to increase area. However, that lowers the inlet BLI, and according to Equation 8.23, also lowers the $k$-value. Clearly, this is a useless procedure. Reed *et al.* (1995) dispose of the difficulty by ignoring Equation 8.23. This regression is an example of the spurious correlation caused by hydraulic loading appearing in both the abscissa and ordinate (see Chapter 6).

## Temperature

The first-order model has been reliable for predicting removal rates of organic matter in most wastewater treatment processes (Metcalf and Eddy Inc., 1991). The modified Arrhenius relationship is commonly used to adjust the removal rate coefficient for temperature in traditional wastewater treatment processes:

$$k_{V1} = k_{V1,20}\theta^{(T-20)} \qquad (8.24)$$

where

$k_{V1}$ = rate constant at temperature $T$, $d^{-1}$

$k_{V1,20}$ = rate constant at 20°C, $d^{-1}$

$T$ = water temperature, °C

$\theta$ = modified Arrhenius temperature factor, dimensionless

Values of $\theta$ for various treatment technologies range from 1.00 to 1.08, with typical values of 1.04 for activated sludge, 1.08 for aerated lagoons, and 1.035 for trickling filters (Metcalf and Eddy, Inc., 1991). The temperature dependence of the BOD test itself is generally taken to be 1.047 (Crites and Tchobanoglous, 1998). These traditional process units differ considerably from wetlands, in terms of functional complexity and operating conditions. They are designed to provide intense focus on microbial processes alone, without other biotic components or the spatial heterogeneity of a treatment wetland.

The treatment wetland literature is replete with the assertion that a $\theta$-value of about 1.06 applies to FWS wetlands (Reed *et al*., 1988: 1.10; U.S. EPA, 1988b: 1.10; Reed *et al*., 1995: 1.06; Crites and Tchobanoglous, 1998: 1.06; Campbell and Ogden, 1999: 1.06; U.S. EPA, 2000a: 1.04). These reports all referred to the plug flow model with $C^* = 0$. However, Kadlec and Knight (1996) could not find a temperature dependence in wetland BOD data. That finding was subsequently supported by analysis of more systems (Kadlec and Reddy, 2001).

The two most closely related companion technologies for BOD reduction are overland flow and stabilization ponds. The former involves very shallow (a few centimeters depth at most) water flow over a vegetated surface, and the latter represent algal-aquatic systems with typical depths of one to two meters. Thus, these technologies may be regarded as the shallow- and deepwater extremes of treatment wetlands. The

data from those systems yield temperature coefficients that are close to 1.00 for ponds (1.005 ± 0.014) and overland flow (1.01 ± 0.01) (Kadlec and Reddy, 2001). U.S. EPA (1983a) suggests several different design approaches for facultative ponds, including equivalents to the first-order model presented above. The suggested design temperature factors are θ = 1.085 and 1.090. However, U.S. EPA (1983a) show a data basis that produces θ = 0.995. The authors explain this as follows: "The logical explanation for the lack of influence by temperature is that the pond systems are so large that the temperature effect is masked by other factors." No explanation was offered for rejecting the observed behavior in the recommended design calculations. This lack of a temperature effect in ponds has more recently been reported by Abis (2002).

Here the temperature effect on performance of several wetland systems has been re-analyzed with the *P-k-C** model, with $P = 1$ (Table 8.4). The θ-value is 0.985 ± 0.021, meaning slightly worse performance at higher temperatures. Little or no variance is removed by adding a θ-factor to the model. It is clear that the complex of wetland ecosystem processes is masking the known microbial temperature sensitivity expected for suspended growth systems. One candidate explanation is oxygen transfer, which must be adequate to justify the first-order approximation. However, as seen in the earlier section on carbon processing, many other processes can influence BOD removal.

The preponderance of evidence suggests that wetland BOD removal is not improved at higher wetland water temperatures.

### Seasonal Trends

There are typically gentle annual cycles in the effluent BOD from FWS wetlands (Figure 8.15). A maximum is seen in spring or summer, and the amplitude of the annual cycle is on the order of 30% of the mean (Table 8.5). The trend is described by

$$C = C_{avg}\left[1 + A \cdot \cos\left(\omega(t - t_{max})\right)\right] \qquad (8.25)$$

where

$A$ = trend fractional amplitude, dimensionless
$C$ = concentration, mg/L
$C_{avg}$ = mean annual concentration, mg/L
$t$ = yearday, d
$t_{max}$ = yearday for maximum concentration, d
$\omega$ = annual period, 0.01721 $d^{-1}$

These cycles often do not reflect contemporary influent BOD or the contemporary hydraulic loading to the wetland. This is evidenced in Figure 8.15, where minima of the inlet concentration correspond to maxima of the outlet concentration, for relatively uniform hydraulic loading throughout the year.

**TABLE 8.4**
**Temperature Factors for the *P-k-C** Model for Example FWS Wetlands**

| Wetland | Cell | Data (years) | $k_{20}$ (m/yr) | $C^*$ (mg/L) | θ |
|---|---|---|---|---|---|
| Brighton, Ontario | 1 | 4 | 25 | 4 | 0.946 |
| Columbia, Missouri | 1 | 2 | 450 | 8 | 0.996 |
| Listowel, Ontario | 1 | 4 | 30 | 2 | 1.002 |
| | 2 | 4 | 36 | 5 | 1.035 |
| | 3 | 4 | 19 | 3 | 0.932 |
| | 4 | 4 | 89 | 5 | 0.986 |
| | 5 | 4 | 49 | 6 | 0.977 |
| Arcata, California | 1 | 2 | 51 | 0 | 0.993 |
| | 2 | 2 | 92 | 9 | 0.973 |
| | 3 | 2 | 44 | 4 | 0.993 |
| | 4 | 2 | 56 | 4 | 0.999 |
| | 5 | 2 | 60 | 6 | 0.978 |
| | 6 | 2 | 76 | 3 | 0.988 |
| | 7 | 2 | 33 | 0 | 0.989 |
| | 8 | 2 | 50 | 11 | 0.999 |
| | 9 | 2 | 25 | 0 | 0.980 |
| | 10 | 2 | 23 | 0 | 0.975 |
| | 11 | 2 | 54 | 4 | 0.992 |
| | 12 | 2 | 73 | 5 | 0.976 |
| **Mean** | | | | | **0.985** |

*Note:* The value $P = 1.0$ has been selected. Model fits are not good, in the sense that $R^2$-values do not increase much when a θ-factor is added.

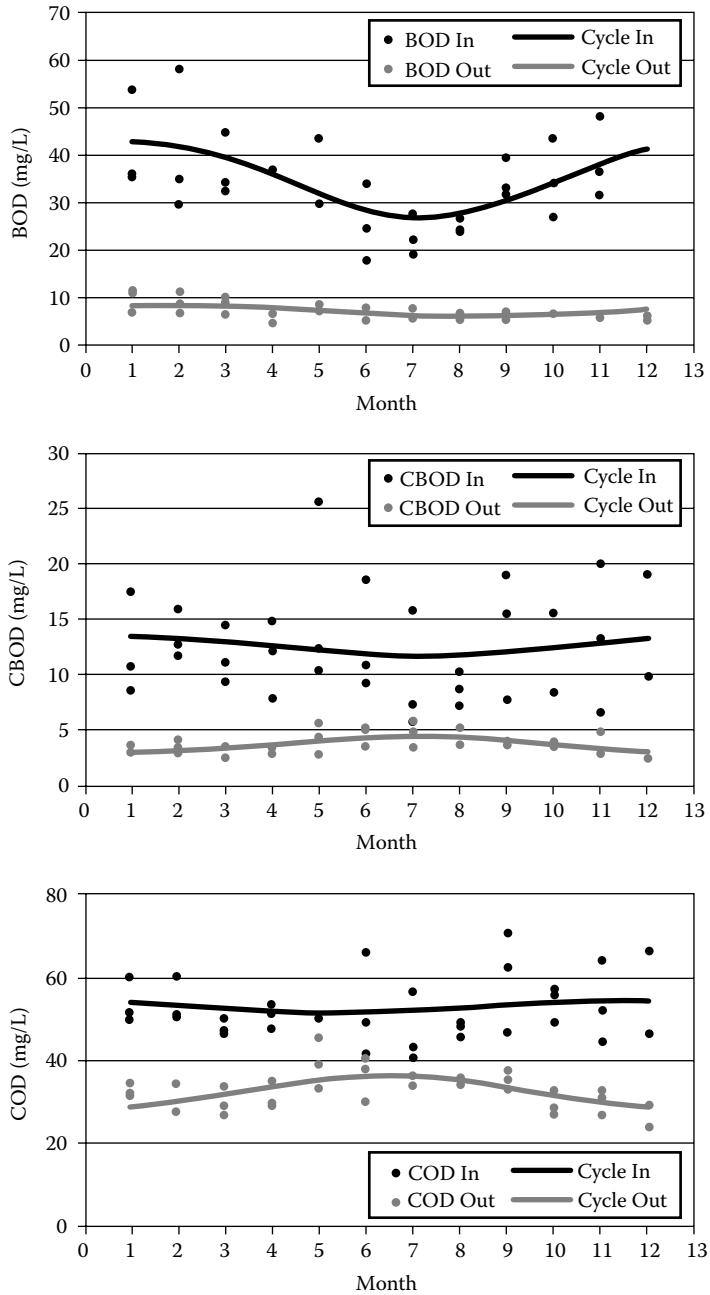



**FIGURE 8.15** Annual cycles of BOD, CBOD, and COD for three years of monthly averages of daily data from Columbia, Missouri.

The considerable scatter in effluent concentrations contributes to low $R^2$-values for the trend lines (Table 8.5). This behavior is of concern in wetland sizing, if the peak values of the concentrations are of importance in the regulatory compliance for the project.

## Variability around Seasonal Trends

Because stochastic behavior is present in moderate amount, it is necessary to quantify performance variability, and ultimately to modify sizing based upon that understanding. Average effluent BOD values over short time periods are subject to variation from the annual mean. The longer the averaging period, the closer the short-term mean value is to the annual mean value. For FWS wetlands, average effluent BOD concentrations are distributed approximately according to the log normal distribution. Examples of these distributions are given in Figure 8.16.

The averaging period has a very strong influence on the higher percentiles, which form the basis for permit requirements. The example given in Figure 8.17 shows that for the Columbia, Missouri, system, the daily maximum is about triple the monthly maximum, and the weekly maximum is about double the monthly maximum. These ratios shrink as

**TABLE 8.5**
**Sinusoidal Trends in FWS Wetland Effluent BOD Concentrations during the Course of the Year**

| Site | POR* Months | Operation Period | Sample Frequency | Averaging Period | Trend Mean (mg/L) | Trend Fractional Amplitude | Trend $t_{max}$ (Julian day) | Trend $R^2$ |
|---|---|---|---|---|---|---|---|---|
| **BOD** | | | | | | | | |
| Estevan | 59 | Summer | Weekly | None | 4.8 | 0.31 | 201 | 0.02 |
| Listowel 1 | 48 | Annual | Weekly | Monthly | 8.1 | 0.36 | 78 | 0.12 |
| Listowel 2 | 48 | Annual | Weekly | Monthly | 11.0 | 0.55 | 67 | 0.22 |
| Listowel 3 | 48 | Annual | Weekly | Monthly | 7.3 | 0.44 | 67 | 0.40 |
| Listowel 4 | 48 | Annual | Weekly | Monthly | 9.5 | 0.38 | 145 | 0.13 |
| Listowel 5 | 48 | Annual | Weekly | Monthly | 14.1 | 0.30 | 67 | 0.10 |
| Cannon Beach | 192 | Dry season | Monthly | Monthly | 7.3 | 0.12 | 105 | 0.11 |
| Columbia | 36 | Annual | Daily | Monthly | 7.3 | 0.16 | 54 | 0.27 |
| **CBOD** | | | | | | | | |
| Columbia | 15 | Annual | Daily | None | 10.8 | 0.31 | 114 | 0.25 |
| Columbia | 15 | Annual | Daily | Weekly | 10.9 | 0.32 | 122 | 0.41 |
| Columbia | 15 | Annual | Daily | Monthly | 10.8 | 0.31 | 116 | 0.71 |
| Brighton | 39 | Annual | Weekly | Monthly | 4.4 | 0.29 | 18 | 0.40 |
| Orlando Easterly | 120 | Annual | 3× Monthly | 3× Monthly | 0.9 | 0.09 | 62 | 0.01 |
| Tres Rios H1 | 84 | Annual | Weekly | Monthly | 2.9 | 0.53 | 215 | 0.12 |
| Tres Rios H2 | 84 | Annual | Weekly | Monthly | 2.4 | 0.23 | 190 | 0.07 |
| Arcata Treatment | 156 | Annual | Weekly | Weekly | 23.2 | 0.13 | 285 | 0.04 |
| Arcata Enhancement | 120 | Annual | Weekly | Weekly | 3.8 | 0.33 | 22 | 0.12 |
| **TOC** | | | | | | | | |
| Estevan | 59 | Summer | Weekly | None | 18.4 | 0.21 | 251 | 0.06 |
| Orlando Easterly | 120 | Annual | 3× Monthly | 3× Monthly | 10.2 | 0.11 | 160 | 0.22 |
| Tres Rios H1 | 84 | Annual | Weekly | Monthly | 8.5 | 0.17 | 164 | 0.20 |
| Tres Rios H2 | 84 | Annual | Weekly | Monthly | 7.9 | 0.13 | 124 | 0.34 |
| **COD** | | | | | | | | |
| Tres Rios H1 | 84 | Annual | Weekly | Monthly | 42.2 | 0.38 | 156 | 0.28 |
| Tres Rios H2 | 84 | Annual | Weekly | Monthly | 41.7 | 0.23 | 132 | 0.06 |
| Columbia | 36 | Annual | Daily | Monthly | 32.6 | 0.12 | 179 | 0.38 |

* POR = period of record

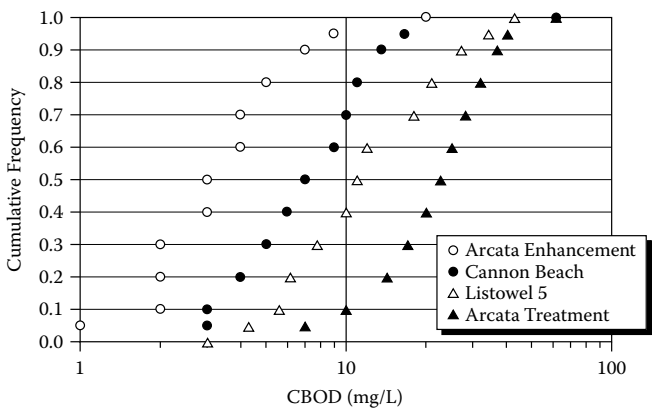



**FIGURE 8.16** Frequency distributions for monthly BOD (Cannon Beach, Oregon; Listowel, Ontario) and weekly CBOD (Arcata, California).

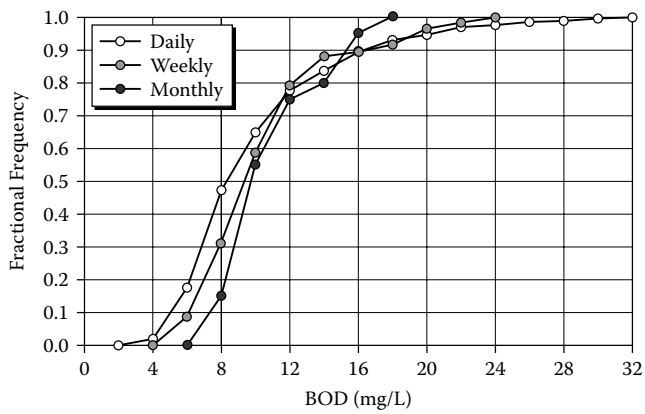



**FIGURE 8.17** Frequency distributions at the Columbia, Missouri, wetlands for daily (five days out of seven), weekly (average of five dailies), and monthly BOD values (average of 22 dailies). The 90th percentile is about 1.6 times the mean. However, the maximum daily value is about triple the mean.

**TABLE 8.6**
**Trend Multipliers for Effluent BOD Distributions in FWS Wetlands**

| Site | Trend Mean (mg/L) | Trend Multiplier (80th percentile) | Trend Multiplier (90th percentile) | Trend Multiplier (95th percentile) | Trend Multiplier (100th percentile, maximum) |
|---|---|---|---|---|---|
| **BOD** | | | | | |
| Estevan, Saskatchewan | 4.8 | 1.53 | 2.37 | 2.88 | 3.68 |
| Listowel 1, Ontario | 8.1 | 1.66 | 1.85 | 2.36 | 2.74 |
| Listowel 2, Ontario | 11.0 | 1.46 | 1.82 | 2.20 | 3.26 |
| Listowel 3, Ontario | 7.3 | 1.60 | 2.26 | 2.78 | 3.45 |
| Listowel 4, Ontario | 9.5 | 1.46 | 1.97 | 2.51 | 2.90 |
| Listowel 5, Ontario | 14.1 | 1.51 | 2.15 | 2.35 | 2.60 |
| Cannon Beach, Oregon | 7.3 | 1.39 | 1.60 | 1.88 | 2.39 |
| **CBOD** | | | | | |
| Columbia, Missouri | 10.8 | 1.25 | 1.55 | 1.72 | 2.41 |
| Columbia, Missouri | 10.9 | 1.25 | 1.38 | 1.51 | 1.65 |
| Columbia, Missouri | 10.8 | 1.15 | 1.18 | 1.19 | 1.20 |
| Brighton, Ontario | 4.4 | 1.32 | 1.55 | 1.75 | 2.09 |
| Orlando Easterly, Florida | 0.9 | 1.30 | 1.45 | 1.57 | 1.80 |
| Tres Rios H1, Arizona | 2.9 | 1.43 | 1.96 | 2.37 | 4.11 |
| Tres Rios H2, Arizona | 2.4 | 1.44 | 1.77 | 2.18 | 3.49 |
| Arcata, California Treatment | 23.2 | 1.24 | 1.35 | 1.42 | 1.66 |
| Arcata, California Enhancement | 3.8 | 1.36 | 1.80 | 2.24 | 4.98 |
| **TOC** | | | | | |
| Estevan, Saskatchewan | 18.4 | 1.30 | 1.37 | 1.56 | 2.80 |
| Orlando Easterly, Florida | 10.2 | 1.09 | 1.15 | 1.21 | 1.36 |
| Tres Rios H1, Arizona | 8.5 | 1.10 | 1.27 | 1.36 | 1.74 |
| Tres Rios H2, Arizona | 7.9 | 1.11 | 1.15 | 1.18 | 1.42 |
| **COD** | | | | | |
| Tres Rios H1, Arizona | 42.2 | 1.34 | 1.56 | 1.72 | 2.10 |
| Tres Rios H2, Arizona | 41.7 | 1.16 | 1.47 | 1.80 | 4.52 |
| **Average All** | | **1.33** | **1.56** | **1.78** | **2.51** |

*Note:* The corresponding trend line information is given in Table 8.5. Trend multiplier is $(1+\Psi)$; see Equation 6.61.

the percentile is relaxed to 90th, where they are all about equal.

Example seasonal trend information has been given (Table 8.5). An exploration of the scatter of data around those trends establishes the percentile rank of multipliers to the trend values. Table 8.6 shows that the carbon compound multiplier of the trend is 1.78. That means that excursions in individual monthly samples greater than 78% higher than the trend may be expected in 1 month out of 20.

## Model Dynamics

The response of a FWS wetland to changes in operating conditions does not necessarily follow a first-order model, or any other deterministic model that pertains only to the surface water body. Changes in inlet concentrations, for instance, may not be reflected in outlet concentrations, even if allowance is made for transport delay. For example, the Columbia wetlands had a detention time of three to four days, and experienced month-to-month variations in inlet BOD spanning 10–50 mg/L (Figure 8.18). During the same period, outlet BOD ranged from 5 to 30 mg/L, but there is not a correspondence between peaks, i.e., there is no tracking of the inlet to be seen in the outlet.

The conclusion is that no deterministic removal model now available in the literature (*P-k-C** model included) should be used to predict high frequency BOD events, even to the scale of monthly variations.

## Oxygen Supply

If removal of BOD is via Equation 8.10, oxygen transfer must be adequate to justify the first-order removal approximation. However, processes detailed in Equations 8.11 through 8.18 can also influence BOD removal, especially in heavily loaded systems. Fermentation, nitrate, iron, and sulfate reduction are all potential consumers of carbon compounds in the absence of free oxygen. Ultimately, under very low redox conditions, methanogenesis may take place. Therefore, the implied

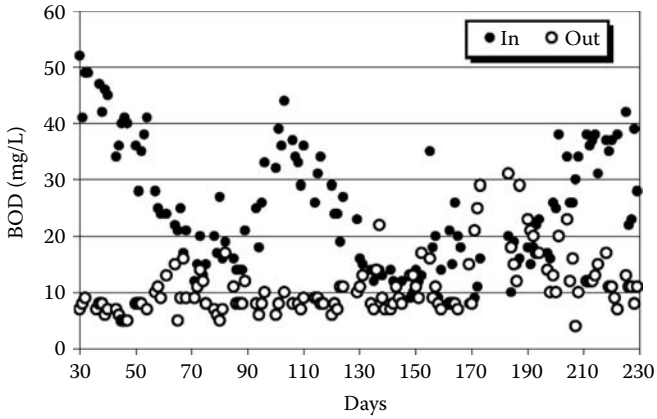



**FIGURE 8.18** Daily BOD for a 200-day period at Columbia, Missouri, commencing October 24, 1994. The hydraulic loading rate during this period was relatively steady at 15 ± 2 cm/d, equivalent to a nominal detention time of about three days.

*maximum* oxygen supply for BOD removal is simply the load of BOD removed. The systems that form the basis for Figure 8.9 have the median oxygen requirements shown earlier in Table 8.7. The supply to the water in FWS wetlands is likely to be no more than 2–4 g/m²·d (see Chapter 5). Therefore, as the incoming BOD increases to the levels seen in primary and super treatment situations, it is unlikely that oxidative processes are entirely responsible for the destruction of BOD compounds. Additional mechanisms, such as anaerobic digestion (methanogenesis) become important contributors to removal. It is tempting to speculate that aeration of the wetland water may be enhanced by open water sections, but that is a questionable hypothesis as seen in the next section.

### Open Water Fraction

BOD is reduced in both ponds and wetlands. However, there are differences in several aspects of these systems that argue for differences in their relative BOD removal capabilities. The loading graph may be used to explore intersystem effects of open water. In a broad context, multiple data sets are represented by trends that show decreasing $C_o$ with decreasing BLI (see Figure 8.19). For BLI less than about 100 kg/ha·d, there appears to be little difference between ponds and wetlands for BOD removal (Figure 8.19). At higher loadings, there is a strong suggestion that ponds are better than wetlands, although wetland data is sparse at high loadings (Kadlec, 2005e). It is perhaps ironic that the upper BLI limit sometimes imposed for pond operation of 80–90 kg/ha·d (Shilton, 2005; Crites *et al*., 2006) represents the lower limit for which pond performance is distinctly better than wetland performance.

Open water areas have been suggested as necessary and optimal for BOD reduction in FWS systems, for loadings up to 60 kg/ha·d (U.S. EPA, 2000a). Performance data do not support that hypothesis (Figure 8.20). However, open water zones do not appear to impair BOD removal.

## 8.3 BOD REMOVAL IN HSSF WETLANDS

A large amount of BOD data now exists for HSSF wetlands, mostly treating domestic wastewaters. The same ways are used to summarize this information as for FWS wetlands, including removal rate models and graphical summaries. As for FWS systems, when waters with moderate to large concentrations of BOD flow through a HSSF wetland, a decrease in concentration to a nonzero plateau is typically observed. This behavior is illustrated in Figure 8.21 for two continuous flow

**TABLE 8.7**
**Load Reduction of $BOD_5$ in FWS Wetlands**

| | Tertiary (g/m²·d) | Secondary (g/m²·d) | Primary (g/m²·d) | Super (g/m²·d) |
|---|---|---|---|---|
| $C_i$ = | 3–30 mg/L | 30–100 mg/L | 100–200 mg/L | >200 mg/L |
| N = | 204 | 77 | 63 | 43 |
| **Percentile** | | | | |
| 0.05 | 0.11 | 0.19 | 1.32 | 6.37 |
| 0.10 | 0.15 | 0.30 | 1.52 | 7.24 |
| 0.20 | 0.27 | 0.56 | 2.18 | 7.75 |
| 0.30 | 0.45 | 0.71 | 2.49 | 8.36 |
| 0.40 | 0.51 | 0.85 | 2.63 | 8.89 |
| 0.50 | 0.66 | 1.33 | 2.88 | 9.39 |
| 0.60 | 0.80 | 1.55 | 3.97 | 9.74 |
| 0.70 | 0.93 | 2.35 | 4.99 | 10.84 |
| 0.80 | 1.08 | 3.74 | 5.74 | 23.83 |
| 0.90 | 1.45 | 5.11 | 8.10 | 26.42 |
| 0.95 | 2.07 | 8.85 | 10.15 | 27.33 |

*Note:* These amounts are the implied oxygen requirement for aerobic destruction of the compounds that comprise $BOD_5$. N represents wetland·years.

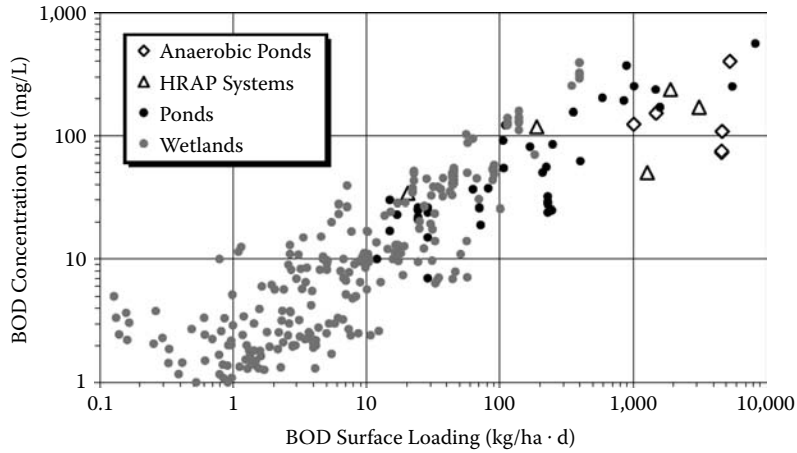



**FIGURE 8.19** Response annual average effluent BOD of aquatic systems to increasing annual average BOD loadings. Wetlands are represented by 265 years of data for 113 systems. Pond data are for 51 systems over their period of data record. Wetland data are from the North American Database (1998); together with unpublished data. Pond data are from U.S. EPA (1983a), Mendes *et al*. (1994), Pearson *et al.* (1995), Soler *et al*. (1995), El Hamouri *et al.* (1995), Abis (2002), Tadesse *et al.* (2003), Craggs *et al.* (2003); together with unpublished data.

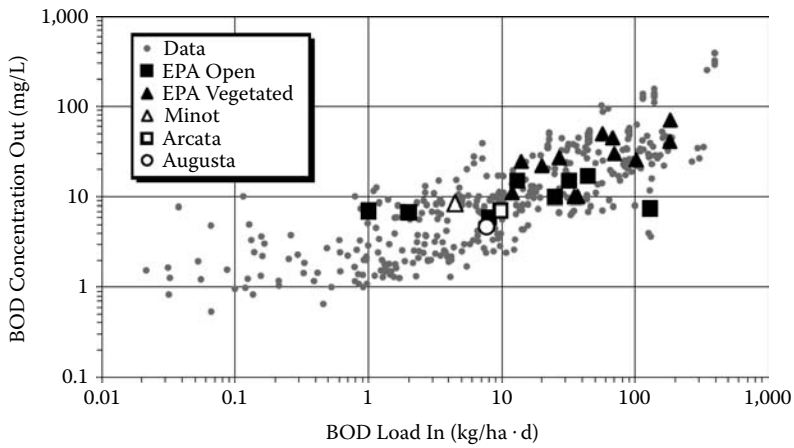



**FIGURE 8.20** Wetlands with open water sections. The solid points are plotted from U.S. EPA (2000a). The open points represent wetlands built with large open water components in their central region. Dots are the general milieu of FWS performances.

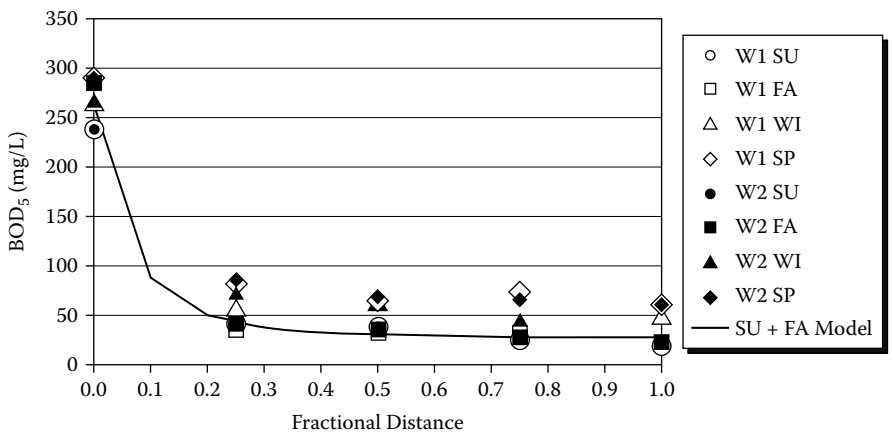



**FIGURE 8.21** Longitudinal profiles of $BOD_5$ at the two NERCC, Minnesota, HSSF wetlands (W1 & W2), over a two-year period of record, by quarter. Note the plateau concentration is somewhat higher in winter and spring. The *P-k-C** model is shown as the solid line, for the summer and fall period. The fit values are $P = 4$, $k = 66$ m/yr, $C^* = 27$ mg/L, with an $R^2 = 0.998$.

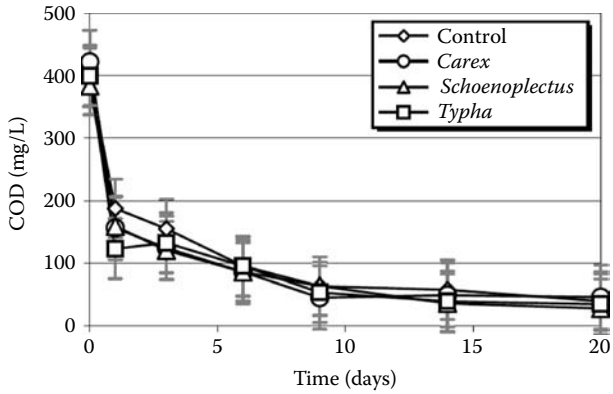



**FIGURE 8.22** Reduction of COD in planted and unplanted batch SSF gravel mesocosms. The pollutant solution was concocted from meat protein and sucrose. (From Stein *et al.* (2006a) *Ecological Engineering* 26(2): 100–112. Reprinted with permission.)

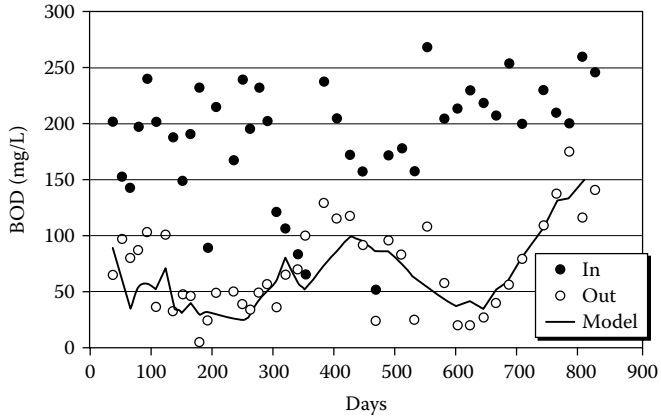



**FIGURE 8.23** First-order *P-k-C** fit for Grand Lake, Minnesota. The value of $P = 2$, $k_{20} = 33$ m/yr. The fitting process is insensitive to $C^* = 20$ mg/L. The temperature coefficient is $\theta = 1.140$, the highest of any system studied here. The value of $R^2 = 0.93$. (From unpublished data.)

HSSF wetlands near Duluth, Minnesota (NERCC project, described in Kadlec *et al*., 2003). Samples were taken along the wetland length, from internal wells, and the midpoint transfer between the two cells in series for each of the two systems. The same sort of response is seen in batch loaded systems (Stein *et al*., 2006a, b). Typical response data showed a sharp decrease in $BOD_5$ and COD to a nonzero background (Figure 8.22). The decrease in these batch experiments is reasonably well fit by the *k-C** model.

In more general terms, the *P-k-C** first-order model can readily account for these observations, for appropriate values of parameters (see Chapter 6). However, as for FWS wetlands, parameter values are known to depend on system hydraulics (Kadlec, 2000), as well as on speciation of the BOD (Crites and Tchobanoglous, 1998; Kadlec, 2003a).

## First-Order Modeling

The considerations of weathering as well as speciation of BOD and chemical oxygen demand (COD) that were discussed for FWS wetlands also apply for HSSF systems. It is anticipated that the *P-k-C** model (Equation 8.20) will apply, with the parameter *P* being somewhat less than the *N*TIS value determined for a nonreactive tracer (Kadlec, 2003a).

The parameter *P* accounts for two effects: the detention time distribution (DTD) and the *k*-value distribution (*k*VD) (see Chapter 6). The value of *P* is always less than the number of tanks determined from a tracer test. For broad distributions of *k*-values, such as may occur for BOD, the typical HSSF hydraulic TIS number of six to ten (see Table 6.2) will be reduced to a *P*-value of three or four. The *C**-value in Equation 8.20 reflects several possible different causes for HSSF, as for FWS. A number of different approaches to data fitting may be used.

Reasonable data fits may be obtained for time series for specific wetlands. As an example, the data from Grand Lake, Minnesota, are shown, along with the model fit, for a two-year period of record (see Figure 8.23). This wetland was tracer-tested, and produced *N*TIS = 3.3. This value was reduced to $P = 2$ for the fitting process. The major time trend is captured, but considerable scatter remains. In order that both *k* and *C** can be determined with a good degree of certainty, the wetland must experience significant changes in loadings and concentrations over the course of time. If the wetland is operated in the batch mode, it is reasonable to expect that the exponential ($P = \infty$) form of Equation 8.20 should be used, perhaps after decrementing *P* for a possible *k*VD (Stein *et al*., 2006a). An example of an exponential fit is shown in Figure 8.23.

Interior distance profiles may be fit for a given wetland. Figure 8.21 displays such a fit for the NERCC wetlands near Duluth, Minnesota. This system had two cells in series, in each of two trains. Therefore, the value $P = 4$ was assigned. The values $k = 66$ m/yr and $C^* = 27$ mg/L fit the data quite well ($R^2 = 1.00$), but the interior points are sparse. The profile shows that a plateau is reached in the front end of the train, with most of the system exhibiting a nondecreasing concentration. This is an extremely important feature of HSSF systems, because it suggests that the use of output information will often reflect the background concentration, and not contain any information on the drop-off to that outlet (plateau) concentration. One disadvantage of the profile fitting method is that it requires extensive interior monitoring, which is often not feasible. A second disadvantage is that interior sample points may not be situated in an "average" part of the flow path. If there are cells in series (there were two at NERCC), the transfer structure will provide a flow-weighted sample, but interior sample points do not necessarily do the same. For instance, attempts to sample at three cross-cell positions, and three distances, at the Benton, Kentucky, facility produced no consistent patterns (TVA, 1990). Multiple internal sample points at Minoa, New York, in three dimensions, also produced

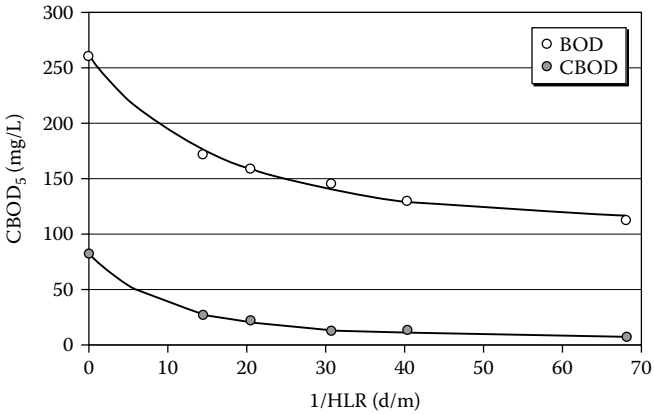



**FIGURE 8.24** Reduction of $CBOD_5$ and $BOD_5$ in side-by-side *Schoenoplectus* SSF gravel wetlands operated at different hydraulic loading rates. The pollutant solution was concocted from meat protein and sucrose. The model lines are for $P = 4$, $C^* = 6.6$ mg/L, and $k = 38$ m/yr ($R^2 = 0.94$) for $CBOD_5$, and $C^* = 106$ mg/L and $k = 22$ m/yr ($R^2 = 0.97$) for $BOD_5$. (Data from Tanner *et al.* (1998b) *Journal of Environmental Quality* 27(2): 448–458.)

**TABLE 8.8**
**First-Order Areal *k*-Values for HSSF Wetlands, Based Upon Period of Record Input–Output Analysis**

| | Tertiary | Secondary | Primary | Super |
|---|---|---|---|---|
| $C_i$ = | 3–30 | 30–100 | 100–200 | >200 |
| $C^*$ = | 1 | 5 | 10 | 15 |
| $P$ = | 3 | 3 | 3 | 3 |
| N = | 52 | 53 | 51 | 27 |
| **Percentile** | | | | |
| 0.05 | 11 | 5 | 9 | 3 |
| 0.1 | 15 | 16 | 10 | 9 |
| 0.2 | 25 | 20 | 12 | 14 |
| 0.3 | 36 | 24 | 15 | 21 |
| 0.4 | 63 | 30 | 23 | 33 |
| 0.5 | 86 | 37 | 25 | 66 |
| 0.6 | 154 | 39 | 28 | 98 |
| 0.7 | 224 | 44 | 44 | 114 |
| 0.8 | 287 | 82 | 62 | 210 |
| 0.9 | 458 | 167 | 107 | 378 |
| 0.95 | 703 | 228 | 132 | 447 |

*Note:* The number of wetlands in each category is N.

erratically variable results from which model parameters cannot reliably be determined (Theis and Young, 2000).

Side-by-side wetlands may be operated at different hydraulic loading rates. These will experience the same inlet concentrations and meteorology, but will be subject to slight unavoidable differences in ecology. Figure 8.24 shows such a fit for the Hamilton, New Zealand, wetlands, which received dairy parlor effluent (Tanner *et al*., 1998b). This system had five cells in parallel, planted with *Schoenoplectus* (*Scirpus*) *tabernaemontani*. As the hydraulic loading rate is decreased (nominal HRT increases), the outlet concentration for the two-year period of record decreases to a plateau value, identified with $C^*$. This technique is obviously a research tool only, because multiple wetlands at different loading rates will not be built for routine service.

Long-term average input–output data may be used to estimate $k$-values for assumed values of $P$ and $C^*$. However, if the outlet concentration is close to or at $C^*$, the estimate will be a lower bound. Under that circumstance, the concentration may have dropped to near $C^*$ well before the system effluent point, meaning the $k$-value could have been much higher. As an example, see Figure 8.21. There are input–output data at the end of Cell 1 ($x = 0.5$), and at the end of Cell 2 ($x = 1.0$). Analysis, using $C^* = 27$ mg/L, shows:

$$\text{Profile fit: } k = 66 \text{ m/yr}$$

$$\text{Cell 1 I/O fit: } k = 23.5 \text{ m/yr}$$

$$\text{Cells 1+2 I/O fit: } k = 16.4 \text{ m/yr}$$

This pitfall can be avoided only if the outlet concentration is well above the presumptive $C^*$-value. The reader is referred to Kadlec (2000) for further details of such potential misleading interpretations, and to the discussion in Kadlec and Knight (1996). To summarize, the first-order model appears to be perfectly capable of describing BOD profiles and time series in HSSF wetlands. However, if only input–output data are analyzed, there is a strong chance that $k$-values will be lower than those from longitudinal transects. In turn, it implies that extrapolation to lower loading rates will be risky, although extrapolation to higher loading rates will be overly conservative. As an indicator of the $k$-values to be expected, Table 8.8 shows the percentile points of distributions of long-term average input–output $k$-values for HSSF for selected $C^*$ and $P = 3$.

It is noteworthy that the central tendency reported by Kadlec and Knight (1996), i.e., $k = 180$ m/yr and $C^* \approx 3.5$ mg/L for $P = \infty$, is not a good estimate for the much larger data set now available. Depending on the strength of the wastewater being treated, $k$-values are lower, and have a broad intersystem distribution of values.

## Loading Effect on *k*-Values

As a consequence of the plateau effect, the $k$-values for BOD are hydraulic load-dependent (Figure 8.25). The values of $k$ are nearly proportional to hydraulic loading or inversely proportional to the detention time. The same result holds for dependence on the BLI to the wetland. The near-proportionality of $k_{V1PF}$ to BLI has been repeatedly recognized (Reed *et al*., 1995; Kadlec, 2000; Water Environment Federation, 2001; Ran *et al*., 2004). WEF (2001) report the following relation for HSSF wetlands:

$$k_{V1PF} = 0.050 + 0.01054 \cdot \text{BLI} \qquad (8.26)$$

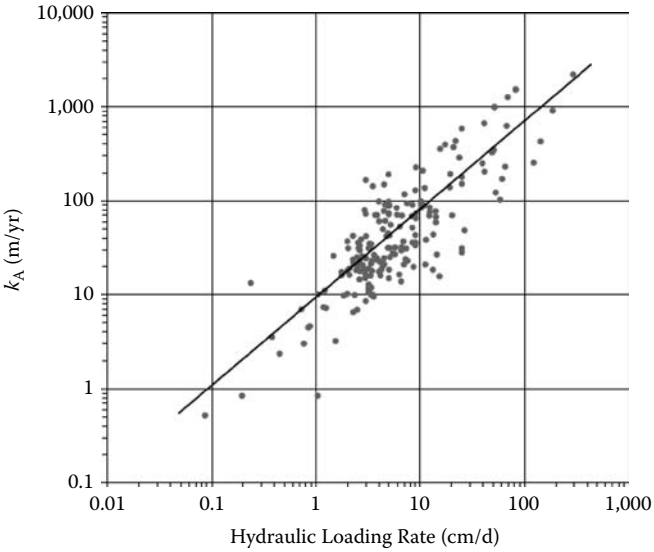



**FIGURE 8.25** Dependence of the first-order areal rate constant on hydraulic loading. The values $P = 3$ and $C^* = 2$ mg/L have been used. The trend line has $R^2 = 0.76$.

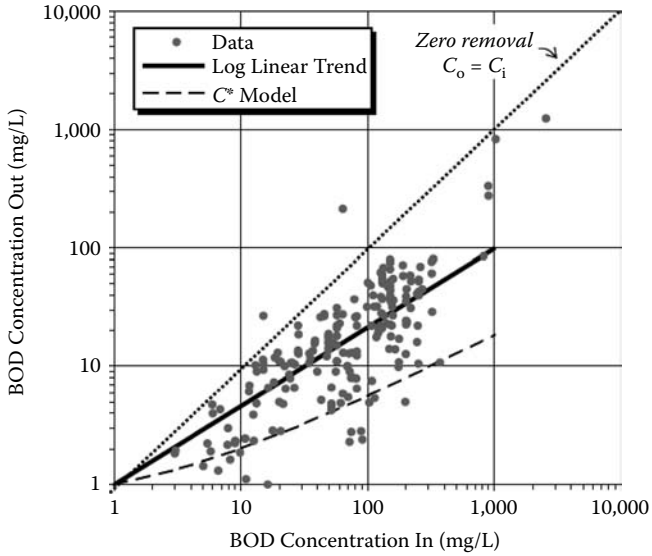



**FIGURE 8.27** BOD input–output concentration graph for 202 HSSF wetlands. There is one data point per wetland, covering the entire period of record. The log-linear central tendency regression is $\log_{10}(C_o) = 0.66 \log_{10}(C_i)$, $R^2 = 0.60$. The lower bound curve, excluding 5% of the lowest values, is $C^* = 0.6 + 0.4(C_i)^{0.55}$.

where

BLI = BOD Loading In, kg/ha·d

$k_{V1PF}$ = plug flow rate constant with $C^* = 0$, $d^{-1}$

As noted in the section on FWS in this chapter, this dependence leads to a design paradox. This graph is also subject to the spurious effect of containing the hydraulic loading in both the abscissa and ordinate.

## Graphical Relations

The graphical display that has often been adopted in the literature (Kadlec and Knight, 1996; U.S. EPA, 2000a; Wallace and Knight, 2006) plots outlet BOD concentrations versus inlet BLI (Figure 8.26). In the broad context, the multiple data sets are represented by a trend that shows decreasing outlet concentration with decreasing BLI, but that relationship is obscured by large scatter. Each point in Figure 8.26 represents the average of the entire period of record data for a given HSSF wetland. Both BOD and CBOD data are represented; therefore, it is understood that some of the scatter is due to the difference between these two measures. The use of period of record averages removes seasonal variability, if any, and precludes the effects of synoptic error (see Chapter 6).

A second display is outlet concentration versus inlet concentration (Figure 8.27). This graph shows a more consistent central trend, with a log-linear regression coefficient

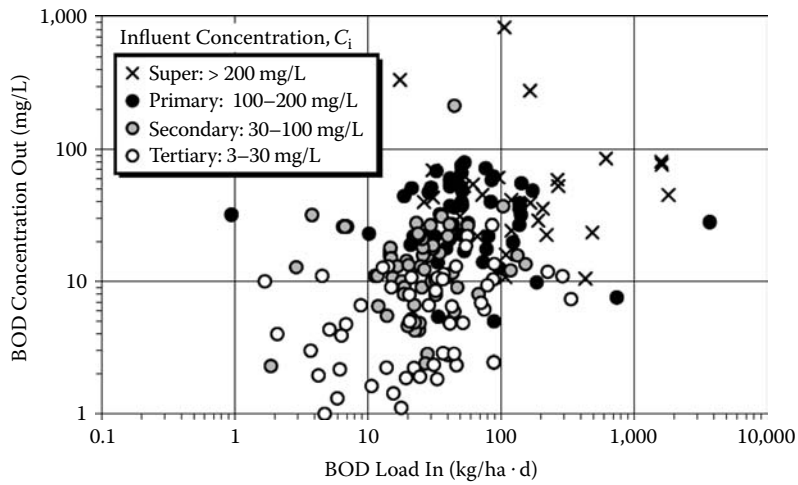



**FIGURE 8.26** BOD loading graph for 202 HSSF wetlands. There is one data point per wetland, covering the entire period of record. The ranges of inlet concentrations are separated into four groups, corresponding to tertiary ($3 < C_i < 30$ mg/L) up to super ($C_i > 200$ mg/L). A slight increasing trend effluent BOD with increased BOD loading is obscured by a very large scatter.

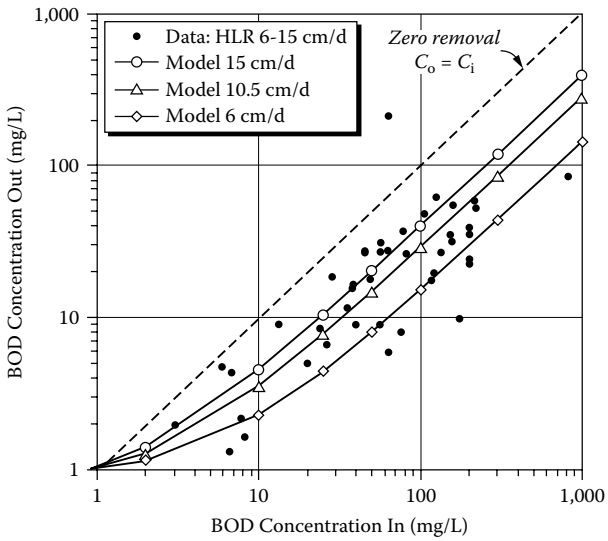



**FIGURE 8.28** BOD input–output concentrations graph for HSSF wetlands with hydraulic loading rates between 6 and 15 cm/d. There is one data point per wetland, covering the entire period of record.

$R^2 = 0.60$. Also shown on this plot is a lower bound curve, excluding about 5% of the points as potential outliers. This bounding curve may be taken as an estimate of $C^*$, and is represented by:

$$C^* = 0.6 + 0.4C_i^{0.55} \tag{8.27}$$

## Model Curves

A subset of the data cloud in Figure 8.26 has been reproduced in Figure 8.28, together with the *P-k-C** model results for parameter values $P = 3$, $k = 60$ m/yr, and $C^* = 1$ mg/L. The hydraulic loading is an independent parameter in that model, and the subset chosen for illustration is selected as those systems with $6 < HLR < 15$ cm/d. It is seen that the model results are representative of the intersystem behavior.

As for FWS wetlands, the central messages of this modeling effort are that (1) the *P-k-C** model spans the intersystem data (as it should), but that (2) there is no resolution of the wide range of parameter values that might be selected.

### Temperature Effects

The modified Arrhenius relationship is commonly used to adjust the removal rate coefficient for temperature in traditional wastewater treatment processes:

$$k = k_{20}\theta^{(T-20)} \tag{8.28}$$

where

$k$ = rate constant at temperature $T$, m/yr
$k_{20}$ = rate constant at 20°C, m/yr
$T$ = water temperature, °C
$\theta$ = modified Arrhenius temperature factor, dimensionless

The treatment wetland literature is replete with the assertion that a θ-value of about 1.06 applies to HSSF wetlands (U.S. EPA, 1993c: 1.06; Cooper *et al*., 1996: 1.10; Water Environment Federation, 2001: 1.06; Crites *et al*., 2006: 1.06). These reports all refer to the plug flow model with $C^* = 0$. However, Kadlec and Knight (1996) could not find a temperature dependence in HSSF wetland BOD data.

Here the temperature effect on performance of several HSSF wetland systems has been analyzed with the *P-k-C** model, with $P = 3$ (Table 8.9). θ-Values range from 0.891 to 1.140, with a median of 0.981. The distribution of θ-values is given in Table 8.10. θ-Values less than unity mean slightly worse performance at higher temperatures. It is clear that the presumptive value of 1.06 is at the extreme end of the distribution, and should not be expected to occur in practice, except on rare occasions. Indeed, some researchers have concluded that there is little or no temperature effect on BOD removal in HSSF wetlands (Brix, 1998). Another feature of some existing literature is a lack of discussion of temperature effects on BOD removal in HSSF wetlands (U.S. EPA, 2000a; Wallace and Knight, 2006). The preponderance of evidence suggests that wetland BOD removal is not improved at higher wetland water temperatures.

For most HSSF systems, little or no variance is removed by adding a θ-factor to the model. It is possible that the $C^*$-values for a given wetland may be temperature-dependent. Decomposition of solids in the wetland may accelerate at higher temperatures, thus providing a greater BOD return rate from wetland solids. This in turn implies that background BOD could be higher in warm periods. If, as is apparently frequently the case, the wetland outlet BOD concentration is related strongly to the $C^*$ background, then outlet BOD could be higher in summer than in winter. A first-order model without a background would show this as a reduced removal in summer. Stein *et al.* (2006b) calibrated the *k-C** model for batch operation ($P = \infty$), and allowed a temperature coefficient for both $k$ and $C^*$. The temperature coefficients for COD $C^*$ were found to be $0.958 < \theta < 1.029$, and thus did not resolve the issue.

### Oxygen Supply

If removal of BOD is via heterotrophic oxidation of carbon compounds, oxygen transfer must be adequate to justify the first-order approximation. However, anaerobic processes can also influence BOD removal, especially in heavily loaded systems. As detailed earlier, fermentation, nitrate, iron, and sulfate reduction are all potential consumers of carbon compounds in the absence of free oxygen. Ultimately, under very low redox conditions, methanogenesis may take place. The implied maximum oxygen supply for BOD removal is simply the load of BOD removed. The systems that form the basis for Figures 8.26 and 8.27 have the median oxygen requirements shown in Table 8.11. The supply to the water in HSSF wetlands is likely to be no more than 2–4 g/m²·d (see Chapter 5). Therefore, as the incoming BOD increases to the levels seen in primary and super treatment situations,

**TABLE 8.9**
**Arrhenius Temperature Factors for HSSF Wetlands**

| System | Reference | Vegetation Type | T range (°C) | Mean *C* in (mg/L) | Mean *C* out (mg/L) | Mean HLR (cm/d) | θ |
|---|---|---|---|---|---|---|---|
| Richmond, New South Wales | Bavor *et al.* (1988) | Gravel only | 11–24 | 52 | 4.3 | 3.8 | 0.961 |
| Richmond, New South Wales | Bavor *et al.* (1988) | *Typha* | 11–24 | 52 | 4.7 | 4.6 | 0.960 |
| Richmond, New South Wales | Bavor *et al.* (1988) | *Schoenoplectus* | 11–24 | 52 | 5.8 | 5.1 | 0.975 |
| Richmond, New South Wales | Bavor *et al.* (1988) | Mixed A | 11–24 | 52 | 4.3 | 4.6 | 1.024 |
| Richmond, New South Wales | Bavor *et al.* (1988) | Mixed B | 11–24 | 52 | 4.6 | 3.8 | 0.985 |
| Manhattan, Kansas | He & Mankin (2002) | PFL | 5–27 | 178 | 21 | 13.7 | 0.976 |
| Manhattan, Kansas | He & Mankin (2002) | PCL | 5–27 | 178 | 58 | 15.3 | 1.018 |
| Manhattan, Kansas | He & Mankin (2002) | PCR | 5–27 | 178 | 12 | 11.4 | 1.001 |
| Manhattan, Kansas | He & Mankin (2002) | UFL | 5–27 | 178 | 27 | 14.7 | 1.019 |
| Manhattan, Kansas | He & Mankin (2002) | UCL | 5–27 | 178 | 69 | 13.6 | 1.028 |
| Manhattan, Kansas | He & Mankin (2002) | UCR | 5–27 | 178 | 31 | 17.1 | 1.023 |
| Benton, Kentucky | TVA unpublished | 3 | 5–25 | 25 | 8 | 8.4 | 0.921 |
| Hardin, Kentucky | TVA unpublished | 1 | 6–27 | 55 | 10 | 9.7 | 0.924 |
| Grand Lake, Minnesota | Unpublished data | 1 | 1–17 | 183 | 67 | 1.0 | 1.140 |
| NERCC, Minnesota | Unpublished data | 1 + 2 | 1–16 | 239 | 22 | 1.4 | 1.056 |
| Portland, New Zealand | Unpublished data | 1 | 11–21 | 30 | 10 | 5.2 | 0.936 |
| Waipoua, New Zealand | Unpublished data | 1 | 11–21 | 64 | 11 | 0.4 | 0.936 |
| North Yorkshire 1, England | CWA (2006) | 1 | 4–15 | 191 | 58 | 4.5 | 1.073 |
| Cumbria, England | CWA (2006) | 1 | 4–17 | 9 | 2 | 15.6 | 0.983 |
| Lake Capri, Missouri | Regmi *et al.* (2003) | Nonvegetated | 2–24 | 126 | 31 | 2.3 | 1.048 |
| Lake Capri, Missouri | Regmi *et al.* (2003) | Vegetated | 2–24 | 126 | 24 | 2.3 | 1.064 |
| Fife, Scotland | CWA (2006) | 1 | 5–16 | 201 | 35 | 10.3 | 0.993 |
| Fife, Scotland | CWA (2006) | 2 | 4–15 | 201 | 24 | 6.0 | 0.978 |
| Fife, Scotland | CWA (2006) | 3 | 4–15 | 201 | 23 | 11.0 | 0.991 |
| Fife, Scotland | CWA (2006) | 4 | 4–15 | 201 | 39 | 8.3 | 1.002 |
| Hamilton, New Zealand | Tanner *et al.* (1998b) | L1 | 10–25 | 193 | 62 | 1.5 | 1.039 |
| Hamilton, New Zealand | Tanner *et al.* (1998b) | L2 | 10–25 | 193 | 73 | 2.5 | 0.896 |
| Hamilton, New Zealand | Tanner *et al.* (1998b) | L3 | 10–25 | 193 | 84 | 3.3 | 0.891 |
| Hamilton, New Zealand | Tanner *et al.* (1998b) | L4 | 10–25 | 193 | 100 | 4.9 | 0.947 |
| Hamilton, New Zealand | Tanner *et al.* (1998b) | L5 | 10–25 | 193 | 113 | 6.9 | 0.909 |
| Bozeman, Montana | Stein *et al.* (2006a) | *Carex* | 4–24 | 385 | COD | Batch | 0.954 |
| Bozeman, Montana | Stein *et al.* (2006a) | *Schoenoplectus* | 4–24 | 385 | COD | Batch | 0.965 |
| Bozeman, Montana | Stein *et al.* (2006a) | *Typha* | 4–24 | 385 | COD | Batch | 0.956 |
| Bozeman, Montana | Stein *et al.* (2006a) | Control | 4–24 | 385 | COD | Batch | 0.943 |

*Note:* Site names for U.K. systems are approximate.

**TABLE 8.10**
**Percentile Points of the Distribution of Arrhenius Temperature Factors for HSSF Wetlands, Based on Table 8.9**

| Percentile | θ |
|---|---|
| 0.05 | 0.904 |
| 0.10 | 0.922 |
| 0.20 | 0.940 |
| 0.30 | 0.956 |
| 0.40 | 0.967 |
| 0.50 | 0.981 |
| 0.60 | 0.993 |
| 0.70 | 1.018 |
| 0.80 | 1.026 |
| 0.90 | 1.054 |
| 0.95 | 1.067 |

it is unlikely that oxidative processes are entirely responsible for the destruction of BOD compounds. Additional mechanisms, such as anaerobic digestion (methanogenesis) become important contributors to removal. This lack of adequate oxygen may be overcome by (1) resorting to vertical intermittent flow wetlands, or (2) adding aeration to the HSSF bed. Vertical flow is the subject of the next section.

It is possible to design SSF wetlands that do not rely on passive diffusional processes to transfer oxygen. These systems typically operate on principles of fill-and-drain (tidal flow) (Behrends, 1999a; Austin *et al*., 2002), or HSSF wetlands that are mechanically aerated (Dufay, 2000; Wallace, 2001; Flowers, 2002; Wallace and Lambrecht, 2003), and are proprietary (patented) systems in the United States and Canada.

**TABLE 8.11**
**Load Reduction of $BOD_5$ in HSSF Wetlands**

| $C_i$ = | Tertiary ($g/m^2 \cdot d$) 3–30 mg/L | Secondary ($g/m^2 \cdot d$) 30–100 mg/L | Primary ($g/m^2 \cdot d$) 100–200 mg/L | Super ($g/m^2 \cdot d$) >200 mg/L |
|---|---|---|---|---|
| **Percentile** | | | | |
| 0.05 | 0.16 | 0.31 | 1.44 | 2.12 |
| 0.10 | 0.21 | 0.92 | 1.80 | 2.26 |
| 0.20 | 0.49 | 1.11 | 2.27 | 4.23 |
| 0.30 | 0.82 | 1.34 | 2.52 | 5.85 |
| 0.40 | 1.21 | 1.63 | 2.98 | 10.03 |
| 0.50 | 1.55 | 1.79 | 3.46 | 10.60 |
| 0.60 | 2.02 | 2.04 | 3.74 | 16.33 |
| 0.70 | 2.81 | 2.25 | 4.65 | 19.60 |
| 0.80 | 3.17 | 2.93 | 6.70 | 42.11 |
| 0.90 | 3.79 | 3.96 | 10.52 | 76.79 |
| 0.95 | 7.39 | 6.09 | 12.86 | 122.76 |

*Note:* These amounts are the implied oxygen requirement for aerobic destruction of the compounds that comprise $BOD_5$.

As these systems incorporate active aeration into the wetland, it is possible to design wetlands that utilize aerobic degradation of BOD exclusively, commensurate with higher *k* rates. For instance, volumetric *k* rates for degradation of BOD in propylene glycol runoff (generated from aircraft deicing) has been demonstrated to be approximately 10–30 times higher in aerated HSSF wetlands when compared to nonaerated HSSF wetlands (Wallace *et al.*, 2007a). Design of aerated HSSF wetlands is discussed in more detail in Part II of this book.

### Seasonal Trends

There are typically gentle annual cycles in the effluent BOD from HSSF wetlands (Figure 8.29). The trend is described by:

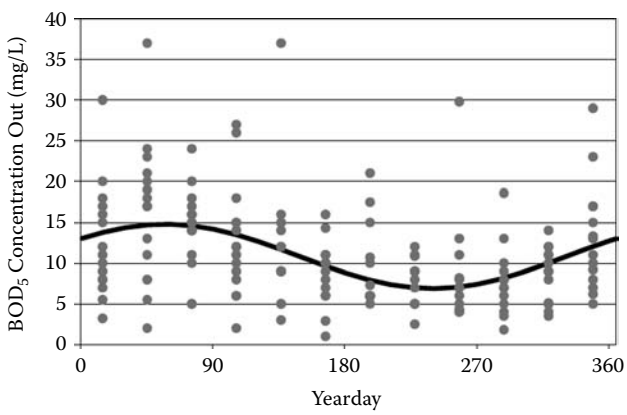



**FIGURE 8.29** The annual cycle in effluent $BOD_5$ for the Pocahontas, Arkansas, HSSF wetland. The period of record is 14 years. (Data from WERF database (2006) *Small-Scale Constructed Wetland Treatment Systems Database* (*Project-01 CTS-5;* Final Report by Wallace and Knight, 2006). Compiled by J. Nivala and R. Clarke. Water Environment Research Foundation (WERF): Alexandria, Virginia.)

$$C = C_{avg}\left[1 + A \cdot \cos\left(\omega(t - t_{max})\right)\right] \tag{8.29}$$

where

$A$ = trend fractional amplitude, dimensionless
$C$ = concentration, mg/L
$C_{avg}$ = mean annual concentration, mg/L
$t$ = yearday, d
$t_{max}$ = yearday for maximum concentration, d
$\omega$ = annual period, 0.01721 $d^{-1}$

The maximum may be at any time of the year (Table 8.12). The mean fractional amplitude is 35% of the mean.

### Variability around Seasonal Trends

The considerable scatter in effluent concentrations contributes to low $R^2$-values for the trend lines. This behavior is of concern in wetland sizing, if the peak values of the concentrations are of importance in the permit for the project. Because stochastic behavior is present in moderate amounts, it is necessary to quantify performance variability, and ultimately to modify sizing based upon that understanding. Therefore, excursion frequencies are shown in Table 8.13.

## Effects of Design and Operating Conditions

### Water Depth

Bed depth (water depth) is a design variable for HSSF wetlands. As the depth is increased, the root zone changes from occupying the entire depth to occupying only the upper portion of the water column. Rooting depths are variable, but in general, roots are observed to penetrate only about 30–40 cm into HSSF beds (see Chapter 3). Deep beds will, therefore, contain a zone under the roots in which there are neither profits from root chemical effects, nor penalties from root hydraulic

**TABLE 8.12**
**Sinusoidal Annual Trends in Effluent $BOD_5$ for HSSF Treatment Wetlands**

| Site | POR (years) | Frequency | Trend Mean (mg/L) | Trend Fractional Amplitude | Trend $t_{max}$ (Julian day) | Trend ($R^2$) |
|---|---|---|---|---|---|---|
| Cumbria, England | 3.5 | Weekly | 2.2 | 0.07 | 347 | 0.01 |
| Leicestershire 2, England | 4.5 | Weekly | 2.8 | 0.14 | 153 | 0.02 |
| Staffordshire 3, England | 3.9 | Weekly | 4.5 | 0.58 | 210 | 0.08 |
| Fish-Royer, Indiana | 2.1 | Monthly | 3.1 | 0.38 | 350 | 0.13 |
| Haughton, Louisiana | 2.0 | Monthly | 6.5 | 0.54 | 1 | 0.29 |
| Calahan, Colorado | 4.1 | Monthly | 10.4 | 0.80 | 61 | 0.25 |
| Pocahontas, Arkansas | 14.3 | Monthly | 10.8 | 0.36 | 58 | 0.15 |
| Waipoua, New Zealand | 3.6 | Monthly | 11.1 | 0.43 | 185 | 0.17 |
| Judsonia, Arkansas | 14.3 | Monthly | 11.5 | 0.28 | 64 | 0.17 |
| Clarendon, Arkansas | 14.3 | Monthly | 11.7 | 0.04 | 197 | 0.00 |
| Eudora, Arkansas | 14.3 | Monthly | 16.7 | 0.13 | 326 | 0.02 |
| Dierks, Arkansas | 5.0 | Monthly | 18.7 | 0.13 | 255 | 0.05 |
| Lewisville, Arkansas | 7.0 | Monthly | 20.1 | 0.32 | 292 | 0.10 |
| Monterrey, Virginia | 2.3 | Monthly | 20.6 | 0.20 | 215 | 0.22 |
| Fife, Scotland (Cell 3) | 1.7 | Monthly | 23.0 | 0.40 | 131 | 0.18 |
| Las Animas, Colorado | 4.0 | Monthly | 23.5 | 0.17 | 359 | 0.03 |
| Fife, Scotland (Cell 2) | 1.7 | Monthly | 24.8 | 0.39 | 176 | 0.10 |
| North Yorkshire 2, England | 6.5 | Monthly | 31.6 | 0.35 | 118 | 0.19 |
| Valleyfield 1, United Kingdom | 1.7 | Monthly | 37.8 | 0.54 | 122 | 0.30 |
| Fife, Scotland (Cell 1) | 1.7 | Monthly | 39.7 | 0.35 | 134 | 0.20 |
| North Yorkshire 1, England | 7.9 | Monthly | 55.4 | 0.22 | 287 | 0.61 |
| **Mean** | | | | **0.35** | | **0.17** |

*Note:* POR = period of record. All systems operate year-round. Site names for U.K. systems are approximate.

blockage. As in the case of FWS wetlands, increases in depth provide more detention time without adding area (footprint), but do not lower the hydraulic loading rate (see Chapter 6). The intuition of the designer is strongly influenced by the presumed form of the first-order model. If it is written for detention time, using a volumetric rate constant $k_V$ (Equation 8.21), then it seems logical that a deeper bed provides more detention and is therefore preferable. If it is written for hydraulic loading rate, using an areal rate constant $k$ (Equation 8.21), then the conclusion is invited that depth does not matter.

The issue may be somewhat elucidated by examining the results of two side-by-side studies of HSSF wetlands, both of which used a form of replication. The studies at Baxter, Tennessee, utilized 14 wetlands, 7 operated at 30 cm and 7 at 46 cm. These gravel cells were vegetated with bulrushes (*Scirpus validus*), and operated for three years, in two different modes (George *et al*., 1994; Kemp and George, 1997; George *et al*., 1998). In Mode 1, all were operated in parallel at different loading rates. In Mode 2, there was series and parallel operation, with recycle. Tracer testing showed approximately $N$TIS = 4, and inlet $BOD_5$ was 40 – 60 mg/L. The studies near Barcelona, Spain (García *et al*., 2004a), involved eight wetlands—six operated at 50-cm and two at 27-cm depth. Tracer tests showed approximately $N$TIS = 4, and inlet $BOD_5$ was 40 mg/L. Values of $k$ and $k_V$ were determined for $P$ = 3 and $C^*$ = 3 mg/L for both studies (Table 8.14). It is seen that both $k$ and $k_V$ are lower at deeper depth, meaning that deeper beds perform much more poorly than shallow. The effect is larger for $k_V$, which shows decreases of up to a factor of four. However, the areal $k$-values are also smaller at deeper depth. Thus, no matter which model is used, deeper beds are not as effective. It is apparently of no use to increase detention time by deepening the bed. These studies do not permit determination of a lower limit on bed depth, i.e., how shallow should the bed be. Coleman *et al.* (2001) also compared shallow (45 cm) and deep (60 cm) beds, and found no difference in performance at the same hydraulic loading rate, thus emphasizing that the extra depth provided no benefit.

## Media Size

The concept of the HSSF wetland as a horizontal trickling filter invites the viewpoint that microbial biofilms on the media are responsible for the reduction in BOD. Small size media have greater surface area, about 150 $m^2/m^3$ for 25-mm spheres, and 360 $m^2/m^3$ for 10-mm spheres. If biofilms do the work, and if they coat the media, then a factor of two improvement would be expected for the smaller media (Khatiwada and Polprasert, 1999a). However, the evidence that such a

**TABLE 8.13**
**Trend Multipliers for Effluent BOD Distributions in HSSF Wetlands**

| Site | Trend Mean (mg/L) | Trend Multiplier (80th percentile) | Trend Multiplier (90th percentile) | Trend Multiplier (95th percentile) | Trend Multiplier (99th percentile) |
|---|---|---|---|---|---|
| Cumbria, England | 2.2 | 1.26 | 1.85 | 2.80 | 6.96 |
| Leicestershire 2, England | 2.8 | 1.50 | 1.97 | 2.66 | 3.51 |
| Staffordshire 3, England | 4.5 | 1.21 | 1.30 | 1.38 | 1.52 |
| Fish-Royer, Indiana | 3.1 | 1.28 | 1.72 | 2.96 | 4.48 |
| Haughton, Louisiana | 6.5 | 1.46 | 1.75 | 1.92 | 2.84 |
| Calahan, Colorado | 10.4 | 1.57 | 1.87 | 2.14 | 2.28 |
| Pocahontas, Arkansas | 10.8 | 1.69 | 2.11 | 2.85 | 3.22 |
| Waipoua, New Zealand | 11.1 | 1.50 | 1.69 | 2.31 | 2.92 |
| Judsonia, Arkansas | 11.5 | 1.35 | 1.77 | 1.85 | 2.64 |
| Clarendon, Arkansas | 11.7 | 1.35 | 1.57 | 1.84 | 2.73 |
| Eudora, Arkansas | 16.7 | 1.47 | 1.97 | 2.19 | 2.53 |
| Dierks, Arkansas | 18.7 | 1.54 | 2.07 | 2.46 | 2.67 |
| Lewisville, Arkansas | 20.1 | 1.65 | 1.80 | 2.07 | 3.47 |
| Monterrey, Virginia | 20.6 | 1.49 | 2.14 | 2.66 | 3.46 |
| Fife, Scotland (cell 3) | 23.0 | 1.37 | 1.56 | 1.74 | 1.86 |
| Las Animas, Colorado | 23.5 | 1.67 | 1.95 | 2.50 | 3.40 |
| Fife, Scotland (cell 2) | 24.8 | 1.26 | 1.63 | 1.94 | 3.10 |
| North Yorkshire 2, England | 31.6 | 1.30 | 1.60 | 2.11 | 3.50 |
| Fife, Scotland (cell 1) | 37.8 | 1.44 | 1.71 | 2.10 | 2.49 |
| Fife, Scotland (cell 4) | 39.7 | 1.34 | 1.86 | 2.54 | 3.78 |
| North Yorkshire 1, England | 55.4 | 1.58 | 1.69 | 1.79 | 1.90 |
| **Mean** | | **1.45** | **1.78** | **2.18** | **2.88** |
| **SD** | | **0.14** | **0.21** | **0.41** | **0.73** |

*Note:* For illustration, a trend multiplier of 1.58 for the 80th percentile for North Yorkshire 1 means that one time out of five, the effluent $BOD_5$ will be more than 58% higher than the trend value at that time of year. On average, the outlet $BOD_5$ at North Yorkshire 1 is 55.4 mg/L, and hence one sample in five may exceed 88 mg/L. Trend multiplier is $(1 + \Psi)$; see Equation 6.61. Site names for U.K. systems are approximate.

marked effect of media size is sparse, and thus the simplistic trickling filter concept should not be carried too far.

García *et al.* (2004a) performed side-by-side analyses of two media sizes (3.5 and 10 mm) in eight wetlands, using different paired depths and aspect ratios. The *k*-values display no marked effect of media size for any of the pairs, with no consistent pattern of the fine media doing better or worse. Conversely, He and Mankin (2002) found that fine media (19 mm) outperformed coarse media (38 mm). The *k*-values for *Typha* systems were 116 and 68 m/yr, and for unplanted systems were 108 and 52 m/yr for small and large media, respectively.

The selection of media size is also controlled by the need for adequate hydraulic conductivity, and by the clogging characteristics of the bed and water to be treated. Beds of sand or soil typically cannot carry the water in the subsurface mode, and then it is a moot point as to whether the finer particle size is better or not.

## Vegetation

Plants provide a number of useful functions in treatment wetlands, including the possibility of oxygen release from roots

**TABLE 8.14**
**Effect of Depth on Rate Constants for $BOD_5$ in HSSF Wetlands (see text for sources)**

| | Baxter Mode 1 | | Baxter Mode 2 | | Barcelona | |
|---|---|---|---|---|---|---|
| | Shallow | Deep | Shallow | Deep | Shallow | Deep |
| Depth (cm) | 30 | 46 | 30 | 46 | 27 | 50 |
| $k$ (m/yr) | 45.0 | 23.3 | 18.5 | 16.79 | 25.0 | 11.7 |
| $k_V$ (1/d) | 1.027 | 0.347 | 0.422 | 0.256 | 0.635 | 0.160 |

and an increase in the sites available for bacteria (Brix, 1994b). The evidence suggests that root oxygen release is small (see Chapter 5), but other plant functions are potential contributors to improved BOD removal.

Plants are the most visible attribute of treatment wetlands, and choice of plants represents one of the few design decisions to be made. Personal preference may reflect a number of aesthetic factors. Therefore, it is not surprising that many studies have attempted to determine which plants may provide better treatment. Brisson *et al.* (2006) reviewed 27 experimental studies of which 16 assessed BOD or COD performance. They concluded: "Most studies comparing planted versus nonplanted subsurface flow constructed wetland systems for wastewater treatment show a significant and positive effect of macrophytes on pollutant removal." However, the Brisson *et al.* (2006) review utilized removal percentage as the metric, thus allowing differences in loading or detention time to cloud the issue.

One way to remove bias is via comparison of loadings and load removals, as was done by Tanner (2001b). When examined in that way, it was found that there was a slight improvement in concentration reduction for both COD and BOD—about 2–5 mg/L. The first-order areal *k*-values also remove that potential bias, and these have been computed for BOD for a sampling of side-by-side studies in Table 8.15. For simplicity, only results for *Typha* and *Schoenoplectus* are shown. Although on average the presence of plants is beneficial, that result is not unfailingly true, even in the small sample in Table 8.15. The type of plant appears to have some effect, and the Brisson *et al.* (2006) conclusion—that we should pay more attention to plant effects—seems warranted.

The European choice is most often *Phragmites*, based upon the presumption that this is the "best" plant. However, that presumption remains to be rigorously tested. Indeed, studies like that at Santee, California, Gersberg *et al.* (1986) found that bulrushes (*Schoenoplectus* spp.) were clearly superior to *Phragmites*. Theis and Young (2000) found no evidence that *Phragmites* was superior to *Schoenoplectus* in side-by-side testing.

## Aspect Ratio

The Barcelona studies of García *et al.* (2004a) found no effect of aspect ratio on performance, within the range 1:1 < L:W < 2.5:1; but that range of aspect ratio is quite small. Bounds *et al.* (1998) studied three aspect ratios of L:W = 4, 10, and 30, utilizing septic tank effluent. The authors found no significant difference in percent removals due to aspect ratio, but removals were all quite high, and thus not sensitive to differences in outlet concentrations. A difference becomes apparent in terms of rate constants calculated from the Bounds *et al.* (1998) data: a 20% improvement for L:W = 30 compared to L:W = 4 during the first year, and 44% improvement during the second year. The improvements were also reflected in longitudinal profiles that decreased along the flow direction, and did not reach a plateau for any aspect ratio, but were increasingly steep as aspect ratio increased. These fractional improvements in *k*-values are commensurate with those forecast

**TABLE 8.15**
**Comparison of First-Order Areal *k*-Values for BOD Removal for Various Plantings in SSF Wetlands**

| Source | Cell | *Typha* spp. | *Schoenoplectus* spp. | Gravel Only |
|---|---|---|---|---|
| Tanner *et al.* (1995b) | 1 | — | 24 | 16 |
| Tanner *et al.* (1995b) | 2 | — | 19 | 21 |
| Tanner *et al.*(1995b) | 4 | — | 19 | 18 |
| Regmi *et al.* (2003) | — | — | 23 | 18 |
| Theis and Young (2000) | — | — | 16 | 23 |
| Bavor *et al.* (1988) | — | 61 | 89 | 98 |
| Stein *et al.* (2006a) | — | 77 | 79 | 64 |
| Gersberg *et al.* (1986) | — | 31 | 117 | 26 |
| Heritage *et al.* (1995) | — | 89 | 73 | 84 |
| Coleman *et al.* (2001) | — | 40 | 32 | 31 |
| He and Mankin (2002) | FL | 116 | — | 108 |
| He and Mankin (2002) | CL | 68 | — | 52 |
| He and Mankin (2002) | R | 124 | — | 119 |
| **Averages** | | | | |
| ***Typha*/Gravel** | | **76** | — | **73** |
| ***Schoenoplectus*/Gravel** | | — | **46** | **38** |

*Note:* The model parameters used were $P = 3$ and $C^* = 3$, and period of record data was fit. The Theis and Young data are for COD, and the He and Mankin data are for COD according to a plug flow model.

for increases in *P*-values from about 3, up to about 10 for the highest L:W ratio. Therefore, the experimental results are in agreement with the improvements forecast from improved internal hydraulics.

## 8.4 BOD REMOVAL IN VF WETLANDS

Vertical flow (VF) wetlands are one of the newest forms of constructed wetlands. Typically, these systems have been dimensioned based on an empirical basis; a specified unit area ($m^2$) for a given organic loading (typically expressed as population equivalents, or PE). The data analysis presented in this chapter is restricted to the pulse-loaded, unsaturated downflow VF wetlands typically implemented in Europe. Other technology variants, such as tidal flow wetlands, are discussed in more detail in Part II of this book.

The operational regime of pulse-loading, followed by rest periods, allows VF systems to operate in a mode of unsaturated flow, which allows the introduction of air (and oxygen) into the VF bed (Platzer and Mauch, 1997; Cooper, 1999). Consequently, VF wetlands are more amenable to aerobic reduction of organic matter and associated BOD than HSSF wetlands.

VF wetlands have been in widespread use for over ten years now, and an increasing amount of BOD data is available to characterize the performance of these systems. This data set is largely restricted to hydraulic loadings and simple input–output relations. VF wetlands are typically operated under load-and-rest regimes, which affect system performance. At present, data on temperature and energy fluxes in VF wetlands is limited, and very few VF wetlands have been tracer tested to produce *N*TIS values. Due to these data limitations, the current state of the art cannot characterize VF wetland performance to the same extent as FWS and HSSF wetlands.

### Graphical Relationships

The graphical display has often been adopted in the literature to characterize treatment wetland performance (Kadlec and Knight, 1996; U.S. EPA, 2000a; Wallace and Knight, 2006). Two relationships are of potential use: the concentration in–concentration out ($C_i$–$C_o$) graph and the concentration out–BOD loading ($C_o$–BLI) graph.

Figure 8.30 illustrates the $C_i$–$C_o$ relationship for VF wetlands, based on 110 system-years of performance data. As seen in Figure 8.30, effluent concentrations ($C_o$) are only weakly dependent on influent concentrations ($C_i$), but when this data is redacted to the lowest 5% of effluent concentrations; the estimated background concentration ($C^*$) is 2.0 mg/L across all influent concentration ranges. A concentration of 2 mg/L is essentially the method detection limit for the BOD test, indicating that VF wetlands can be highly effective in BOD reduction.

The concentration out–load in ($C_o$–BLI) graph (Figure 8.31) further illustrates that treatment performance of VF wetlands is more dependent on the influent BLI. As seen in Figure 8.31,

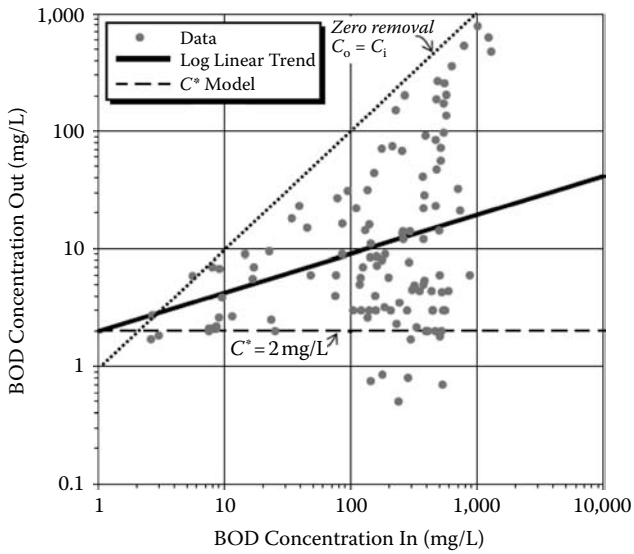



**FIGURE 8.30** BOD input–output concentration graph for 62 VF wetlands. Each data point represents one system-year of performance (110 system-years total). The log-linear central trend is $\log_{10}(C_o) = 0.33 \log_{10}(C_i) + 0.3$, $R^2 = 0.14$. The lower bound line, excluding approximately 5% of the lowest values, is $C^* = 2.0$ mg/L.

effluent concentrations ($C_o$) reflect a dependency of BOD influent loadings across different influent concentration ranges. The log-linear central tendency can be expressed by:

$$\log_{10}(C_o) = 0.3 + 0.71\left[\log_{10}(\text{BLI})\right] \qquad (8.30)$$
$$R^2 = 0.49$$

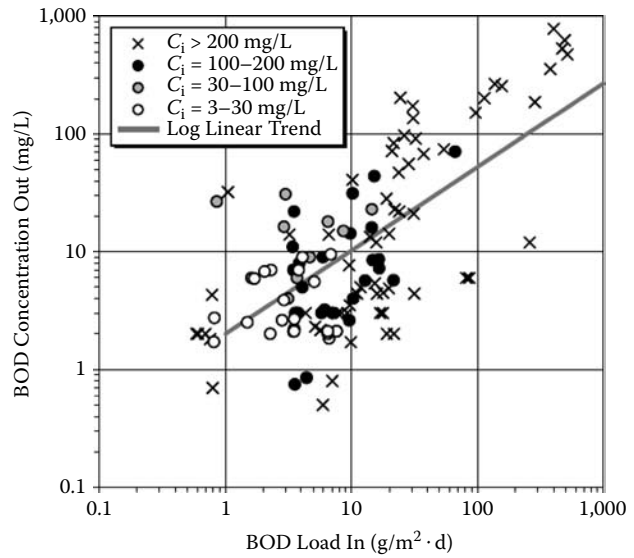



**FIGURE 8.31** BOD Load In (BLI) versus BOD Concentration Out for 62 VF wetlands. Each data point is represented by one system-year of data (110 system-years total). The log-linear central trend is represented by $\log_{10}(C_o) = 0.71 \log_{10}(\text{BLI}) + 0.3$; $R^2 = 0.49$.

where

$C_o$ = effluent BOD concentration, mg/L

BLI = BOD loading in, g/m$^2$·d

## First-Order Modeling

In general terms, the *P*-*k*-*C** first-order model (Equation 8.20) can be used to characterize BOD removal in VF wetlands. The parameter *P* will always be less than the *N*TIS value due to weathering and speciation of the BOD and COD mixtures as treatment proceeds. These effects have been previously discussed in this chapter for FWS and HSSF wetlands.

A major limitation of VF wetlands is that data on *N*TIS, as determined by tracer studies, is not widely available. Data from other types of VF wetlands can be extrapolated to broadly estimate *N*TIS. For saturated upflow VF mesocosms, *N*TIS has been observed to be approximately 2 (Tanner *et al*., 2002a), and for aerated saturated downflow mesocosms, *N*TIS has been observed to be approximately 1, although a value of 2 has been proposed for full-scale systems (Wallace *et al*., 2006a). At the time of this writing, a value of $P = 2$ is postulated to represent system performance in unsaturated downflow VF wetlands. This *P*-value will likely change as additional data becomes available.

Based on $P = 2$ and $C^* = 2.0$ mg/L, areal *k*-values can be estimated on input–output data averaged over a time period sufficiently long enough to minimize seasonal changes and stochastic events. Data from 62 VF wetlands, representing 110 system-years, is summarized in Table 8.16.

By examining multiple data sets which provide a distribution of *k*-values, effects of BOD removal mechanisms can be examined. At low influent loadings and inlet concentrations, degradation of BOD is dominated by aerobic mechanisms. As loadings increase, aerobic processes become less probable, and anaerobic mechanisms are more likely to occur. Finally, BOD is a lumped parameter, and includes both soluble and particulate forms. If organic matter can be segregated on the surface of the wetland bed (see Figure 7.31), this will also contribute to BOD removal at high influent concentrations.

### Removal as a Function of Bed Depth

Detailed transects of BOD removal as a function of bed depth are not currently available for a variety of VF wetlands. However, it has been demonstrated that the majority of microbial biomass is located in the top 20 cm of the VF bed (Langergraber *et al*., 2006b); therefore, it is highly likely that organics removal occurs preferentially in this upper region, due to filtration of particulate organic matter, greater availability of oxygen, and greater microbial biomass.

### Loading Effect on *k*-Values

The plateau effect of the background concentration (*C**), affects the apparent *k*-values observed from input–output relationships. For VF wetlands, the area used to calculate *k* is normal to the flow direction; so, no underestimate of *k* is induced by the wetland area as the entire bed area is used for reduction of organics and BOD. However, due to the load-and-rest operational regime employed with most VF wetlands, these systems can accept very high hydraulic loading rates if loaded at influent concentrations close to *C**, leading to high estimates of *k* (Figure 8.32). Unlimited application of this observation leads to a design paradox; *k* becomes higher

**TABLE 8.16**
**First-Order Areal Rate Constants for VF Wetlands, Based on System-Years of Performance Data**

| | Estimated *k*-Rates (m/yr) | | | | |
|---|---|---|---|---|---|
| | $C_i$ (3–30 mg/L) | $C_i$ (30–100 mg/L) | $C_i$ (100–200 mg/L) | $C_i$ (>200 mg/L) | Overall |
| *C** (mg/L) | 2.0 | 2.0 | 2.0 | 2.0 | 2.0 |
| *P* | 2 | 2 | 2 | 2 | 2 |
| N | 21 | 9 | 22 | 58 | 110 |
| **Percentile** | | | | | |
| 0.05 | –6 | 11 | 52 | 31 | 19 |
| 0.1 | 21 | 17 | 62 | 41 | 34 |
| 0.2 | 77 | 32 | 73 | 47 | 51 |
| 0.3 | 160 | 50 | 110 | 71 | 78 |
| 0.4 | 210 | 57 | 153 | 101 | 113 |
| 0.5 | 471 | 73 | 187 | 143 | 146 |
| 0.6 | 751 | 91 | 253 | 160 | 191 |
| 0.7 | 1,025 | 101 | 280 | 246 | 275 |
| 0.8 | 1,756 | 113 | 343 | 316 | 408 |
| 0.9 | 2,234 | 116 | 392 | 554 | 1,006 |
| 0.95 | 2,402 | 130 | 473 | 1,165 | 1,694 |

*Note:* The number of data points in each concentration category is represented by N.

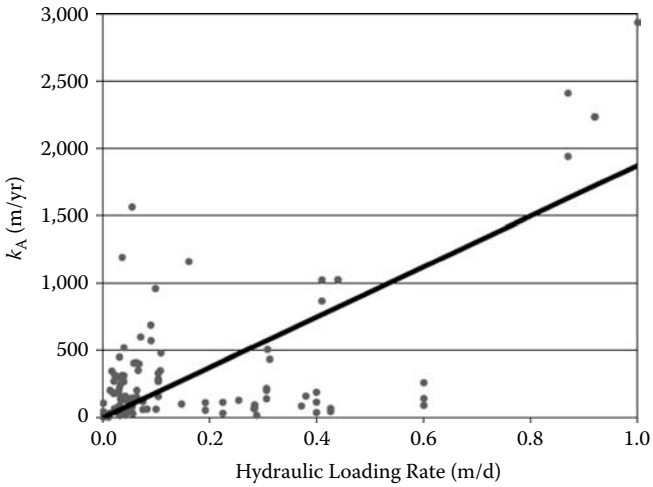



**FIGURE 8.32** Dependence of the first-order areal rate constant, $k_A$, on hydraulic loading rate. The values $P = 2$ and $C^* = 2$ mg/L have been used. The trend line has $R^2 = 0.64$, which is obscured by a large scatter in the data, and is strongly dependent on the four datapoints at the right side of the abscissa.

with increasing loading rates, and repeated applications of this observation leads to the iterative effect of $k = \infty$, with a corresponding wetland area of zero. This is, of course, useless as a design methodology. Design of VF wetlands is addressed in more detail in Part II of this book.

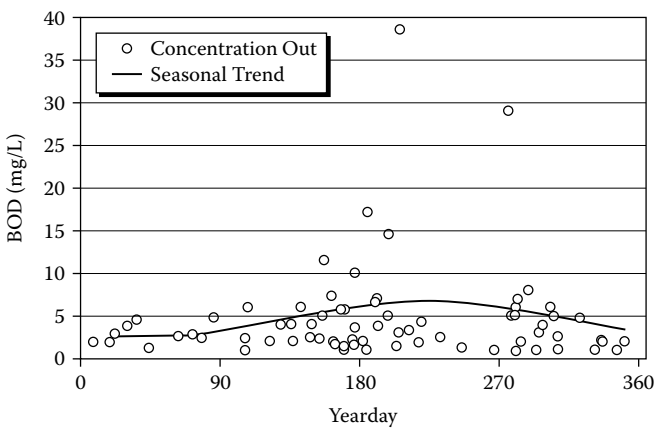



**FIGURE 8.33** Example of seasonal variability in effluent concentrations; Buckinghamshire, England. Three years of performance data are represented. The mean trend is 4.6 mg/L; the fractional amplitude is 0.46, $t_{max}$ is at 223 days, and $R^2 = 0.66$ (Data from CWA database (2006) *Constructed Wetlands Interactive Database, Version 9.02*. Compiled by G.D. Job and P.F. Cooper. United Kingdom Constructed Wetland Association (CWA): Gloucestershire, United Kingdom.)

### Seasonal Effects

There are typically weak annual cycles in the effluent BOD from VF wetlands (Figure 8.33). The seasonal trend model is described in Equation 8.29. The maximum may be at any time of the year. Based on the limited data currently available, the mean fractional amplitude is approximately 35%, indicating that VF wetlands likely have seasonal changes comparable to HSSF wetlands (see Table 8.11).

At present, there is insufficient data to assess stochastic variability and temperature effects (θ-factors) for VF wetlands.

## SUMMARY

Wetlands are effective in the reduction of $BOD_5$, as long as incoming $BOD_5$ exceeds the natural level at which the wetland operates. A wealth of carbon conversion processes operate in wetlands, some of which consume $BOD_5$, and others produce it. Both anaerobic and aerobic processes have been measured to consume carbon compounds in the wetland environment. Litter and sediment decomposition produce soluble carbon compounds. Consequently, the simplest mass balance model must include both consumption and generation of these substances.

Profiles of $BOD_5$ along the flow direction may be described by the *P-k-C** model. However, there are wide ranges of parameter values; so, no universal recommended values of *P*, *k*, and *C** can be developed. The background $BOD_5$ depends somewhat on season of the year. Neither the rate constant nor the background $BOD_5$ depends strongly on temperature. Because $BOD_5$ is a mixture, and subject to weathering, rate constants change during the removal process. This leads to *P*-values that are considerably less than determined from inert tracer testing. Variations in these parameters with temperature cannot be quantified adequately from the existing database, and consequently all variability must be absorbed into the performance spectrum.

Most operating FWS wetlands are overdesigned for $BOD_5$ removal, and hence effluent concentrations are at or near background levels, in the 1–10 mg/L range.

HSSF wetlands inherently oxygen-transfer limited systems; therefore, there is presumptively a shift in degradation processes (from aerobic to anaerobic) and the influent mass load increases, although this relationship is obscured by the considerable scatter in the available HSSF data set.

VF wetlands (operating on pulse-load, unsaturated flow principles) are considerably more effective in degradation of organic matter (including BOD) as the load-and-rest operating protocol for these wetlands allows the introduction of atmospheric oxygen directly into the wetland bed.

All types of treatment wetlands have seasonally variable changes in BOD effluent quality. These seasonal changes are driven by climate, plant biomass cycling, and water temperatures.

# 9 Nitrogen

Nitrogen compounds are among the principal constituents of concern in wastewater because of their role in eutrophication, their effect on the oxygen content of receiving waters, and their toxicity to aquatic invertebrate and vertebrate species. These compounds also augment plant growth, which in turn stimulates the biogeochemical cycles of the wetland. The wetland nitrogen cycle is very complex, and control of even the most basic chemical transformations of this element is a challenge in ecological engineering. This chapter describes the wetland nitrogen cycle, summarizes current knowledge about environmental factors that control nitrogen transformations, and provides alternative approaches that can be used to design wetland treatment systems to treat nitrogen.

## 9.1 NITROGEN FORMS IN WETLAND WATERS

The most important inorganic forms of nitrogen in wetlands treating municipal or domestic wastewater are ammonia ($NH_4^+$), nitrite ($NO_2^-$), nitrate ($NO_3^-$), nitrous oxide ($N_2O$), and dissolved elemental nitrogen or dinitrogen gas ($N_2$). Nitrogen is also invariably present in FWS wetlands in organic forms. Both dissolved and particulate forms may be present, but in most cases there is little particulate nitrogen in settled wetland surface waters.

Common analytical methods include procedures for determination of total or dissolved forms (APHA, 2005). These include

- Nitrate
- Nitrite
- Ammonia
- Total Kjeldahl nitrogen (TKN) = (organic + ammonia nitrogen)

From these basic measures, several derived concentrations may be computed:

- Oxidized nitrogen = nitrate + nitrite
- Inorganic nitrogen = oxidized nitrogen + ammonia
- Organic nitrogen = TKN − ammonia
- Total nitrogen = TKN + oxidized nitrogen

Each category can be the subject of wetland effluent quality regulation, and each may represent an important feature of wetland water quality, depending upon the nature of source waters.

As treatment wetland technology develops, nondomestic source waters are of increasing interest, thus bringing attention to other nitrogen compounds. Examples include

- Polymer industry wastewaters, which contain amines ($RNH_2$, where $R$ is an aliphatic hydrocarbon) (Beeman and Reitberger, 2003)
- Potato wastewaters, which contain imides ($RCO–NH–OCR'$, where $R$ and $R'$ are aliphatic hydrocarbons) (Kadlec *et al*., 1997)
- Aluminum and gold processing waste leachates, which contain cyanide ($CN^-$) (Bishay and Kadlec, 2005; Gessner *et al*., 2005)
- Chlorinated effluents, which develop chloramines in the wetland ($NH_xCl_y^-$) (Zheng *et al*., 2004)
- Triazine pesticides in agricultural runoff (e.g., atrazine, $C_8H_{13}N_5Cl$) (Moore *et al*., 2000b)

These and other specialty applications of interest are discussed in Chapters 13 and 25.

### Organic Nitrogen

Organic nitrogen is made up of a variety of compounds including amino acids, urea and uric acid, and purines and pyrimidines. Amino acids are the main components of proteins, which are a group of complex organic compounds essential to all forms of life. Amino acids consist of an amine group ($–NH_2$) and an acid group (–COOH) attached to the terminal carbon atom of a variety of straight carbon chain and aromatic organic compounds. Organic forms of nitrogen, primarily as amino acids, typically makes up from 1–7% of the dry weight of plants and animals.

Urea ($CNH_4O$) and uric acid ($C_4N_4H_4O_3$) are among the simplest forms of organic nitrogen in aquatic systems. Urea is formed by mammals as a physiological mechanism to dispose of ammonia that results when amino acids are used for energy production. Because ammonia is toxic, it must be converted to a less toxic form, urea, by the addition of carbon dioxide. Uric acid is produced by insects and birds for the same purpose. These organic forms of nitrogen are important in wetland treatment because they are readily hydrolyzed, chemically or microbially, resulting in the release of ammonia.

Pyrimidines and purines are heterocyclic organic compounds in which nitrogen replaces two or more of the carbon atoms in the aromatic ring. Pyrimidines consist of a single heterocyclic ring, and purines contain two interconnected rings. These compounds are synthesized from amino acids to become the main building blocks of the nucleotides that make up DNA in living organisms.

Wastewaters contain varying amounts of organic nitrogen, depending upon the source. Nitrogen in domestic sewage comprises about 60% ammonia and 40% organic

nitrogen (U.S. EPA, 1993b). Activated sludge treatment processes typically reduce this fraction considerably, but facultative lagoon effluents may retain the same proportions while reducing total nitrogen (TN). Food processing effluents may contain very high amounts of organic nitrogen.

### Ammonia

Ammonia exists in water solution as either as un-ionized ammonia ($NH_3$) or ionized ammonia ($NH_4^+$, ammonium ion), depending on water temperature and pH:

$$NH_3 + H_2O \rightleftharpoons NH_4^+ + OH^- \tag{9.1}$$

Total ammonia is equal to the sum of the un-ionized and the ionized ammonia, and is designated as *ammonia nitrogen* in this book. The fraction of un-ionized ammonia in water may be estimated from equilibrium conditions, given by

$$\log_{10} K_d = \log_{10} \frac{C_{IA}}{C_{UA}} = 0.09018 - \left( \frac{272{,}992}{T + 273.16} \right) - \text{pH} \tag{9.2}$$

where

$C_{IA}$ = ionized ammonia concentration, mg/L
$C_{UA}$ = unionized ammonia concentration, mg/L
$K_d$ = dissociation constant, dimensionless
$T$ = water temperature, °C

The ionized form is predominant in most wetland systems because of moderate pH and temperature, and is designated as ammonium nitrogen in this book. For a typical "average" environmental condition of 25°C and a pH of 7, un-ionized ammonia is only 0.6% of the total ammonia present. At a pH of 9.5 and a temperature of 30°C, the percentage of total ammonia present in the un-ionized form increases to 72%. At lower pH and temperature values, this percentage decreases significantly and presumably from wetlands under high pH and temperature conditions. Un-ionized ammonia is toxic to fish and other forms of aquatic life at low concentrations typically at concentrations >0.2 mg/L. U.S. EPA promulgates acute and chronic criteria for toxicity, and the reader is encouraged to consult the latest publication of such limits. Wetlands are useful for modulation of un-ionized ammonia, because they create circumneutral pH, and may lower water temperatures for warm effluents (Kadlec and Pries, 2004).

Ammonia typically comprises more than half of the TN in a variety of municipal and domestic effluents, where concentrations often are in the range of 20–60 mg/L. However, ammonia concentrations in food processing wastewaters treated in wetlands can exceed 100 mg/L (Van Oostrom and Cooper, 1990; Kadlec *et al*., 1997). Landfill leachates, particularly from recently closed and capped landfills, can contain hundreds of mg/L (Bulc *et al*., 1997; McBean and Rovers, 1999; Kadlec, 2003c).

Because ammonia is one of the principal forms of nitrogen found in many wastewaters and because of its potential role in degrading the environmental condition of wetlands and other receiving waters, reducing ammonia concentration drives the design process for many wetland treatment systems.

### Oxidized Nitrogen

Nitrite ($NO_2^-$) is an intermediate oxidation state of nitrogen (oxidation state of +3) between ammonia (–3) and nitrate (+5). Because of this intermediate energetic condition, nitrite is not chemically stable in most wetlands and is generally found only at very low concentrations. Nitrate ($NO_3^-$) is the most highly oxidized form of nitrogen (oxidation state of +5) found in wetlands. Because of this oxidation state, nitrate is chemically stable and would persist unchanged if not for several energy-consuming biological nitrogen transformation processes that occur. Nitrate can serve as an essential nutrient for plant growth, but in excess, it leads to eutrophication of surface water. Nitrate and nitrite are also important in water quality control because they are potentially toxic to infants (they result in a potentially fatal condition known as methylglobanemia) when present in drinking waters derived from polluted surface or groundwater supplies. The current regulatory criteria for nitrate in groundwater and drinking water supplies in the United States is 10 mg/L.

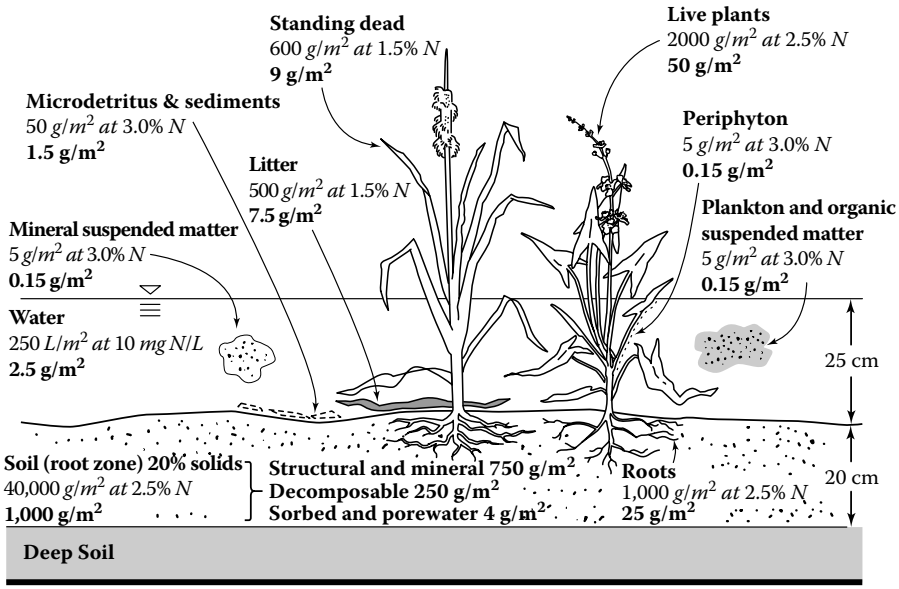
Oxidized nitrogen is typically near zero in sewage and in secondarily treated effluents, including secondary activated sludge and facultative lagoon waters. However, nitrate may seasonally be the dominant form in nitrified secondary effluents. It is present in agricultural runoff due to the oxidation of ammonia fertilizers in the vadose zone of farm fields, and may reach 40 mg/L in some cases.

## 9.2 WETLAND NITROGEN STORAGES

Organic nitrogen compounds are a significant fraction of the dry weight of wetland plants, detritus, microbes, wildlife, and soils. The mass of these nitrogen storages varies in different wetland types. A general idea of the sizes of these different storage compartments is necessary to understand the nitrogen fluxes discussed in this chapter (Figure 9.1).

### Soils and Sediments

The total of newly accreted organic materials at the Sacramento, California, FWS site had about 1.5% nitrogen (Nolte and Associates, 1998b). At the Houghton Lake, Michigan, and WCA2A, Florida, FWS sites, the organic sediments and soils averaged 3.13 ± 0.26 and 2.97 ± 0.37% nitrogen by dry weight, respectively. At both these sites, there was essentially no vertical profile in mass nitrogen percentage, but there was an increase in soil bulk density with depth for both. As a result, the volumetric storage of nitrogen increased with depth (Figure 9.2). The resulting




Note: Dry mass is in *italics* and standing stock is in **bold**.

**FIGURE 9.1** Nitrogen storages in a densely vegetated hypothetical FWS treatment wetland. Note that most of the stored nitrogen is in soils and sediments (≈1,000 gN/m²), second most is in plant materials (≈100 gN/m²), and least is in mobile forms in the water column (≈5 gN/m²).

nitrogen storage is about 500–2,000 gN/m² in the upper 30 cm of organic wetland sediments. For instance, the data of Figure 9.2 indicate approximately 700–800 gN/m² for Houghton Lake and WCA2A, respectively.

It is not common for the new sediments and soils in a treatment wetland to be inorganic in character. However, systems treating runoff may receive considerable quantities of inorganic solids from soil erosion in the watershed, which then combine with organic materials generated within the wetland. An example is Chiricahueto marsh in Mexico (Soto-Jiménez *et al*., 2003). Agricultural runoff brought water at about 15 mg/L of TN to the marsh for over 50 years. The soil column is now mostly inorganic, with less than 5% carbon (Figure 9.3). Mineral matter typically has a low nitrogen content, and consequently the nitrogen percentages were low, less than 0.4% dry weight. Both carbon and nitrogen decreased together as depth increased, indicating that most of the soil nitrogen was associated with the organic content. The nitrogen content of the upper 30 cm at Chiricahueto was 330 gN/m².

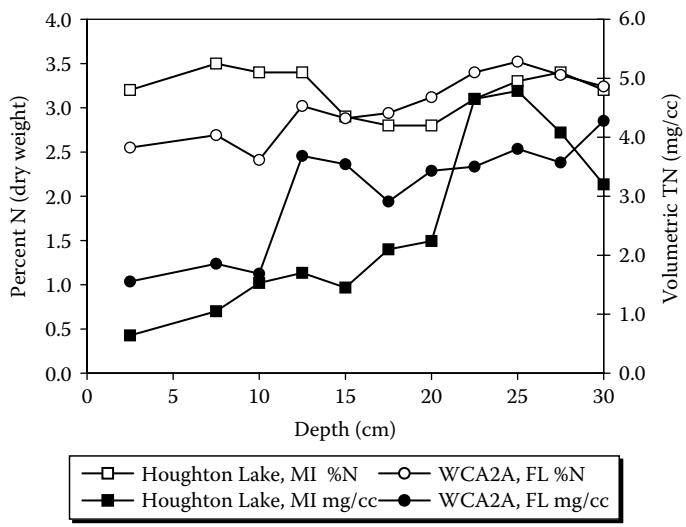



**FIGURE 9.2** Vertical variation in mass and volume concentrations soil of nitrogen in two FWS treatment wetlands. Houghton Lake, Michigan, data were acquired beneath waters at about 10 mg/L TN after nine years' exposure, and WCA2A, Florida, data were acquired at a site with pore water ammonia of 1.5–3.5 mg/L, and surface water of about 2.4 mg/L total nitrogen, after about 20 years' exposure. (Data for Houghton Lake: unpublished data; data for WCA2A: unpublished data; and Reddy *et al*. (1991) *Physico-Chemical Properties of Soils in the Water Conservation Area 2 of the Everglades*. Report to the South Florida Water Management District, West Palm Beach, Florida.)

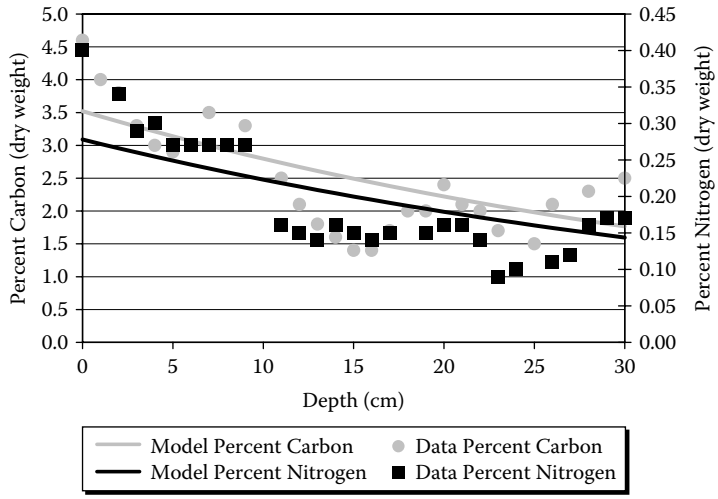



**FIGURE 9.3** The decline of carbon and nitrogen with depth in a FWS wetland receiving agricultural runoff, at Chiricahueto, Mexico. (Data from Soto-Jiménez *et al*. (2003) *Water Research*, 37: 719–728.)

## Biomass

The TN content of living biomass in marsh wetlands varies considerably among species, among plant parts, and among wetland sites. There is little variation from location to location within a homogeneous stand (Boyd, 1978). Example ranges of dry weight nitrogen percentages in natural wetlands are: 0.9–2.6% for emergent plants; 1.96–3.8% for floating leaved plants; and 2.4–2.9% for submersed plants (Boyd, 1978).

Treatment wetlands are often nutrient-enriched and display higher values of tissue nutrient concentrations than natural wetlands. For instance, live cattail leaves in the discharge area of the Houghton Lake, Michigan, FWS wetland averaged 2.0% N; those in nutrient-poor control areas averaged 1.1% N; dead leaves showed 1.6 versus 0.7% N, and litter leaves showed 3.6 versus 1.5% N, respectively (Table 9.1). Total biomass is enhanced by fertilization with effluent, and this compounds the effect of increased nutrient content, to produce large

**TABLE 9.1**
**Nitrogen Content ($gN/m^2$) of Vegetation in Treatment and Natural Areas at the Houghton Lake, Michigan, Treatment Wetland Site**

| | Control (DIN ≤ 0.1 mg/L) | | | Discharge (DIN ≈ 15 mg/L) | | |
|---|---|---|---|---|---|---|
| | Biomass ($g/m^2$) | Content (%) | Crop ($gN/m^2$) | Biomass ($g/m^2$) | Content (%) | Crop ($gN/m^2$) |
| **Live** | | | | | | |
| 1995 | 368 | 1.08 | 4.0 | 1,086 | 1.98 | 18.9 |
| 1996 | 773 | 1.08 | 8.2 | 1,323 | 2.37 | 30.3 |
| 1997 | 504 | 1.00 | 5.1 | 1,200 | 2.11 | 25.4 |
| 1998 | 311 | 1.11 | 3.4 | 1,333 | 1.65 | 22.5 |
| *4-year mean* | *489* | *1.07* | *5.2* | *1,235* | *2.03* | *24.3* |
| **Standing Dead** | | | | | | |
| 1995 | 642 | 0.69 | 4.8 | 917 | 1.07 | 10.1 |
| 1996 | 390 | 0.58 | 2.3 | 392 | 1.54 | 5.8 |
| 1997 | 190 | 0.77 | 1.4 | 1,642 | 2.02 | 32.6 |
| 1998 | 401 | 0.61 | 2.1 | 1,336 | 1.59 | 22.3 |
| *4-year mean* | *406* | *0.66* | *2.7* | *1,072* | *1.55* | *17.7* |
| **Litter** | | | | | | |
| 1996 | 84 | 1.60 | 1.4 | 1,769 | 3.60 | 62.2 |
| 1997 | 42 | 1.75 | 0.8 | 2,193 | 3.63 | 79.3 |
| 1998 | 135 | 1.75 | 2.3 | 3,090 | 3.55 | 111.1 |
| *3-year mean* | *87* | *1.70* | *1.5* | *2,351* | *3.59* | *84.2* |
| **Total Above** | **982** | **1.08** | **9.4** | **4,658** | **2.37** | **126** |

*Note:* DIN = dissolved inorganic nitrogen = oxidized plus ammonia nitrogen.

*Source:* Unpublished data.

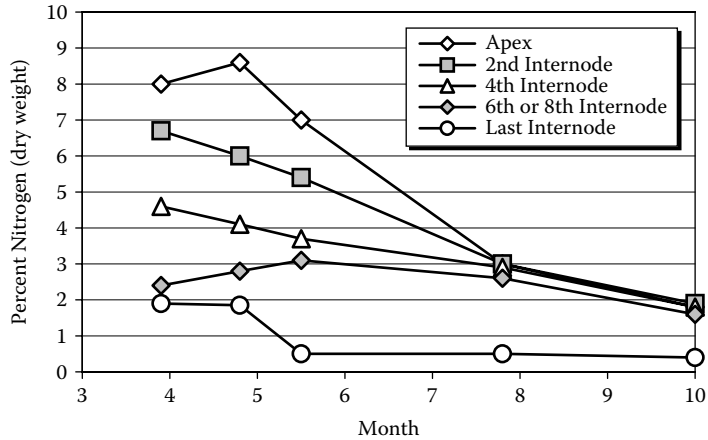



**FIGURE 9.4** Nitrogen content in *Phragmites australis* as a function of season and position aboveground. The site was a highly productive reed stand, which generated 1,500 g/m$^2$ of biomass over the June–August period. Redrawn from the data of Kühl and Kohl (1993). (Graph from Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

storages in treatment areas compared to unfertilized natural wetlands.

Different plant parts may show large differences in nitrogen content, and the seasonal variability may be very large. The extent of this variability is shown in Figure 9.4 for *Phragmites australis*, for a reed stand in the margin of Templiner See, a heavily loaded eutrophic shallow lake in Germany (Kühl and Kohl, 1993). Biomass collected at the end of the growing season displays much lower nitrogen content than in spring. Klopatek (1978) has shown trends of the same magnitude for cattail roots and shoots. It is apparent that the timing and location of vegetation samples can greatly affect subsequent calculations of nitrogen storage in biomass. The decline of aboveground tissue nutrient content is a common phenomenon in both treatment and natural wetlands (Table 9.2) and results in a markedly lower tissue nitrogen concentration at the end of the growing season. This is partly due to translocation to belowground rhizomes, which is discussed in a following section.

These seasonal storages reflect the growth cycle of the plant in question. The processes of growth, death, litterfall, and decomposition operate year-round, and with different speed and seasonality depending on climatic conditions and genotypical habit. Even in cold climates, the total annual growth is slightly larger than the end-of-season standing crop, by about 20% (Whigham *et al*., 1978). In warm climates, measurements show 3.5–10 turnovers of the live aboveground standing crop in the course of a year (Davis, 1994). Decay and translocation processes release most of the nitrogen uptake, with the residual accreting as new sediments and soils.

**TABLE 9.2**
**Whole Plant, Aboveground Foliar Nitrogen Concentration Declines through the Growing Season**

| Plant Species | Location | Water | Initial N (%) | Decline Rate (%/d) | $R^2$ | Reference |
|---|---|---|---|---|---|---|
| *Typha latifolia* | South Carolina | N | 2.47 | 0.0133 | 0.90 | Boyd (1971) |
| *Typha latifolia* | Michigan | S | 1.00 | 0.0004 | 0.75 | Houghton Lake, Michigan, unpublished data |
| *Typha angustifolia* | Michigan | S | 1.33 | 0.0027 | 0.77 | Houghton Lake, Michigan, unpublished data |
| *Typha* spp. | Minnesota | N | 1.80 | 0.0063 | 0.99 | Pratt *et al*. (1980) |
| *Typha* spp. | Minnesota | N | 1.70 | 0.0075 | 0.86 | Pratt *et al*. (1980) |
| *Scirpus validus*[a] | New Zealand | P | 1.46 | 0.0061 | 0.80 | Tanner (2001a) |
| *Scirpus validus* | New Zealand | P | 1.61 | 0.0059 | 0.82 | Tanner (2001a) |
| *Scirpus validus* | New Zealand | P | 1.79 | 0.0058 | 0.82 | Tanner (2001a) |
| *Scirpus validus* | New Zealand | P | 1.93 | 0.0087 | 0.88 | Tanner (2001a) |
| *Phragmites australis* | The Netherlands | N | 2.74 | 0.0100 | 0.90 | Mueleman *et al*. (2002) |
| *Phragmites australis* | Australia | AR | 4.22 | 0.0146 | 0.93 | Hocking (1989a, b) |
| *Phragmites australis* | The Netherlands | P | 2.54 | 0.0070 | 0.96 | Mueleman *et al*. (2002) |

*Note:* Initial %N is at the start of the growing season. Water type is N = no wastewater; S = nutrients at secondary treatment levels; P = nutrients at primary treatment levels; AR = agricultural runoff.

[a] Currently known as *Schoenoplectus tabernaemontani.*

**TABLE 9.3**
**End of Season Plant Biomass in Wetlands**

| Species | Location | Reference | Water | S/P/E | Live Above (g/m²) | Total Above (g/m²) | Roots and Rhizomes (g/m²) |
|---|---|---|---|---|---|---|---|
| **Cattails** | | | | | | | |
| *Typha latifolia* | Wisconsin | Smith *et al.* (1988) | N | 105/245/290 | — | 1,400 | 450 |
| *Typha latifolia* | Texas | Hill (1987) | N | 60/240/345 | — | 2,500 | 2,200 |
| *Typha glauca* | Iowa | van der Valk and Davis (1978) | N | 120/265/290 | 2,000 | — | 1,340 |
| *Typha latifolia* | Michigan | Houghton Lake, Michigan, unpublished data | N | 120/245/275 | 490 | 890 | 6,200 |
| *Typha latifolia* | Michigan | Houghton Lake, Michigan, unpublished data | S | 120/245/275 | 1,240 | 2,310 | 2,900 |
| *Typha latifolia* | Kentucky | Pullin and Hammer (1989) | P | — | 5,602 | — | 3,817 |
| *Typha angustifolia* | Kentucky | Pullin and Hammer (1989) | P | — | 5,538 | — | 4,860 |
| **Bulrushes** | | | | | | | |
| *Scirpus fluviatilis* | Iowa | van der Valk and Davis (1978) | N | 130/265/285 | 790 | — | 1,370 |
| *Scirpus validus*[a] | Iowa | van der Valk and Davis (1978) | N | 120/210/300 | 2,100 | — | 1,520 |
| *Scirpus validus* | New Zealand | Tanner (2001a) | P | 30/205/350 | 2,100 | 2,650 | 1,200 |
| *Scirpus validus* | Kentucky | Pullin and Hammer (1989) | P | — | — | 2,355 | 7,376 |
| *Scirpus cyperinus* | Kentucky | Pullin and Hammer (1989) | P | — | — | 3,247 | 12,495 |
| **Phragmites** | | | | | | | |
| *Phragmites australis* | United Kingdom | Mason and Bryant (1975) | N | 75/220/305 | 942 | 1,275 | — |
| *Phragmites australis* | Iowa | van der Valk and Davis (1978) | N | — | — | 1,110 | 1,260 |
| *Phragmites australis* | The Netherlands | Mueleman *et al.* (2002) | N | 105/255/350 | 2,900 | 3,200 | 7,150 |
| *Phragmites australis* | Brisbane | Greenway (2002) | S | — | 1,460 | 2,520 | 1,180 |
| *Phragmites australis* | The Netherlands | Mueleman *et al.* (2002) | P | 105/255/355 | 5,000 | 5,500 | 3,890 |
| *Phragmites australis* | New York | Peverly *et al.* (1993) | L | 100/270/330 | 10,800 | — | 8,700 |

*Note:* Water type is N = no wastewater; S = nutrients at secondary treatment levels; P = nutrients at primary treatment levels; L = landfill leachate with about 300 gN/m³. S/P/E refers to the start, peak, and end year-days of the growing season (182 days added for southern hemisphere).

[a] Currently known as *Schoenoplectus tabernaemontani.*

A common point of reference often used to assay biomass nitrogen is the end of the growing season. The compartments most often analyzed are live aboveground plant tissues, standing dead and litter, and belowground roots and rhizomes (Table 9.3). It is seen that a considerable fraction of the biomass is belowground, which is particularly troublesome from the standpoint of sampling, and hence often omitted. *A rough estimate of nitrogen storages in Table 9.3 may be obtained by multiplying the dry biomass by 2% nitrogen, resulting in a range of about 100–300 gN/m².* In treatment wetlands that are lightly loaded, this storage may be an important factor in the nitrogen budget, on a seasonal basis.

## 9.3 NITROGEN TRANSFORMATIONS IN WETLANDS

Figure 9.5 shows the principal components of the nitrogen cycle in wetlands. The various forms of nitrogen are continually involved in chemical transformations from inorganic to organic compounds and back from organic to inorganic. Some of these processes require energy (typically derived from an organic carbon source), and others release energy, which is used by organisms for growth and survival. Most of the chemical changes are controlled through the production of enzymes and catalysts by the living organisms they benefit.

The several nitrogenous chemical species are interrelated by a reaction sequence. Nitrogen is speciated in several forms in wetlands, as well as partitioned into water, sediment, and biomass phases. An FWS wetland is also stratified vertically into zones which promote different nitrogen reactions. As a further complicating factor, microenvironments around individual plant roots may differ from the bulk surroundings (Reddy and D'Angelo, 1994). Although the detailed processes are well known, they have not been adequately quantified as an integrated network for the wetland environment.

A number of processes transfer nitrogen compounds from one point to another in wetlands without resulting in a molecular transformation. These physical transfer processes include, but are not limited to the following: (1) particulate settling and resuspension, (2) diffusion of dissolved forms, (3) plant translocation, (4) litterfall, (5) ammonia volatilization, and (6) sorption of soluble nitrogen on substrates. In addition to the physical translocation of nitrogen compounds in wetlands, five principal processes transform nitrogen from one form to another: (1) ammonification (mineralization), (2) nitrification, (3) denitrification, (4) assimilation, and (5) decomposition. A detailed understanding of these nitrogen transfer and transformation processes is important for understanding wetland treatment systems. The sections below describe these processes and the environmental factors that

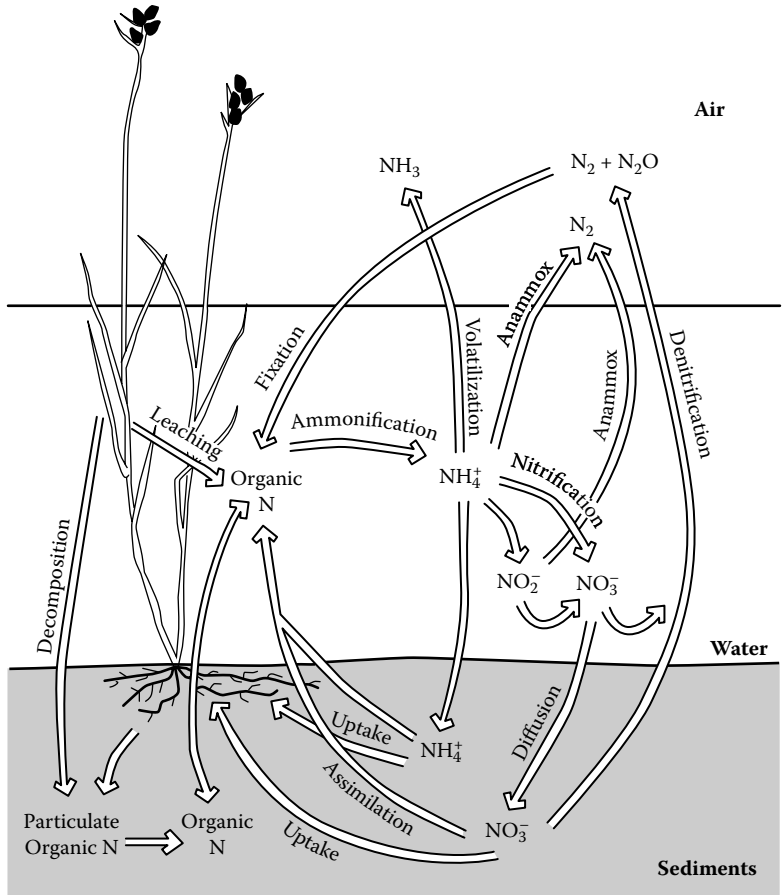



**FIGURE 9.5** Simplified nitrogen cycle for a FWS treatment wetland. (Modified from Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

regulate the transformations. Later in this chapter, empirical and theoretical design methods are presented for predicting the treatment wetland area necessary to accomplish the given nitrogen transformations.

## Physical Processes

The wetland nitrogen cycle includes a number of pathways that do not result in a molecular transformation of the affected nitrogen compound. These physical processes include atmospheric nitrogen inputs, ammonia adsorption, and ammonia volatilization. Sedimentation may also remove particulate nitrogen from the water, either as a structural component of the total suspended solids (TSS), or as sorbed ammonia (see Chapter 7).

### Atmospheric Deposition

Atmospheric deposition of nitrogen contributes measurable quantities of nitrogen to receiving land areas. All forms are involved: particulate and dissolved, and inorganic and organic. Wetfall contributes more than dryfall, and rain contributes more than snow (Table 9.4). The nitrogen concentration of rainfall is highly variable depending on atmospheric conditions, air pollution, and geographical location. A typical range of TN concentrations associated with rainfall is 0.5–3.0 mg/L, with more than half of this present as ammonia and nitrate nitrogen.

Some dryfall of nitrogen is also from deposition of organic dust containing organic and ammonia nitrogen. Typical dryfall nitrogen inputs are less than wetfall amounts. These concentrations can be used with local rainfall amounts to estimate rainfall inputs in nitrogen mass balances (Table 9.4). Annual total atmospheric nitrogen loadings are 10–20 kg/ha·yr. Consequently, atmospheric sources are almost always a negligible contribution to the wetland nitrogen budget for all but ombrotrophic, nontreatment wetlands.

### Ammonia Sorption

Oxidized nitrogen forms (e.g., nitrite and nitrate) do not bind to solid substrates, but ammonia is capable of sorption to both organic and inorganic substrates. Because of the positive charge on the ammonium ion, it is subject to cation exchange. Ionized ammonia may therefore be removed from water through exchange with detritus and inorganic sediments in FWS wetlands, or the media in SSF wetlands. The adsorbed ammonia is bound loosely to the substrate and can be released easily when water chemistry conditions change.

**TABLE 9.4**
**Atmospheric Deposition of Nitrogen**

| Location and Nitrogen Form | | Type of Deposition | Estimated Precipitation (mm) | Concentration (mg/L) | Load (kg/ha·yr) | Reference |
|---|---|---|---|---|---|---|
| **Geneva, New York** | Inorganic | Wet + dry | 993 | 1.1 | 10.9 | U.S. EPA (1993b) |
| **Coshocton, Ohio** | Inorganic | Wet + dry | 939 | 0.8 | 7.5 | U.S. EPA (1993b) |
| | Organic | Wet + dry | — | 0.37 | 3.5 | |
| **Cincinnati, Ohio** | Inorganic | Wet + dry | 1,020 | 0.69 | 7.0 | U.S. EPA (1993b) |
| | Organic | Wet + dry | — | 0.58 | 5.9 | |
| **Seattle, Washington** | Nitrate | Dry | — | — | 0.7 | U.S. EPA (1993b) |
| **Ottawa, Ontario** | Inorganic | Snow | 147 | 0.85 | 1.3 | U.S. EPA (1993b) |
| | Nitrate | Rain | 724 | 0.35 | 15.6 | |
| | Ammonia | Rain | — | 1.8 | 13.0 | |
| **Hamilton, Ontario** | Total nitrogen | Wet | 818 | 0.49 | 4 | U.S. EPA (1993b) |
| | Total nitrogen | Dry | — | — | 2.5 | |
| **Southern Florida** | Inorganic | Wet + dry | 1,500 | 0.75 | 6.1 | South Florida Water Management |
| | Organic | Wet + dry | — | 1.13 | 9.3 | District, unpublished data |
| | Particulate | Wet + dry | — | 0.94 | 7.7 | |
| **Midwestern United States** | Ammonia | Wet + dry | 889 | 0.34–0.45 | 3–4 | U.S. EPA (2001b) |
| **North Carolina** | Nitrate | Wet | 1,355 | 0.25 | 3.4 | Whitall and Paerl (2001) |
| | Ammonia | Wet | — | 0.23 | 3.1 | |
| | Organic | Wet | — | 0.23 | 3.2 | |
| **Chesapeake Bay** | | Wet (2/3) | | | | Sheeder *et al.* (2002) |
| | Inorganic | Dry (1/3) | 1,143 | 0.34–1.62 | 4–19 | |
| **Southern Sweden** | Total nitrogen | Wet + dry | 569 | 2.6–4.4 | 15–25 | U.S. EPA (1993b) |
| **Central Europe** | Total nitrogen | Wet + dry | 866 | 2.3–3.5 | 20–30 | U.S. EPA (1993b) |

At a given ammonia concentration in the water column, a fixed amount of ammonia is adsorbed to and saturates the available attachment sites.

The character of the substrate is an important determinant of the amount of sorption or exchange (Figure 9.6). Natural zeolites have more exchange capacity than do the gravels usually employed in SSF wetlands, by more than a factor of 100. Organic sediments and peats in FWS wetlands have capacities intermediate to zeolites and gravels. The exchange reaction involves protons on the substrate and ammonia:

$$R\mathrm{H}^+ + \mathrm{NH}_4^+ + \mathrm{OH}^- \rightleftharpoons R\mathrm{NH}_4^+ + \mathrm{H}_2\mathrm{O} \qquad (9.3)$$

where $R$ represents a ligand, such as the humic substances found in peat. Other cations, including sodium ($Na^+$), calcium ($Ca^{2+}$) and magnesium ($Mg^{2+}$), compete for exchange sites, and reduce the potential for ammonia exchange (Weatherly and Miladinovic, 2004). Hydrogen ions are also important, because these too reduce the exchange capacity. For example, McNevin and Barford (2001) found the direct dependence for Killarney peat, over the range $3.9 < \mathrm{pH} < 7.5$ to follow:

$$K_{\mathrm{exch}} = \frac{C_S}{C_L} = 0.0018(\mathrm{pH})^{5.438} \qquad (9.4)$$

where

$C_L$ = ammonia concentration in water, mg/L
$C_S$ = ammonia concentration on solid, mg/kg
$K_{exch}$ = partition coefficient, L/kg

When the ammonia concentration in the water column is reduced, some ammonia will be desorbed to regain equilibrium

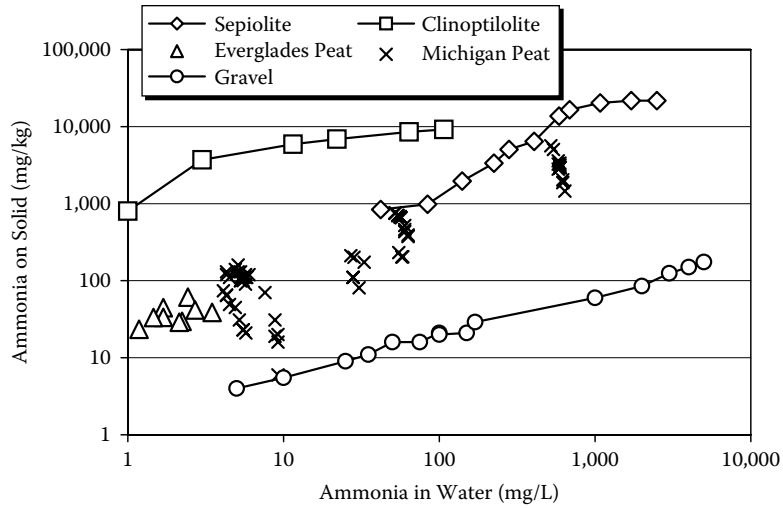



**FIGURE 9.6** Ammonium adsorption on FWS and SSF wetland substrates. (The gravel data are from Sikora *et al*. (1994) *Ammonium and phosphorus removal in constructed wetlands with recirculating subsurface flow: Removal rates and mechanisms*. Jiang (Ed.), Proceedings of the 4th International Conference on Wetland Systems for Water Pollution Control, 6–10 November 1994; IWA: Guangzhou, P.R. China, pp. 147–161. Everglades peat data from Reddy *et al*. (1991) *Physico-Chemical Properties of Soils in the Water Conservation Area 2 of the Everglades*. Report to the South Florida Water Management District, West Palm Beach, Florida. Michigan peat data from unpublished results at Houghton Lake. Sepiolite data from Balci (2004) *Water Research* 38(5): 1129–1138. Clinoptilolite data from Weatherly and Miladinovic (2004) *Water Research* 38(20): 4305–4312.)

with the new concentration. If the ammonia concentration in the water column is increased, the adsorbed ammonia will also increase.

The mass of sorbed ammonia nitrogen on detritus and sediment in an FWS wetland is not large, and is very labile. The top 20 cm of the wetland substrate may contain up to 20 gN/m$^2$ in exchangeable form for a peat exposed to 10 mg/L ammonium nitrogen. This pool of nitrogen is quickly established at moderate nitrogen loadings (see Chapter 10 for an analogous discussion of sorption saturation times for phosphorus). At light nitrogen loadings, a short start-up period may be influenced by this storage.

Wittgren and Maehlum (1997) suggest that seasonal sorption could store ammonia for later use and release. Riley *et al*. (2005) found rapid uptake to sorption, with little or no subsequent ammonia loss. Their linear sorption $K_D = 0.083$ L/kg. (Sorption relationships are discussed in more detail in Chapter 10—the following discussion focuses in ammonia sorption only.)

$$\text{Gravel:} \qquad 0.3-1.3 \text{ cm} \qquad C_S = 0.083 C_L^{1.00} \tag{9.5}$$

Sikora *et al*. (1995b) provided data from which Freundlich constants could be fit:

$$\text{Fine gravel:} \qquad 0.5-1.0 \text{ cm} \qquad C_S = 0.77 C_L^{0.64} \tag{9.6}$$

$$\text{Coarse gravel:} \qquad 0.5-2.0 \text{ cm} \qquad C_S = 1.63 C_L^{0.55} \tag{9.7}$$

Weatherly and Miladinovic (2004) provided Langmuir constants for the zeolites clinoptilolite and mordenite:

$$\text{Clinoptilolite:} \qquad \frac{1}{K} = 2.5 \text{ mg/L} \qquad S_{max} = 6.9 \text{ g/kg} \tag{9.8}$$

$$\text{Mordenite:} \qquad \frac{1}{K} = 19.6 \text{ mg/L} \qquad S_{max} = 13.1 \text{ g/kg} \tag{9.9}$$

Lahav and Green (2000) provided Langmuir constants for the zeolite chabazite:

$$\text{Chabazite:} \qquad \frac{1}{K} = 10.0 \text{ mg/L} \qquad S_{max} = 50.5 \text{ g/kg} \tag{9.10}$$

The median ammonia loading for HSSF systems is about 1.0 g/m$^2$·d, and the median concentration is 20 mg/L. For the parameters above, the equilibrium ammonia sorbed at 20 mg/L is 2–25 g/m$^2$ for a 60-cm deep bed. Therefore, the bed solids can hold approximately 2–25 days' supply of ammonia via sorption phenomena.

However, if the wetland substrate is exposed to oxygen, perhaps by periodic draining, sorbed ammonium may be oxidized to nitrate. Nitrate is not bound to the substrate, and is washed out by subsequent rewetting. This concept forms the basis for intermittently fed and drained, vertical flow treatment wetlands, and for other wetland systems that are alternately flooded and drained.

## Ammonia Volatilization

Un-ionized ammonia is relatively volatile and can be removed from solution to the atmosphere through diffusion through water upward to the surface, and mass transfer from the water surface to the atmosphere.

### Theoretical Considerations

Total dissolved ammonia exists in the two forms, free or un-ionized ($NH_3$), and ionized ($NH_4^+$). These interconvert readily in water, according to Equation 9.2, which allows the computation of the concentration of free ammonia in terms of total ammonia:

$$C_{AL} = \frac{C_{ATL}}{1 + K_d} \tag{9.11}$$

where

$C_{AL}$ = concentration of free ammonia in the bulk water, g/m$^3$
$C_{ATL}$ = concentration of total ammonia in the bulk water, g/m$^3$

Free ammonia may also exist as a gas, whereas ionized ammonia is nonvolatile. The process of volatilization carries free ammonia from the water into the air above. That overall process comprises four major components in series (see Chapter 5): (1) partial conversion of ionized ammonia to free ammonia (dissociation), (2) diffusion of free ammonia to the air–water interface (water-side mass transfer), (3) release of free ammonia to the air at the interface (volatilization), and (4) diffusion of free ammonia from the air–water interface into the air above (air-side mass transfer). These component processes are conceptually well understood because of studies associated with ammonia stripping as an engineering technology.

The loss of free ammonia may be described by a two-film mass transfer equation (Welty *et al*., 1983; Liang *et al*., 2002):

$$J = k(C_{AL} - C^*_{AL}) \tag{9.12}$$

$$\frac{1}{K_L} = \frac{1}{k_L} + \frac{1}{Hk_G} \tag{9.13}$$

where

$C^*_{AL}$ = water concentration of free ammonia that would be in equilibrium with the free ammonia in the bulk air, g/m$^3$
$H$ = Henry's Law coefficient, dimensionless
$K_L$ = overall mass transfer coefficient, m/d
$k_G$ = air-side mass transfer coefficient, m/d
$k_L$ = water-side mass transfer coefficient, m/d

Water–air equilibrium, or solubility, is governed by Henry's law:

$$C^*_{AL} = \frac{C_{AG}}{H} \tag{9.14}$$

where

$C_{AG}$ = concentration of free ammonia in the bulk air, g/m$^3$

The value of $H$ is temperature-dependent (Liang *et al*., 2002):

$$H = \left(\frac{2.395 \times 10^5}{T + 273.16}\right) \exp\left(\frac{-4151}{T + 273.16}\right) \tag{9.15}$$

Under almost all circumstances, the ammonia concentration in the air above the wetland will be negligibly small, and hence may be presumed to be zero. Additionally, total ammonia rather than free ammonia is used in the overall vapor loss equation:

$$J = K_L C_{AL} \tag{9.16}$$

$$J = K_L \left(\frac{C_{ATL}}{1 + K_d}\right) = kC_{ATL} \tag{9.17}$$

where

$k$ = first-order volatilization rate constant based on total ammonia, m/d

There are two choices for a first-order removal rate: one based on the free ammonia concentration in the water (Equation 9.16), and one based on the total ammonia concentration in the water (Equation 9.17); the latter is used here.

#### *Practical Application*

Many factors influence component processes, most of which will not be known or measured for field situations involving treatment wetlands. Solubility depends on temperature, and degree of ionization depends on temperature and pH. However, the process of ammonia volatilization involves proton transfer, and a theoretical decrease in pH. Such a decrease has been observed in laboratory volatilization tests (Shilton, 1996). Additionally, both temperature and pH undergo large diurnal swings in some treatment wetlands up to 8°C and 2 pH units. In some few situations, there may be vertical stratification of the water column, leading to interfacial temperature and pH conditions that deviate from those in the bulk water (Jenter *et al*., 2003).

The water-side mass transfer coefficient ($k_L$) depends upon the degree of turbulence (mixing) in the water, which in turn depends on depth, velocity, and the amount of submersed plant and litter material (Serra *et al*., 2004), together with the wind speed (Liang *et al*., 2002). The air-side mass



**FIGURE 9.7** Ammonia losses were measured directly at ponds at Greensboro, North Carolina. (Photo courtesy M. Poach.)

transfer coefficient ($k_G$) depends upon the degree of turbulence (mixing) in the air, which in turn depends on wind speed and amount of emergent plant biomass. The studies of Liang *et al*. (2002) suggest that both air-side and water-side mass transfer resistance are important for ammonia losses from ponds. That is in contrast to the work of Freney *et al*. (1985), which suggested that for a rice crop, the mass transfer resistance was entirely in the air. Therefore, ammonia loss rates should depend not only upon temperature and pH, but also on site-specific conditions (see Figure 9.7).

Several studies of ammonia volatilization from ponds and wetlands provide data from which first-order rate constants may be calculated (Table 9.5). Values of $k$ range from 0.11 to 28 m/yr, which is an unacceptably large range. A modified Arrhenius temperature factor developed from the data of Stratton (1969) is $\theta = 1.094$. This was used to adjust rate constants to 20°C in Table 9.5. The $k_{20}$ values so computed for wetland systems span a much narrower range 0.28–0.68 m/yr, with mean ± SD = 0.47 ± 0.14. For pond systems, the values are much higher, mean ± SD = 4.2 ± 4.6. There is also a clear trend of increasing $k$ with pH for ponds, which has been reported in several studies (Stratton, 1968; Shilton, 1996; Liang *et al*., 2002). The reduced rates for wetlands may be attributed to the vegetation, which breaks the wind and thus lowers both the water-side and air-side mass transfer coefficients. Presumably, there would be a pH effect for wetlands, but FWS wetland pH values are most often tightly clustered in the range 7.0–7.5, thus preventing the manifestation of a pH effect.

These considerations indicate that emergent FWS wetlands will lose much less ammonia to volatilization than will ponds. Therefore, inclusion of open water sections in FWS treatment wetlands encourages ammonia loss (Poach *et al*., 2004; see Figure 9.8). Volatilization rate constants for vegetated wetlands are quite small compared with rate constants for other mechanisms, as will be discussed in the following text. However, the same is not necessarily true for open water components.

## Microbial Processes

Wetlands are a rich environment for a large suite of microbes that mediate or conduct numerous chemical reactions involving nitrogen. Heterotrophic bacteria derive carbon from preformed organic compounds, whereas autotrophs acquire energy and carbon from inorganic sources. Denitrification is often, but not always, accomplished by heterotrophs in wetlands, while nitrification is carried out autotrophically. Microbes also produce enzymes that can break down complex molecules, both inside and outside the cell. Microbes are preferentially associated with solid surfaces, rather than as free-floating organisms. The principal nitrogen microbial wetland processes are therefore carried out in biofilms located on soils, sediments, and submerged plant parts.

In the following sections, the principal nitrogen conversions are discussed in more detail (see Figure 9.5).

### Ammonification of Organic Nitrogen

Ammonification is the biological transformation of organic nitrogen to ammonia and is the first step in mineralization of organic nitrogen (Reddy and Patrick, 1984). This process occurs both aerobically and anaerobically, and releases ammonia from dead and decaying cells and tissues. Heterotrophic microorganisms are considered to be the group involved (U.S. EPA, 1993b). The reactions can take place intracellularly or extracellularly, via the action of enzymes acting upon proteins, nucleic acids, and urea (Maier *et al*., 2000). The sources of nitrogenous organics are plant and animal tissues, and direct excretion of urea.

Typical ammonification reactions are:

Urea breakdown

$$NH_2CONH_2 + H_2O \rightarrow 2NH_3 + CO_2 \tag{9.18}$$

Amino acid breakdown

$$RCH(NH_2)COOH + H_2O \rightarrow NH_3 + CO_2 \tag{9.19}$$

**TABLE 9.5**
**Rate Constants for Ammonia Volatilization for Ponds and Wetlands**

| Site | T (°C) | pH | Total $NH_3$–N ($g/m^3$) | Un-ionized $NH_3$–N ($g/m^3$) | Loss rate ($g/m^2 \cdot yr$) | Total $NH_3$–N $k$ (m/yr) | Total $NH_3$–N $k_{20}$ (m/yr) | Reference |
|---|---|---|---|---|---|---|---|---|
| **Duplin County, North Carolina** | | | | | | | | Poach *et al*. (2002, 2003) Field: large-scale chambers |
| Grass | 23 | 7.2 | 55 | 0.51 | 30 | 0.46 | 0.35 | |
| Bulrush | 24 | 7.2 | 46 | 0.42 | 19 | 0.40 | 0.28 | |
| Cattail | 22 | 7.2 | 46 | 0.36 | 20 | 0.57 | 0.48 | |
| **Greensboro, North Carolina** | | | | | | | | Poach *et al*. (2004) Field: large-scale chambers |
| Bulrush and Cattail | 23 | 7.0 | 60 | 0.54 | 46 | 0.70 | 0.53 | |
| Pond | 25 | 7.4 | 40 | 1.08 | 237 | 10.4 | 6.6 | |
| **Ujjain, India** | | | | | | | | Billore *et al*. (1994) Field: small-scale chambers |
| Water | 35 | — | 4 | — | 15 | 4.2 | 1.09 | |
| Duckweed | 35 | — | 7 | — | 46 | 6.5 | 1.69 | |
| Cattail | 35 | — | 14 | — | 37 | 2.6 | 0.68 | |
| **New Zealand** | | | | | | | | Shilton (1996) Lab: small-scale chambers |
| Pond | 20 | 8.6 | 549 | 86 | 389 | 0.69 | 0.69 | |
| **Al-Bireh, Palestine** | | | | | | | | Zimmo *et al*. (2003) Field: small-scale chambers |
| Duckweed | 17 | 7.8 | 50 | 1.12 | 5.4 | 0.11 | 0.14 | |
| Pond | 17 | 8.1 | 38 | 1.38 | 6.7 | 0.19 | 0.25 | |
| **Griffith, Australia** | | | | | | | | Freney *et al*. (1985) Field: air-side measurements |
| Rice | 20 | 8.0 | 73 | 2.78 | 34.9 | 0.48 | 0.48 | |
| **San Diego, California** | | | | | | | | Stratton (1969) Lab: flow chambers |
| Pond | 29 | 9.8 | 0.47 | 0.39 | 0.89 | 28 | 12.5 | |
| Pond | 32 | 9.1 | 1.75 | 0.92 | 37 | 20 | 6.8 | |

*Note:* Values based on total ammonia are shown.

It is curious that the wastewater treatment literature does not directly address ammonification, despite the considerable proportion of organic nitrogen in raw wastewaters. The ammonification step is identified on diagrams, but no mention of chemistry or rates is found in manuals (Brown and Caldwell, 1975; U.S. EPA, 1993b) or texts (Metcalf and Eddy Inc., 1991). In some instances, it is recommended to lump organic and ammonium (as TKN) in calculations of

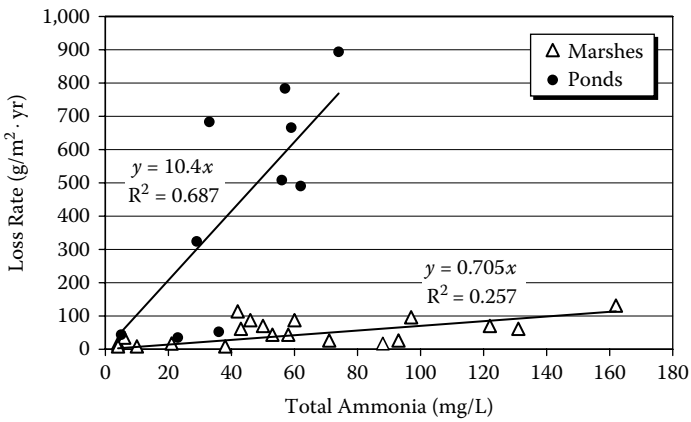



**FIGURE 9.8** Ammonia volatilization losses from 12 marshes and 6 ponds at Greensboro, North Carolina. Conditions in the marshes were $T = 23°C$, pH = 7.0; in the ponds $T = 25°C$, pH = 7.4; wind was 0.2–1.5 m/s. (Replotted from Poach *et al*. (2003) *Ecological Engineering,* 20(2): 183–197, with zero intercept.)

ammonia processing, on the presumption that organic nitrogen will add to the potential ammonia concentrations (U.S. EPA, 2000a). That procedure can be misleading for two reasons. First, ammonification is not instantaneous, and conversion proceeds at rates that influence the removal of TKN in many instances. Kinetically, ammonification proceeds more rapidly than nitrification, thus creating the potential for increasing ammonia concentrations along the flow-path of a wetland and requiring design for nitrogen removal to include both ammonification and the slower nitrification process. Second, the ammonification process does not proceed to completion in wetlands, although the removal of ammonia can go to completion for long enough detention. There is an organic nitrogen background concentration which may consist of irreducible residuals, or be due to return fluxes of organic nitrogen from decomposing solids.

### Nitrification of Ammonia

Nitrification is the principal transformation mechanism that reduces the concentration of ammonia nitrogen in many wetland treatment systems, by converting ammonia nitrogen to oxidized nitrogen, van de Graaf *et al*. (1996) defined nitrification as the biological formation of nitrate or nitrite from compounds containing reduced nitrogen with oxygen as the terminal electron acceptor. Nitrification has been typically associated with the chemoautotrophic bacteria, although it is now recognized that heterotrophic nitrification occurs and can be of significance (Keeney, 1973; Paul and Clark, 1996).

### Results from Conventional Wastewater Treatment Processes

Biological nutrient removal systems may be broadly categorized as suspended growth (e.g., activated sludge) or attached growth (e.g., trickling filters). In such devices, nitrification is considered to be a two-step, microbially mediated process in U.S. EPA (1993b):

$$\text{Nitritation} \quad 2NH_4^+ + 3O_2 \xrightarrow{Nitrosomonas} 2NO_2^- + 2H_2O + 4H^+ \tag{9.20}$$

$$\text{Nitrification} \quad 2NO_2^- + O_2 \xrightarrow{Nitrobacter} 2NO_3^- \tag{9.21}$$

The first step, nitritation, is mediated primarily by autotrophic bacteria in the genus *Nitrosomonas* and the second step, nitrification, by bacteria in the genus *Nitrobacter*. Both steps can proceed only if oxygen is present, and thus the actual nitrification rate may be controlled by the flux of dissolved oxygen into the system.

Based on this stoichiometric relationship, the theoretical oxygen consumption by the first nitritation reaction is about 3.43 g $O_2$ per gram of $NH_3$–N oxidized, and 1.14 by the second nitrification reaction, for a total of 4.57. Actual consumption is reportedly somewhat less, 4.3 g $O_2$ per gram of $NH_3$–N oxidized (Metcalf and Eddy Inc., 1991). The oxidation reactions release energy used by both *Nitrosomonas* and *Nitrobacter* for cell synthesis. The combined processes of cell synthesis create 0.17 g of dry weight biomass per gram of ammonia nitrogen consumed (U.S. EPA, 1993b). Nitrification of ammonia to nitrate consumes approximately 7.1 g of alkalinity (as $CaCO_3$) for each nitrified gram of ammonia nitrogen, as two moles of $H^+$ are released for each mole of ammonia nitrogen consumed in Equation 9.20 (U.S. EPA, 1993b). Thus nitrification lowers the alkalinity and pH of the water.

The optimal pH range observed for nitrification in suspended growth treatment systems is between about 7.2 and 9.0 (Metcalf and Eddy, Inc., 1991). Treatment wetlands almost always operate at circumneutral pH (see Chapter 5); consequently, this factor should be a minor influence on nitrification in those systems.

### Wetland Environments

Natural environments are considerably more complex than the situations in biological nutrient removal systems in conventional wastewater treatment plants (WWTPs). There are now enough wetland data to begin to understand some differences, and to appreciate that WWTP results may not apply to wetlands.

There are more genera potentially involved in natural systems than those identified above. Ammonia oxidizing bacteria (AOB) include *Nitrosospira* and *Nitrosococcus* in addition to *Nitrosomonas* (Bothe *et al*., 2000). Austin *et al*. (2003) found *Nitrosospira* just as abundant as *Nitrosomonas* in a treatment wetland, with lesser numbers of *Nitrosococcus*. Likewise, nitrite is oxidized by *Nitrospira* as well as *Nitrobacter*, and the former was found to be much more prevalent in a treatment wetland (Austin *et al*., 2003). Furthermore, heterotrophic bacteria are capable of nitrification, such as *Paracoccus denitrificans* and *Pseudomonas putida* (Bothe *et al*., 2000). Nevertheless, *Nitrosomonas* is found in treatment wetlands (Silyn-Roberts and Lewis, 2001). The oxidation of ammonia to nitrite in natural systems is suggested to comprise two steps, not one (Bothe *et al*., 2000), catalyzed by enzymes:

$$NH_3 + O_2 + 2H^+ + 2e^- \xrightarrow{\text{Ammonia monooxygenase}} NH_2OH + H_2O \tag{9.22}$$

$$NH_2OH + H_2O \xrightarrow{\text{Hydroxylamine oxidoreductase}} NO_2^- + 5H^+ + 4e^- \tag{9.23}$$

This scheme suggests that hydroxylamine is an intermediate in the process, which presents alternate nitrogen processing possibilities. Further, one of the oxygen atoms in nitrite derives from $O_2$, the other from water.

Nitrite oxidizing bacteria (NOB) were found not to include *Nitrobacter* in two FWS treatment wetlands (Flood *et al*., 1999). Similarly, Austin *et al*. (2003) found *Nitrospira* (4% of total) to be much more abundant than *Nitrobacter*

(0.1% of total) in a treatment wetland. Importantly, nitrite may be also be destroyed by processes other than conversion to nitrate, as shall be discussed in a later section.

On a practical level, these considerations cast doubt about the applicability to wetlands of the stoichiometry advocated for WWTP environments (Equations 9.20 and 9.21). For instance, the dissolved oxygen requirement for Equations 9.22 and 9.23 is 1.14 g $O_2$ per gram of ammonia nitrogen, rather than the 3.43 suggested by Equation 9.20. Alkalinity requirements are also greatly reduced. The stoichiometric factor of 4.3 g $O_2$ per gram of $NH_4$–N oxidized has been used in many treatment wetland publications as a means of inferring the maximum amount of oxygen transferred into the water (e.g., Platzer, 1999; Cooper, 2001, 2005). But, in many wetland situations, the 4.3 factor does not seem to be applicable (Tanner and Kadlec, 2002). These alternative pathways with the potential to substantially reduce the oxygen fluxes required to drive $NH_4$–N removal need to be investigated further in both natural and constructed wetlands to develop an understanding of their role in wetland nitrogen removal.

The necessity of a low carbon-to-nitrogen ratio, another concept from activated sludge and attached growth technologies, appears dubious for wetlands. It has been suggested that the biochemical oxygen demand (BOD) level "must be below (BOD/TKN < 1.0)" for "successful nitrification" in treatment wetlands (Reed *et al*., 1995; Crites *et al*., 2006). In conventional devices, the carbon consumption activity of heterotrophs may cause them to dominate the overall bacterial population, but with a smooth transition from 3% to 35% nitrifiers as the $BOD_5$:TKN ratio decreases from 9 to 0.5 in activated sludge plants (Metcalf and Eddy Inc., 1991). Similarly, the result is a smooth decrease in nitrification rates in attached growth systems, from a relative level of 100% in the absence of BOD to 40% at $BOD_5$:TKN = 5.0 (Brown and Caldwell, 1975).

Free water surface treatment wetlands operate with a variety of inlet carbon-to-nitrogen ratios, ranging from 0.28 to 4.41 (5th to 95th percentiles, N = 126 wetlands). The mean inlet ratio is 2.0, and the mean outlet ratio is 1.6. Only one third of the 126 FWS wetlands met the criterion BOD:TKN < 1.0. This distribution is rather narrow, and would not lead to marked differences in potential nitrification rates. Considering direct evidence, there is essentially no correlation between the BOD:TKN ratio and measures of nitrification performance. For example, the TKN load removed versus BOD:TKN ratio has an $R^2 = 0.037$. Transect data sets display no nitrogen removal lag as carbon is removed (Tanner *et al*., 2002a). Therefore, it is not reasonable to accept this ratio as a controlling factor in FWS wetlands.

## Denitrification

Denitrification is most commonly defined as the process in which nitrate is converted into dinitrogen via intermediates nitrite, nitric oxide, and nitrous oxide (Hauck, 1984; Paul and Clark, 1996; Jetten *et al*., 1997).

Denitrification (nitrate dissimilation) is carried out by facultative heterotrophs, organisms that can use either oxygen or nitrate as terminal electron acceptors. Starting from nitrate via nitrite, there is sequential production of nitric oxide (NO), nitrous oxide ($N_2O$), and nitrogen gas ($N_2$) (e.g., Cox and Payne, 1973; Koike and Hattori, 1978):

$$2NO_3^- \rightarrow 2NO_2^- \rightarrow 2NO \rightarrow N_2O \rightarrow N_2 \qquad (9.24)$$

Diverse organisms are capable of denitrification. In an array are organotrophs (e.g., *Pseudomonas, Alcaligenes, Bacillus, Agrobacterium, Flavobacterium, Propionibacterium, Vibrio*), chemolithotrophs (e.g., *Thiobacillus, Thiomicrospira, Nitrosomonas*), photolithotrophs (e.g., *Rhodopseudomonas*), diazotrophs (e.g., *Rhizobium*, *Azospirillum*), archaea (e.g., *Halobacterium*), and others such as *Paracoccus* or *Neisseria* (Focht and Verstraete, 1977; Knowles, 1982; Killham, 1994; Paul and Clark, 1996).

### Results from Conventional Wastewater Treatment Processes

The overall stoichiometric nitrate dissimilation reaction based on methanol ($CH_3OH$) as a carbon source is summarized by the following (U.S. EPA, 1993b):

$$NO_3^- + 0.833\,CH_3OH \rightarrow 0.5\,N_2 + 0.833\,CO_2 + 1.167\,H_2O + OH^- \qquad (9.25)$$

Other carbon sources also may drive denitrification, such as glucose (Reddy and Patrick, 1984):

$$NO_3^- + 0.208\,C_6H_{12}O_6 \rightarrow 0.5\,N_2 + 1.25\,CO_2 + 0.75\,H_2O + OH^- \qquad (9.26)$$

The carbon (energy) requirements are 1.90 g methanol and 2.67 g glucose per gram of nitrate nitrogen, respectively. Some nitrate and carbon are also used by denitrifying bacteria for cell synthesis. For instance, another 0.57 g methanol is required for bacterial growth, bringing the total to 2.47 g of methanol to support the denitrification of 1 g of nitrate nitrogen. This translates to an optimum carbon level of 2.3 g BOD per g $NO_3$–N (Gersberg *et al*., 1984). In the absence of this or another equivalent carbon source, denitrification is inhibited.

As indicated by Equations 9.25 and 9.26, denitrification produces alkalinity. The observed yield of this process is about 3.0 g alkalinity as $CaCO_3$ per gram of $NO_3$-N reduced. This increase in alkalinity is accompanied by an increase in the pH of the wetland surface water.

Theoretically, denitrification does not occur in the presence of dissolved oxygen. However, denitrification has been observed in suspended and attached growth treatment systems that have relatively low measured dissolved oxygen concentrations, but not above 0.3–1.5 mg/L (U.S. EPA, 1993b).

This is presumably due in part to the activity of aerobic denitrifiers, such as *Paracoccus denitrificans*.

## Wetland Environments—Carbon Sources

The carbon source in wetlands is neither methanol nor glucose, but rather organic matter that is sometimes characterized by the Redfield ratio C:N:P = 106:16:1 (Davidsson and Stahl, 2000). The denitrification reaction is then written:

$$84.8\,NO_3^- + (CH_2O)_{106}(NH_3)_{16}(H_3PO_4) \rightarrow$$
$$42.4\,N_2 + 106\,CO_2 + 16\,NH_3 + 148.8\,H_2O + H_3PO_4 \quad (9.27)$$

This reaction is irreversible in nature, and occurs in the presence of available organic substrate only under anaerobic or anoxic conditions ($E_h$ = +350 to +100 mV), where nitrogen is used as an electron acceptor in place of oxygen. More and more evidence is being provided from pure culture studies that nitrate reduction can occur in the presence of oxygen. Hence, in waterlogged soils, nitrate reduction may also start before the oxygen is depleted (Kuenen and Robertson, 1987; Laanbroek, 1990).

The carbon (energy) requirement is 3.02 g organic matter per gram of nitrate nitrogen. Further, some ammonia is theoretically liberated, which can support growth or add to the overall wetland ammonia pool.

As most denitrification is accomplished by heterotrophic bacteria, the process is strongly dependent on carbon availability. There is a general correlation between total soluble organic matter content and denitrification potential, but much better correlation occurs with the supply of easily decomposable organic matter or water-extractable organic carbon (Bremner and Shaw, 1958; Broadbent and Clark, 1965; Paul and Clark, 1996). Organic substances able to act as sources of energy and as hydrogen donors may be present in sediments and soils through the decomposition of tissues or be provided by living roots exudates (Stefanson, 1973; Bailey, 1976).

A number of treatment wetland studies have investigated the use of carbon supplements in the form of added plant biomass (Gersberg *et al*., 1983, 1984; Burchell *et al*., 2002; Hume *et al*., 2002a). Another study added methanol (Gersberg *et al*., 1983), with good effect. Burgoon (2001) provided carbon by feed-forward of un-nitrified influent to wetlands receiving nitrified potato processing waters. All such studies have shown that carbon can be limiting in wetlands at high nitrate loadings. The amount of total carbon in dead and decomposing biomass is on the order of 40% of the dry biomass (Ingersoll and Baker, 1998; Baker, 1998; Hume *et al*., 2002b). Not all of the total carbon produced is available for denitrifiers. Baker (1998) has suggested that the C:N loading ratio be at least 5:1 so that carbon does not become limiting, which in his work translated to 20% availability. Hume *et al*. (2002b) suggest 8% availability. Presuming a carbon content of 40%, the required productivities are at the lower end of the range for emergent marshes (Kadlec and Knight, 1996). However, realization of higher nitrate removal rates, corresponding to higher inlet concentrations, may stress the ability of the wetland to generate the required carbon energy source. If carbon is limiting, the rate of denitrification will depend strongly on the rate of carbon supply (Hume *et al*., 2002a).

It should be noted that the most labile form of organic carbon in wetland environments is the influent BOD, which is likely used preferentially (when available) to reduce oxidized forms of nitrogen.

## Wetland Environments—Oxygen Inhibition

Denitrification has been observed in numerous wetland treatment systems which have considerable dissolved oxygen in their surface waters (Van Oostrom and Russell, 1994; Phipps and Crumpton, 1994). This apparent anomaly is due to the complicated spatial zonation in a wetland. Oxygen gradients occur between surface waters and bottom sediments in wetlands, allowing both aerobic and anoxic reactions to proceed in close vertical proximity (millimeters) near the sediment–water interface (Figure 9.9). Thus, nitrate formed by nitrification in surface waters may diffuse into top anoxic soil layers where it is effectively denitrified (Reddy and Patrick, 1984).

Significant quantities of oxygen pass down through the airways to the roots (Brix and Schierup, 1990; Brix, 1993); and significant quantities of other gases, such as carbon dioxide and methane, pass upward from the root zone. Some—perhaps most—of the oxygen passing down the plant into the root zone is used in plant respiration (Brix, 1990). However, there is a great deal of chemical action in the microzones near the roots of wetland plants. Figure 9.10 shows that the oxygenated microzone around a rootlet can conduct nitrification reactions, whereas denitrification reactions can be occurring only microns away in the anaerobic bulk soil. Diffusion easily connects these zones because of their close proximity.

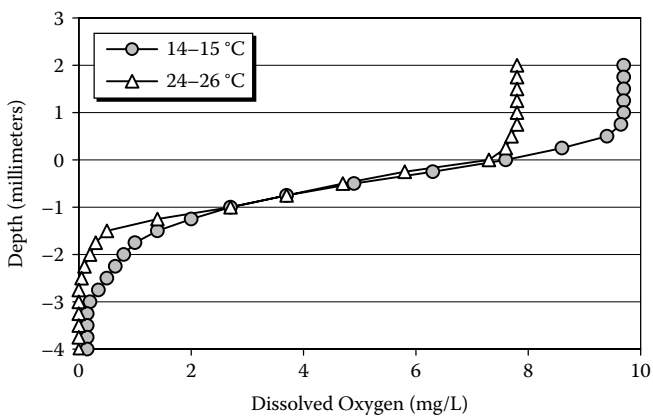



**FIGURE 9.9** Oxygen distribution above and below the sediment–water interface at two different temperatures. (Data from Crumpton and Phipps (1992) *The Des Plaines River Wetlands Demonstration Projects.* Vol. III, chap 5. Wetlands Research, Inc., Chicago, Illinois.)

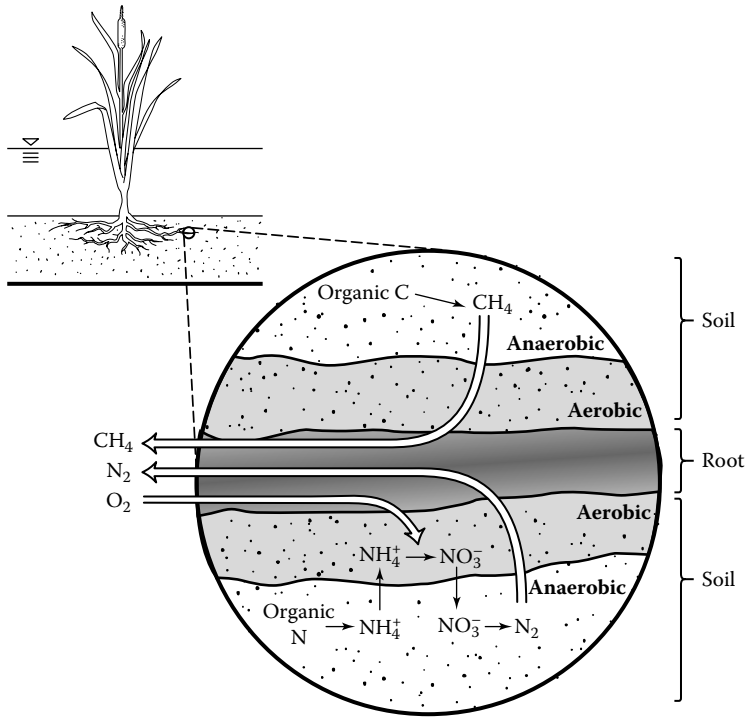



**FIGURE 9.10** Pathways of nitrogen transformations in the immediate vicinity of a plant root.

Bacteria attached to surfaces are usually more numerous than free-living (planktonic) bacteria (Bastviken *et al*., 2003, 2005). Attached bacteria form microbial communities that are embedded in polysaccharide matrixes, e.g., biofilms, and the bacterial activity within these biofilms is regulated by diffusion of nutrients into the biofilm and by internal processes within this layer. In wetlands, these surfaces are as important as the sediment for the nitrogen turnover processes (Eriksson and Weisner, 1997; Eriksson, 2001). Biofilms, therefore, comprise a third type of spatial nonuniformity in the wetland environment. Diffusion within the biofilm controls the internal supplies of oxygen, nitrate, and ammonia, thus regulating the net effects of bacterial conversions. In surface flow treatment wetlands, biofilms have been found to contain $10^8$–$10^9$ organisms/cm$^2$, mostly beta and gamma *Proteobacteria* (Flood *et al*., 1999). Ammonia oxidizers (beta) were more prevalent near the inlet; denitrifiers (gamma) were more prevalent near the outlet. Alum addition was found to totally eliminate these bacteria.

Another type of spatial nonuniformity exists due to the presence of longitudinal gradients in dissolved oxygen in the flow direction. Oxygen may be depleted by heterotrophic activity, as well as nitrification; but atmospheric reaeration also occurs.

Clearly, wetland oxygen environments are much more complex than either the complete-mix situation that dominates activated sludge processing or the attached growth environment of trickling filters. Results from those technologies should not be extrapolated to treatment wetlands.

## Wetland Environments—Dissimilatory Nitrate Reduction to Ammonium Nitrogen

Nitrate loss in treatment wetlands is often attributed to denitrification in the absence of proof that this mechanism is indeed the operative one. Other known and studied candidate mechanisms in wetlands include assimilation by plants and microbiota, and dissimilatory reduction to ammonium nitrogen (DNRA). These alternative reduction routes have been documented to comprise from 1–34% of the total nitrate loss (Bartlett *et al*., 1979; Stengel *et al*., 1987; Cooke, 1994; Van Oostrom and Russell, 1994). Bartlett *et al*. (1979) measured production of ammonium, dinitrogen, and nitrous oxide for microcosms with soils from a treatment wetland, but with no plants. From 1–6% of the product was ammonium nitrogen; the balance was measured as dinitrogen, with only trace amounts of nitrous oxide. Cooke (1994) measured $^{15}$N-labelled nitrate, ammonium, and organic nitrogen in unvegetated microcosms in a treatment wetland. He found 34%, 6%, and 60% of $K^{15}NO_3$ converted by dissimilatory processes, microbial assimilatory processes, and denitrification, respectively, at one site; and 25%, 5%, and 70% at a second site. Stengel *et al*. (1987) used the acetylene blockage technique to establish that 75–90% of the nitrate loss in a flow through, *Phragmites*/gravel SSF unit was due to denitrification. Van Oostrom and Russel (1994) measured 16% dissimilatory nitrate reduction in microcosms containing *Glyceria maxima* mats.

The relative importance of denitrification and dissimilatory reduction of nitrate to ammonium in the soil environment

is far from certain. Denitrification may be the dominant process in environments rich in nitrate but poor in carbon, whereas the dissimilatory reduction of nitrate and nitrite to ammonium tends to dominate in carbon-rich environments, which are preferably colonized by fermentative bacteria (Tiedje *et al*., 1982). So nitrate-ammonifying bacteria may be favored by nitrate-limited conditions (Laanbroek, 1990). Nitrate ammonification is found in facultative anaerobic bacteria belonging to the genera *Bacillus, Citrobacter,* and *Aeromonas*, or in the members of Enterobacteriaceae (Cole and Brown, 1980; MacFarlane and Herbert, 1982; Grant and Long, 1985). However, strictly anaerobic bacteria belonging to the genus *Clostridium* are also able to reduce nitrate to ammonia (Caskey and Tiedje, 1979, 1980). For many of the bacteria responsible for dissimilation to ammonium, formate is a major electron donor both for nitrate and nitrite, although most of the research on the nitrate reductase activity has been restricted to enteric bacteria such as *Escherichia coli* (Killham, 1994).

Conversion of $NO_3^-$ to $NH_4^+$ and organic nitrogen increases markedly with decreasing redox potential, high pH, and large quantities of readily oxidizable organic matter (Nommik, 1956; Buresh and Patrick, 1978, 1981). Nitrate respiration to $NH_4^+$ occurs at $E_h$ values of less than −100 mV (Patrick, 1960; Buresh and Patrick, 1981).

## Wetland Environments—Effects of Vegetation

Wetland vegetation influences nitrogen supplies because of uptake associated with growth, which is the topic of a later section. However, vegetation also serves other functions in nitrate reduction, including carbon supply and microbial attachment sites. Wetlands may contain emergent or submergent vegetation, and areas of unvegetated open water. Plants may be woody or soft-tissued. Community specificity for denitrification is expected, roughly correlated with carbon availability and the amount of immersed surface area.

Unvegetated open water does not promote denitrification, resulting in rate constants about one third of those for vegetated systems (Arheimer and Wittgren, 1994). Smith *et al*. (2000) have shown nitrate removal proportional to number of shoots in a *Schoenoplectus* spp. wetland. Wetlands with woody species—shrubs and trees—also have relatively low rates of denitrification (Westermann and Ahring, 1987; DeLaune *et al*., 1996). Carbon limitation is the likely cause.

Either emergent or submergent vegetation can harbor epiphytic microbial biofilms on living and dead plant material (Eriksson and Weisner, 1997). However, living underwater plants produce oxygen, which inhibits denitrification. Field data do not provide clear guidance on the choice between emergent and submergent plants. Weisner *et al*. (1994) found *Potamageton* to be more effective than *Glyceria*, and *Phragmites* stands to be better than open water. Eriksson and Weisner (1997) measured very high rates of denitrification in a reservoir with dense *Potamageton pectinatus*. Conversely, Gumbricht (1993a) found low rates for *Elodea canadensis*. Toet (2003) found that emergent stands of *Typha* and *Phragmites* yielded nitrate removal rates of 98 and 287 kg/ha·yr, respectively, whereas mixed submerged aquatics (*Elodea, Potamogeton* and *Ceratophyllum*) removed only 16–20 kg/ha·yr.

These considerations lead to the conclusion that fully vegetated marshes with either emergent or submergent communities are the preferred option for denitrification. Weisner *et al.* (1994) reached this conclusion and suggested that an alternating banded pattern perpendicular to flow would additionally provide hydraulic benefits.

Denitrifying bacteria are more abundant than the nitrifiers, in both FWS and SSF treatment wetlands. Listowel results show higher populations in the sediments in spring and summer, about $10^6$/g versus $10^5$/g in fall and winter (Herskowitz, 1986). Denitrifiers were found at higher levels in a U.K. gravel bed, approximately $10^7$–$10^8$/g; and most were associated with roots rather than the gravel (May *et al*., 1990).

## Sulfur-Driven Autotrophic Denitrification

Sulfur-driven autotrophic denitrification, as an alternate to carbon-driven, heterotrophic denitrification, is well known (Koenig and Liu, 2001; Soares, 2002). The bacterium *Thiobacillus denitrificans* can reduce nitrate to nitrogen gas while oxidizing elemental sulfur, or reduced sulfur compounds including sulfide ($S^{2-}$), thiosulfate ($S_2O_3^{2-}$), and sulfite ($SO_3^{2-}$). For example, the chemistry proposed for utilization of elemental sulfur is (Batchelor and Lawrence, 1978):

$$NO_3^- + 1.1\,S + 0.40\,CO_2 + 0.76\,H_2O + 0.08\,NH_4^+ \rightarrow 0.5\,N_2 + 0.08\,C_5H_7O_2N + 1.1\,SO_4^{2-} + 1.2\,H^+ \quad (9.28)$$

If sulfide is the primary species of reduced sulfur, the proposed chemistry is (Komor and Fox, 2002):

$$NO_3^- + 0.74\,S^{2-} + 0.1886\,CO_2 \rightarrow 0.48\,N_2 + 0.74\,SO_4^{2-} + 0.037\,C_5H_7O_2N + 0.1\,H^+ + 0.37\,H_2O \quad (9.29)$$

This reaction requires 1.69 g sulfide sulfur per gram of nitrate nitrogen. Other postulated reactions also exist. For instance, iron pyrite may be oxidized (Pauwels and Talbo, 2004):

$$14\,NO_3^- + 5\,FeS_2 + 4\,H^+ \rightarrow 7\,N_2 + 5\,Fe^{2+} + 10\,SO_4^{2-} + 2\,H_2O \quad (9.30)$$

Treatment wetlands can have many forms of sulfur in sediments, arising from the introduction of sulfate in the incoming water. Reducing conditions can form sulfides and elemental sulfur in the sediments (see Chapter 11). Those sediments also contain carbon compounds, and consequently both heterotrophic, carbon-driven, and sulfur-driven denitrification have been observed to occur simultaneously in wetland sediments (Nahar *et al*., 2000; Komor and Fox,

2001, 2002; Wass, 2003). The production of dinitrogen gas is accompanied by oxidation of sulfide to sulfate by the autotrophic process.

Given the variety of alternate electron acceptors for denitrifying organisms, it is not surprising that carbon is not limiting in some wetland situations where it would be expected (Fleming-Singer and Horne, 2006).

### Aerobic Denitrification

Nitrite reduction to gaseous products by denitrifying bacteria used to be considered to be a strictly anaerobic process, but this fallacy was dispelled with the discovery of aerobic denitrification (Robertson *et al*., 1995). Aerobic denitrification is often coupled to heterotrophic nitrification in one organism. Because nitrification is mostly measured by the formation of nitrate or nitrite under oxic conditions, although (aerobic) denitrification is not expected under such conditions, this coupled process is not easily observed in standard enrichment cultures. The observation that *Thiosphaera pantotropha* and other organisms are not only heterotrophic nitrifiers, but also aerobic denitrifiers forced a reevaluation of this approach (Ludwig *et al*., 1993; Jetten, 2001). Aerobic denitrifiers are present in high number in natural soil samples. Even though the specific activities are not always very high, they are sufficient to allow significant contribution to the turnover of compounds in the nitrogen cycle (Jetten *et al*., 1997).

### Anaerobic Ammonia Oxidation (Anammox)

There is now solid evidence for anaerobic elimination of nitrite by ammonia, also called *anaerobic ammonia oxidation* (anammox), in a number of wastewater treatment environments (van de Graaf *et al*., 1990; Mulder *et al*., 1995; van Loosdrecht and Jetten, 1998). In an environment with nitrite and ammonia present, a reaction to dinitrogen has been demonstrated commercially:

$$NH_4^+ + NO_2^- \xrightarrow[\textit{Nitrosomonas eutropha}]{\textit{Planctomycetes}} N_2 + 2H_2O \qquad (9.31)$$

The overall chemistry, including nitrite formation and bacterial growth requirements, has been proposed to be (Furukawa *et al*., 2001):

$$NH_3 + 0.85\,O_2 \rightarrow 0.44\,N_2 + 0.11\,NO_3^- + 1.43\,H_2O + 0.14\,H^+ \qquad (9.32)$$

The process proceeds through nitrite, formed according to Equations 9.22 and 9.23, and carries an oxygen requirement of only 1.94 g O per gram of $NH_4$–N. It is autotrophic, and has no organic carbon requirement.

Various commercial processes are now available which capitalize on the advantages of this alternative route for nitrogen removal. The completely autotrophic nitrogen removal over nitrite (CANON) process utilizes partial nitritation accompanied by Anammox® in a single vessel (Third *et al*., 2005). The SHARON® Anammox process utilizes partial nitritation in one vessel, and anaerobic elimination of nitrite by ammonia in a second (van Dongen *et al*., 2001). The microbiology has also been demonstrated in sequencing batch reactors (Kuai and Verstraete, 1998; Strous *et al*., 1998; Sliekers *et al*., 2002), activated sludge (Hao and van Loosdrecht, 2004), and rotating biological contactors (RBCs) (Helmer and Kunst, 1998; Koch *et al*., 2000).

Given advances in the ability to search for and detect nitrogen processing organisms, they have also been found in natural treatment systems. Anammox bacteria are present in soil aquifer treatment (Fox and Gable, 2003; Gable and Fox, 2003). They have also been identified in both FWS and SSF wetlands. Austin *et al*. (2003) found 13% of *Plantomycetes* in a vertical flow SSF wetland, of which a small fraction were autotrophic denitrifiers. They were also found in SSF and FWS wetlands treating partially nitrified domestic wastewater (Shipin *et al*., 2004).

The importance of this alternative pathway for ammonia and oxidized nitrogen removal for treatment wetland analysis lies in the reduced carbon and oxygen requirements: less than half the oxygen and no carbon, compared to conventional routes. In many wetland situations, there is adequate oxygen present to allow traditional nitrification (Equations 9.20 and 9.21). Likewise, in other instances, there is adequate carbon present to fuel traditional denitrification (Equation 9.27). But there are wetlands for which ammonia and oxidized nitrogen are removed in amounts that considerably exceed the estimated supplies of carbon and oxygen. Tanner and Kadlec (2002) found ammonia losses that would have required far more oxygen transfer than could reasonably be expected in a VF (saturated upflow) system, and Sun and Austin (2006), demonstrated similar results for highly loaded VF (saturated downflow) columns, while Bishay and Kadlec (2005) found the same for an FWS wetland. In the latter case, nitrite was present in relatively large quantities, and the carbon supply was not adequate to support traditional denitrification. In these instances, Anammox offers a potential explanation, but has not been confirmed.

### Nitrogen Fixation

Biological nitrogen fixation is the process by which nitrogen gas in the atmosphere diffuses into solution and is reduced to ammonia nitrogen by autotrophic and heterotrophic bacteria, cyanobacteria (blue-green algae), and higher plants. The reduction of gaseous nitrogen ($N_2$) to ammonia ($NH_3$) takes place very rapidly and for this reason the individual steps in the reaction have not been investigated in detail. It is supposed that the whole reaction is a three-step, two-electrons-per-step mechanism (Winter and Burris, 1976):

$$N \equiv N \rightarrow \underset{\textit{diimide}}{HN = NH} \rightarrow \underset{\textit{hydrazine}}{H_2N - N_2H} \rightarrow NH_3 \qquad (9.33)$$

There are six main types of $N_2$-fixing organisms that can be found in soil (Killham, 1994):

1. Free-living bacteria such as *Bacillus*, *Klebsiella,* and *Clostridium* that fix $N_2$ anaerobically (the first two are facultative anaerobes and fix nitrogen under reduced oxygen tensions whereas *Clostridium* is an obligate anaerobe)
2. Bacteria of the genus *Rhizobium*, which fix $N_2$ mainly in the root nodules of leguminous plants
3. Actinomycetes of the genus *Frankia*, which fix $N_2$ in the root nodules of nonleguminous angiosperms such as *Alnus glutinosa* (those associations are often referred to as "actinorhizas")
4. Free-living cyanobacteria on the soil surface such as *Nostoc* and *Anabaena*
5. Symbiotic cyanobacteria found in the lichen symbiosis
6. $N_2$-fixing bacteria loosely associated with the roots of certain plants, sometimes referred to as "rhizocoenoses" (e.g., *Azotobacter*, *Beijerinckia* and *Azospirilllum*)

In wetland systems, free-living bacteria, cyanobacteria (blue-green algae), $N_2$-fixing bacteria loosely associated with the roots of certain plants, and probably *Frankia* are the most important $N_2$-fixing organisms.

Also, the aquatic fern, *Azolla*, and a few transitional, wetland vascular plant species in the genera *Alnus* and *Myrica* have been observed to fix atmospheric nitrogen (Waughman and Bellamy, 1980). Because nitrogen fixation uses stored energy from either autotrophic or heterotrophic sources, it is not an adaptive process when nitrogen is otherwise available for growth. The presence of ammonium nitrogen is reported to inhibit nitrogen fixation (Postgate, 1978; as referenced by Van Oostrom and Russell, 1994).

Under anaerobic conditions, microbial assemblages in the root zone of *Typha* spp. and *Glyceria borealis* were shown to fix considerable quantities of atmospheric nitrogen (Bristow, 1974). The majority of the activity was shown to be associated with the plants rather than the soils. Fixation rates at 20°C were determined to be 33.6 and 353 mg/kg roots·day for *Typha* and *Glyceria*, respectively. The measured rates of nitrogen fixation were estimated to be able to supply 10–20% of the growth requirement for *Typha*, and 100% for *Glyceria*. Under aerobic conditions, fixation dropped by an order of magnitude.

The nitrogen fixation potential for the soil-microbe assemblage was studied for 45 sites in 17 peatlands in eight countries by Waughman and Bellamy (1980). The appropriate subset in the context of treatment wetlands was the rich or extremely rich fen category, with $6.5 \leq pH \leq 7.6$, for which $N = 12$ sites. These showed fixation potentials averaging 0.622 mg/L per day of soil. A 30-cm root zone would then fix 70 $gN/m^2 \cdot yr$. Other estimates from natural freshwater wetlands range from 0 to 55 $gN/m^2 \cdot yr$ (Vymazal, 2001b). Estimates of nitrogen fixation in a cypress dome receiving municipal wastewater ranged from 0.012 to 0.19 $g/m^2 \cdot yr$ (Dierberg and Brezonik, 1984) and were concluded to be an insignificant component of the TN loading to this treatment wetland.

These results do not permit quantification of the fixation occurring in treatment wetlands, but do indicate the ability of wetland plants and soils to fix nitrogen. It is unlikely that the rates of fixation in treatment wetlands contribute materially to nitrogen cycling in nitrogen-rich systems.

## 9.4 VEGETATION EFFECTS ON NITROGEN PROCESSING

Plants utilize nitrate and ammonium, and decomposition processes release nitrogen back to the water. There are two direct effects of vegetation on nitrogen processing and removal in treatment wetlands:

- The plant growth cycle seasonally stores and releases nitrogen, thus providing a "flywheel" effect for a nitrogen removal time series.
- The creation of new, stable residuals accrete in the wetland. These residuals contain nitrogen as part of their structure, and hence accretion represents a burial process for nitrogen.

On an instantaneous basis, plant uptake can be important for many wetland systems. A benchmark instantaneous growing season rate is suggested to be 120 $gN/m^2 \cdot yr$ (Kadlec, 2005d). The majority of the assimilated nitrogen is subsequently released during death and decay, but a small amount is permanently stored as new soil and sediment. The net removal of ammonia to accretion, via the vegetative cycle, is on the order of 10 $gN/m^2 \cdot yr$. This amount is of great importance for very lightly loaded wetlands, but of no importance for heavily loaded systems.

The two forms of nitrogen generally used for assimilation are ammonia and nitrate nitrogen. Nitrate uptake by wetland plants is presumed to be less favored than ammonium uptake. But in nitrate rich waters, nitrate may become a more important source of nutrient nitrogen. Aquatic macrophytes utilize enzymes (nitrate reductase and nitrite reductase) to convert oxidized nitrogen to useable forms. The production of these enzymes decreases when ammonium nitrogen is present (Melzer and Exler, 1982). Plants such as cattails (*Typha latifolia*) are very able to utilize either nitrate or ammonia (Brix *et al*., 2002b), and so are algae (Naldi and Wheeler, 2002) and cultivated rice (Kronzucker *et al*., 2000). Dhondt *et al*. (2003) found that about half of the applied nitrate in a riparian wetland was utilized by plants, whereas half was denitrified.

In the Santee, California, study of a *Scirpus*/gravel HSSF wetland (Gersberg *et al*., 1984), the entire nitrate loss was ascribed to plant uptake in the absence of an exogenous carbon source and with essentially no ammonium in the nitrified influent. This process may also be important in other

treatment wetlands. For instance, a short-term $^{15}N$ study of several SSF gravel wetland mesocosms (Zhu and Sikora, 1994) showed 70%–85% of the entire nitrate loss was plant uptake—in the absence of an exogenous carbon source and with essentially no ammonium in the nitrified influent. Different species responded differently: 70% of the nitrate was taken up by *Phragmites australis*, 75% by *Typha latifolia*, and 85% by *Scirpus atrovirens georgianus*. In the absence of definitive results on the proportions of nitrate versus ammonia uptake in treatment wetlands, some authors have opted to presume these are utilized in proportion to the quantities in the water (Martin and Reddy, 1997; Tanner *et al*., 2002a). However, process factors argue against this simple expectation. First, plants extract their nitrogen requirements via their root system, which is predominantly located in the wetland soil, with the possible exception of adventitious roots, which occur in the water column. Nutrients reach the subsurface root system via diffusion under appropriate circumstances, but more importantly via transpiration flux, the vertical water flow driven by the transpiration requirement of the plant (see Chapter 4). The upper soil horizon that contains the roots is typically anoxic and has a high carbon content, and therefore is capable of supporting denitrification (Crumpton *et al*., 1993). Nitrate that moves downward toward the root zone is therefore unlikely to survive in the same proportion as it exists in the water column above the soil.

## The Effects of Vegetation Growth and Cycling

The removal of ammonia from water by wetland plants has been the subject of many studies (e.g., Reddy and DeBusk, 1985; Rogers *et al*., 1991; Busnardo *et al*., 1992; Tanner, 1996). Many such studies have been characterized by measurements of gross nitrogen uptake, with no deduction for subsequent losses due to plant death and decomposition, with the attendant leaching and resolubilization of nitrogen.

From the standpoint of nitrogen removal from wetland water, it is the net effect of the macroflora on water phase concentrations that is of interest. Here the terminology of Mueleman *et al*. (2002) will be used (see Figure 3.7):

- *Phytomass* refers to all vegetative material, living plus dead.
- *Biomass* refers to all living vegetative material.
- *Necromass* refers to all dead vegetative material.

The seasonal patterns of vegetation growth and nitrogen storage embody complex patterns of biomass allocation among plant parts, as well as the nitrogen content of those various portions of living and dead material. However, from the point of view of the annual ecosystem removal of nitrogen, uptake and return from the combination of biomass and necromass are the principal features of concern. On an annual average basis, the only concern is net removal to permanent storage. However, during the course of the year, uptake and return may occur at different times, thus influencing removals differently in different seasons. For these reasons, it is necessary to examine the transfers to and from the collective parts of the macrophytes, which is here defined as phytomass. During the course of the year, especially in temperate climates, phytomass increases during the growing season, and shrinks during the senescence season. The same pattern is followed by phytomass nitrogen.

### A Mass Balance Framework

The purpose here is to make order-of-magnitude assessments of the role of vegetation in the overall set of ammonia nitrogen processes. This choice has the effect of establishing a "green and brown box," which interacts with the balance of the wetland ecosystem (see Figure 3.7). The nitrogen mass balance for that box is (instantaneously)

$$(J_u - J_r - J_b) = \frac{d\mathrm{N}}{dt} \tag{9.34}$$

or for a fixed time period $\Delta t$:

$$(J_u - J_r - J_b)\Delta t = \Delta \mathrm{N} \tag{9.35}$$

where

$J_u$ = uptake of nitrogen by phytomass ( $= U_e$), gN/m$^2$·d
$J_r$ = release of nitrogen from phytomass ( $= L + D_a + D_b$), gN/m$^2$·d
$J_b$ = burial of nitrogen from phytomass ( $= A_a + A_b$), gN/m$^2$·d
$d\mathrm{N}/dt$ = storage change rate of nitrogen in phytomass, gN/m$^2$·d
$\Delta \mathrm{N}$ = increase in nitrogen storage in phytomass, gN/m$^2$
$\Delta t$ = time interval, d

The uptake of nitrogen is via the root system, which is usually belowground. Nitrogen must therefore be transported into the rhizosphere, by processes of diffusion (minor) and vertical movement driven by transpiration flux (major) (Reddy *et al*., 2005). Some of the new plant growth nutrient requirement is supplied by translocation from stores in the rhizomes, and some from uptake from pore water. It is possible that the presence of nitrogen-rich pore waters causes less withdrawal from rhizomes, and causes lesser storage in belowground tissues (Tanner, 2001a).

Nitrogen is returned to surface waters and pore waters by leaching and decomposition. It is likely that the majority of nitrogen in the necromass is returned, with lesser amounts transferred to permanent burial in the form of new soils and sediment. Over the course of a full calendar year, for a repetitively stable ecosystem, there is no change in the total phytomass, and $\Delta \mathrm{N} = 0$. For that annual period, plant uptake is either returned (more) or buried (less). But, as can be seen from Figure 9.11, the total phytomass nitrogen grows in spring and early summer, and recedes in autumn. This annual cycle is more pronounced in cold climates, in response to the more pronounced seasonal conditions.

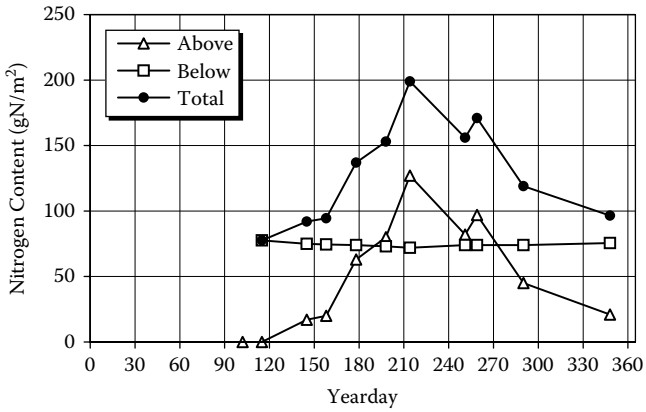



**FIGURE 9.11** Seasonal patterns of nitrogen in *Phragmites australis* in the Netherlands for a fertilized stand. (Data from Mueleman *et al*. (2002) *Wetlands*, 22(4): 712–721.)

At this point in the development of knowledge about wetland plant nitrogen cycling, there is some good idea of the change in storage (ΔN) for a given time interval, but less about the three individual fluxes that lead to the storage ($J_u$, $J_r$, $J_b$).

### A Speculative Numerical Assessment

The green and brown box, consisting of all phytomass nitrogen, expands during the growing season, and contracts during the balance of the year. The purpose here is to assess the approximate magnitude of these nitrogen withdrawals and returns upon the amount of ammonia nitrogen in the water column. Some useful insights may be gained by speculatively assigning uptake and burial (Kadlec, 2005b). These are:

1. A fixed proportion of the necromass nitrogen that returns to water.
2. A constant rate of burial ($J_b$) apportioned to the unfrozen season.
3. Nitrogen release driven by the amount of necromass during the unfrozen season.

As an order-of-magnitude illustration, an annual phytomass nitrogen cycle is presumed to follow a smoothed version of Figure 9.11. An annual accretion of 20 gN/m$^2$·yr is proposed. This is apportioned over a growing season (unfrozen) of eight months, at a constant rate of 2.5 gN/m$^2$·mo. Four times that amount, 80 gN/m$^2$·yr, is presumed to be returned to water. Growth begins at the end of April, and ends in December, causing nitrogen uptake from April through August, totaling 156 gN/m$^2$. During September through December, 56 gN/m$^2$ is returned from senescing and decaying necromass from the current year. TN return is 80 + 56 = 136 gN/m$^2$ for the year, or 87% of the uptake. Only 13% of the nitrogen uptake finds its way into recalcitrant residual forms. However, during the spring growth period, the entire external nitrogen loading is consumed to create the standing crop. These seasonal effects are summarized in Figure 9.12. The loading to the wetland was 240–270 gN/m$^2$·yr. Thus, it is seen that vegetative transfers make up major fractions of the external load.

Treatment wetland data show growing season vegetative uptakes of 20–100 gN/m$^2$, which occurs during a four- to six-month period in temperate climates. This results in growing season uptake rates of 40–200 g/m$^2$·yr. A median benchmark uptake loading of 120 g/m$^2$·yr has been selected here as a basis for evaluating external loadings. Examination of a large number of operational data sets for FWS wetlands leads to the conclusion that emergent and submergent plants are important contributors to the processing of ammonia in free water surface wetlands, for about half of the existing systems (Kadlec, 2005d). For instance, nitrogen storage in the roots and rhizomes in the inlet zone of a FWS *Phragmites/Typha* treatment wetland in Byron Bay, Australia, was 35 g/m$^2$; in the leaves and stems it was 92 g/m$^2$ (Adcock *et al*., 1995). Approximately 65% of the nitrogen added to this treatment wetland was found in the macrophyte biomass, due to low nitrogen loading (approximately 25–40 g/m$^2$·yr).

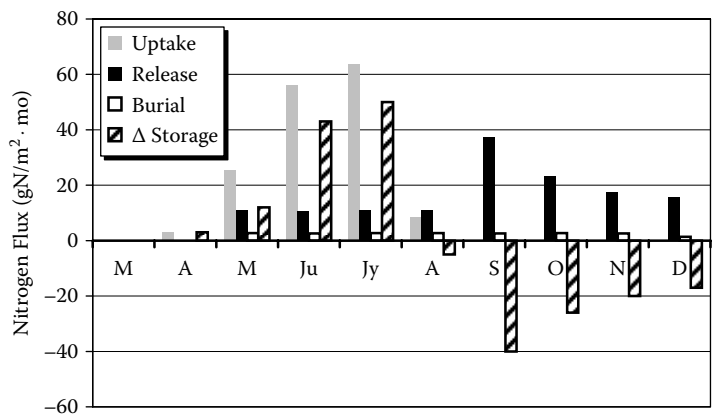



**FIGURE 9.12** Hypothetical seasonal transfers of nitrogen corresponding to the measured growth pattern of Figure 9.11. The loading to the wetland was 240–270 gN/m$^2$·yr. (Data from Mueleman *et al*. (2002) *Wetlands*, 22(4): 712–721.)

**TABLE 9.6**
**Accretion Rates in FWS Wetlands**

| Location | Reference | Method | Water $NH_4$–N (Typical) (mg/L) | Accretion (cm/yr) | Nitrogen Burial ($gN/m^2 \cdot yr$) |
|---|---|---|---|---|---|
| Louisiana | Rybczyk *et al*. (2002); 400–500 gC/gSoil; 2.0% N[a] | Feldspar | 0.05 | 0.14 | — |
| Michigan | Kadlec and Robbins (1984) | Lead 210 | 0.1 | 0.2 | — |
| Everglades WCA2A | Reddy *et al*. (1991); 300–500 gC/gSoil; 3.0% N | Cesium 137 | 0.3 | 0.5 | 9 |
| Everglades WCA2A | Craft and Richardson (1993a,b); 450 gC/gSoil; 3.2% N | Cesium 137 | 0.3 | 0.4 | 11.6 |
| Everglades WCA3 | Craft and Richardson (1993a,b); 450 gC/gSoil; 3.2% N | Cesium 137 | 0.1 | 0.3 | 10.7 |
| Everglades, Florida | Robbins *et al*. (1999); 3.0% N[a] | Lead 210 | 0.3 | 0.5 | 11 |
| Everglades, Florida | Chimney (2000), unpublished data; 500 gC/gSoil; 3.2% N | Feldspar | 0.1 | 0.85 | 35 |
| Iron Bridge, Florida | Miner *et al*. (2002) | Visual | 0.1 | 1.17 | — |
| Louisiana | Rybczyk *et al*. (2002); 400–500 gC/gSoil; 2.0% N[a] | Feldspar | 15 | 1.14 | 23 |
| Sacramento, California | Nolte and Associates (1998b); 4.3% N | Visual | 16 | 1.5 | 44 |
| Houghton Lake, Michigan | Kadlec (1997); 400–500 gC/gSoil; 3.2% N | Resurvey | 10 | 1.8 | 56 |
| Chiricahueto, Mexico | Soto-Jiménez *et al*. (2003); 10–40 gC/gSoil; 0.3% N | Lead 210 | 14 | 1.0 | 1.5 |

[a] Assumed value.

### Accretion of Nitrogenous Residuals

The least studied aspect of nitrogen transfer in wetlands is in the creation of new soils and sediments, with their attendant nitrogen content. Not all of the dead plant material undergoes decomposition. Some small portions of both aboveground and belowground necromass resist decay, and form stable new accretions. Such new stores of nitrogen are presumed to be resistant to decomposition. The origins of new sediments may be from remnant macrophyte stem and leaf debris, remnants of dead roots and rhizomes, and from undecomposable fractions of dead microflora and microfauna (algae, fungi, invertebrates, bacteria).

The amount of such accretion has been quantified in only a few instances for free water surface wetlands (Reddy *et al*., 1991; Craft and Richardson, 1993a,b; Rybczyk *et al*., 2002), although anecdotal reports also exist (Kadlec, 1997a). Quantitative studies have relied upon either atmospheric deposition markers (radioactive cesium or radioactive lead), or introduced horizon markers, such as feldspar or plaster. Either technique requires several years of continued deposition for accuracy.

Reddy *et al*. (1991) used $^{137}Cs$ to estimate the rate of accretion in a mildly fertilized cattail wetland in Florida, which ranged from approximately 5 to 11 mm/yr of low bulk density material, less than 0.1 $g/cm^3$. The nitrogen content of these new accretions was measured to be approximately 3%, resulting in annual accretion rates of 11–24 $gN/m^2 \cdot yr$. Murkin *et al*. (2000) found 4.5–6.5 $gN/m^2 \cdot yr$ annual accretion rates for low nutrient, mixed marshes in Manitoba. Soto-Jiménez (2003) reported net sedimentation of nitrogen of 11.3 $gN/m^2 \cdot yr$ for a marsh receiving strong agricultural runoff. Hocking (1989b) estimated 8 $gN/m^2 \cdot yr$ annual accretion rate for *Phragmites australis* in a nutrient-rich Australian setting. Klopatek (1978) estimated 5 $gN/m^2 \cdot yr$ annual accretion rate for a *Schoenoplectus* (*Scirpus*) *fluviatilis* stand. Representative accretion rates are given in Table 9.6.

The manner of accretion has sometimes been presumed to be sequential vertical layering (Kadlec and Walker, 1999; Rybczyk *et al*., 2002), but that view is likely to be overly simplified. At least two factors argue against simple layering: vertical mixing of the top soils and sediments (Robbins *et al*., 1999), and the injection of accreted root and rhizome residuals at several vertical positions in the root zone. Nonetheless, new residuals are deposited on the wetland soil surface from various sources. The most easily visualized is the litterfall of macrophyte leaves, which results in top deposits of accreted material after decomposition. However, algal and bacterial processing which occurs on submersed leaves and stems results in litterfall and accretion of micro-detrital residuals.

### Short-Term Anomalies

In addition to the considerations of long-term repetitive annual vegetation effects on wetland nitrogen processing, there are transient effects related to start-up of treatment wetlands. These transient events are different from the stable annual pattern of swelling and shrinking of the phytomass nitrogen storage. Results from transient studies must not be construed as being representative of long-term patterns. Some case study transient results are informative.

### FWS Mesocosm Start-Up

Busnardo *et al.* (1992) operated FWS mesocosms vegetated with *Scheonoplectus* (*Scirpus*) *californicus.* The ammonia loading rates to the mesocosms were 330 and 670 $gN/m^2 \cdot yr$ for two consecutive seven-month periods. Approximately 60% of the ammonia nitrogen removed was found in plant growth.

Although this experiment demonstrated that emergent macrophytes have the capacity to assimilate large quantities of ammonia, Busnardo *et al.* (1992) speculated that plants would have a lesser effect in mature wetlands.

### SSF Mesocosm Start-Up

A number of studies in the literature focus upon newly planted mesocosms, which are monitored for performance during the subsequent period of plant development. For example, Rogers *et al.* (1991) reported on nitrogen processing in 25-L buckets filled with gravel and planted with *Schoenoplectus validus* rhizomes. Studies of ammonia removal commenced five weeks later, and continued for 35 weeks. Ammonia loading rates of 60–600 $gN/m^2 \cdot yr$ were applied over periods of 10–15 weeks. Removals ranged from 90–100%, of which about 90% was found in the vegetation. These rates of uptake are not counteracted by return fluxes, because no necromass was formed over the short duration of the tests. It was eventually found that the plants in the buckets remained in the colonizing mode for at least three years. (Rogers *et al.*, 1991).

### Ammonia Loads to a New Wetland

Newly constructed wetlands are typically planted sparsely compared to the ultimate grow-out of vegetation. The development of the new vegetation creates a nitrogen demand that persists only during that grow-in period. For example, Sartoris *et al.* (2000) reported on the first two years of ammonia removal and plant coverage for a 9.9-ha FWS constructed wetland at Hemet, California. As the plant coverage went from near zero (planted clumps on 1.2-m spacing) to about 80% of *Schoenoplectus* spp., and the vegetation density increased by 67%, the ammonia load removed went from 98 down to 15 $gN/m^2 \cdot yr$. Sartoris *et al.* (2000) concluded that plant uptake was most likely the primary sink for nitrogen during the two-year study. In this case of a FWS wetland, the increase in coverage by plants reduced the fraction of open water, and hence created a lesser potential for atmospheric reaeration to support nitrification.

## Harvest to Remove Nitrogen

Nitrogen removal is theoretically possible via the harvest of plants and their associated nitrogen content. However, aboveground standing crops do not display a large potential for removal of nitrogen, even under the assumption that the entire crop could be recovered (Table 9.7). Based on the productivities given by DeBusk and Ryther (1987), potential nitrogen removal for floating large-leaved plants (*Eichhornia, Pistia, Hydrocotyle*) is in the range of 100–250 $gN/m^2 \cdot yr$, and 50–150 $gN/m^2 \cdot yr$ for floating small-leaved plants (*Salvinia, Lemna, Spirodela, Azolla*).

Direct harvesting experience has shown that only a small fraction of the applied nitrogen can be recovered in harvested biomass (Table 9.7). Systems operating in tropical climates may be capable of greater sustained annual vegetative removals, which are enhanceable by harvest. Koottatep and Polprasert (1997) measured from 70 to 275 $gN/m^2 \cdot yr$, depending upon harvesting frequencies ranging from no harvest to every eight weeks, respectively.

Harvest may involve complete removal in the case of floating plants (*Lemna minor*, *Eichhornia crassipes*), or cutting of aboveground parts of rooted plants such as *Typha*, *Schoenoplectus*, and *Phragmites*. Harvesting typically requires expensive mechanical equipment, and is labor-intensive for large systems. For instance, a one-time harvest of floating mats of *Typha* in a Florida treatment wetland cost approximately $16 per cubic meter of wet material, or about $8 per kilogram of nitrogen removed. However, in the small SSF systems, such as those commonly found in

**TABLE 9.7**
**Amount of Nitrogen in the Standing Aboveground Stock Compared to Nitrogen Loadings**

| Location | Reference | Type | Nitrogen Stock ($gN/m^2$) | Applied Nitrogen ($gN/m^2 \cdot yr$) | Percent Removable |
|---|---|---|---|---|---|
| Ondrejov, Czech Republic | Vymazal *et al.* (1999) | SSF *Phragmites* | 51 | 1,183 | 4.3 |
| Kolodeje, Czech Republic | Vymazal *et al.* (1999) | SSF *Phragmites* | 20 | 493 | 4.1 |
| Chmelna, Czech Republic | Vymazal *et al.* (1999) | SSF *Phalaris* | 26.5 | 1,397 | 1.9 |
| Zasada, Czech Republic | Vymazal *et al.* (1999) | SSF *Phalaris* | — | — | 0.8 |
| Hamilton, New Zealand | Tanner (2001a) | SSF *Schoenoplectus* | 23 | 431 | 5.3 |
| Hamilton, New Zealand | Tanner (2001a) | SSF *Schoenoplectus* | 40 | 1,256 | 3.2 |
| Sacramento, California | Nolte and Associates (1998b) | FWS *Typha* + *Scirpus* | 60 | 360 | 16.5 |
| ENR, Florida | Everglades ENR Cell 1, unpublished data | FWS *Typha* | 4.7 | 8 | 60 |
| Byron Bay, Australia | Adcock *et al.* (1995) | FWS *Leersia* + *Urochloa* | 44 | 203 | 21 |
| Houghton Lake, Michigan | Houghton Lake, Michigan–based 50 ha, unpublished data | FWS *Typha* | 25 | 10 | 250 |
| Malham, United Kingdom | Hurry and Bellinger (1990) | FWS *Phalaris* | 49 | 469 | 11 |
| Duplin County, North Carolina | Hunt *et al.* (2002) | FWS *Typha* | 32 | 392 | 8 |
| Duplin County, North Carolina | Hunt *et al.* (2002) | FWS *Scirpus* | 35 | 420 | 8 |
| Greensboro, North Carolina | Hunt *et al.* (2002) | FWS *Typha* | 20 | 971 | 2 |

Europe, harvesting is easy and forms a negligible amount within the annual O&M costs.

The problem of biomass disposal is often not easily resolved. Harvested biomass may either be composted, or digested to form a biogas product. Composting requires transportation costs, and dedicated land area. Biogas generation from water hyacinths has been shown to be feasible (Biljetina *et al*., 1987; Joglekar and Sonar, 1987); however, sludge disposal remains a problem. The capital cost of harvesting and gas generation about is about the same as for the rest of the wastewater treatment plant, and is thus prohibitively expensive (Chynoweth, 1987). As a consequence of these difficulties, plant harvesting is not favored for nitrogen removal (Crites and Tchobanoglous, 1998), and has seldom been used except for floating plants.

### Soil and Sediment Effects on Nitrogen Processing

Apart from accretion, wetland solids form a large pool of nitrogen, some of which is available for exchange with surface waters and pore waters. As noted above, sorption and cation exchange are active processes in the wetland environment. These nitrogen solid storages will stabilize under continuous operation of a treatment wetland, but are nonetheless active, and exchange compounds with their surroundings. Thus the image of nitrogen compounds traveling with the flowing water is incorrect; nitrogen follows a "park and go" trajectory through the wetland.

Kadlec *et al*. (2005) reported these exchanges for SSF treatment wetlands. Four mesocosm trains and one field-scale wetland contained well-established bulrushes (*Schoenoplectus tabernaemontani*), and another field-scale wetland remained unvegetated. The systems were operated at steady inflows, with a nominal detention times of four to five days. The incoming ammonium nitrogen ranged from 18.5–177 g/m$^3$, and removals ranged from 15% to 90% for the various feed waters. Each system was dosed with a single pulse of $^{15}$N ammonium mixed into the feed wastewater, and the fate and transport of the isotopic nitrogen were determined. The $^{15}$N pulses took 120 days to clear the heavily loaded field-scale wetlands. During this period small reductions in $^{15}$N were attributable to nitrification/denitrification, and a larger reduction due to plant uptake. Mesocosm tests ran for 24 days, during which only 1–16% of the tracer exited with water, increasing with nitrogen loading. Very little tracer gas emission was found, about 1%. The majority of the tracer was found in plants (6–48%) and sediments (28–37%). These results indicated a rapid absorption of ammonium into a large sediment storage pool, of which only a small proportion was denitrified during the period of the experiment. Plant uptake claimed a fraction of the ammonium, determined mainly by the plants requirement for growth rather than the magnitude of the nitrogen supply. A rapid return of ammonium to the water was also found, so that movement of $^{15}$N through the wetland mesocosms comprised a "spiral" of uptake and release along the flow path.

## 9.5 NITROGEN MASS BALANCES

The individual process considerations discussed above may be combined to form the integrated concept of nitrogen fluxes in treatment wetlands. This interpretive step is very important, because it

1. Identifies the true rates of ammonification, ammonia oxidation, and denitrification.
2. Places the role of the vegetative nitrogen cycle in the context of the microbial processes.
3. Allocates the fate of added nitrogen to storage, leakage, and gasification.

The use of the percent removal measure may be very misleading for separate nitrogen species. For example, U.S. EPA (1993f) found that approximately half of the SSF wetlands inventoried had negative percent removals for ammonia. In the absence of speciated nitrogen mass balances, that technology assessment ascribed the good performance to lack of algae, oxygen availability and long detention, and poor performance to short rooting depth and oxygen deficiency. However, in the absence of adequate data on ammonification, U.S. EPA (1993f) dismissed that process as not being a contributing factor. Much more information is now available, and it is possible to examine the nitrogen interconversions in more detail.

### Mass Balance Case Studies

Only a few wetland studies have reported mass balances for the interrelated species of nitrogen (Tanner and Kadlec, 2002; Senzia *et al*., 2002b; Bishay and Kadlec, 2005; Kadlec *et al*., 2005). In all cases, the involvement of vegetation in the nitrogen cycle is somewhat speculative, because it depends upon estimates of biomass and tissue nitrogen content. Nonetheless, much is known about standing stocks and turnover rates, as well as the (narrow) bounds on nitrogen percentages in that biomass. Here three examples of FWS wetland nitrogen mass balances will be explored: (1) a lightly loaded polishing wetland, (2) a leaky wetland treating contaminated river water, and (3) a seasonal wetland treating nitrogenous mine wastewaters. In each case, long-term performance is examined, and consequently seasonal effects are not elucidated. One example of mass balance for an HSSF wetland is presented as well.

### Orlando Easterly, Florida, FWS Wetland

This treatment wetland has been in operation since 1987, and is described in general terms in U.S. EPA (1993a). It is a 494-ha constructed free water surface wetland with 17 compartments in a series and parallel arrangement, which receives about 60,000 m$^3$/d of highly treated municipal effluent. The cells were vegetated with soft-tissue emergent plants, and the vegetative communities evolved over time to a mixed marsh condition. In addition to annual and specialty project reports,

**TABLE 9.8A**
**Average Inlet and Outlet Concentrations for the Orlando Easterly, Florida, FWS Wetland for 1993–2002**

| Parameter | Inlet (mg/L) | Outlet (mg/L) | Mean (mg/L) | Fraction | Assumed Rain (mg/L) |
|---|---|---|---|---|---|
| HLR, cm/d | 1.17 | 1.15 | — | — | 0.41 |
| Organic N | 1.67 | 0.98 | 1.32 | — | 1.0 |
| Ammonia N | 0.33 | 0.14 | 0.23 | 0.545 | 0.5 |
| Nitrite N | 0.60 | 0.04 | 0.32 | — | — |
| Nitrate N | 0.26 | 0.13 | 0.19 | 0.455 | 0.5 |
| TSS | 1.2 | 4.0 | — | — | — |
| $CBOD_5$ | 2.0 | 2.6 | — | — | — |
| DO | 6.1 | 8.9 | — | — | — |
| Alkalinity | 94 | 92 | — | — | — |

*Source:* City of Orlando operating data.

there have been several published papers (Jackson, 1989; Jackson and Sees, 2001; Martinez and Wise, 2003a,b; Wang *et al*., 2006a,b). Data used here are from the ten-year period 1993–2002.

Nitrogen totals less than 3 mg/L entering the system, and less than 1.4 mg/L in the effluent from the wetland. Atmospheric contributions are not negligible under these circumstances, and are estimated at 2.0 mg/L based upon other Florida data. The inlet hydraulic loading was 1.2 cm/d, and rainfall averaged about 0.4 cm/d (Table 9.8A). Particulate nitrogen is not a factor, because the TSS content of the incoming water is very low (1.2 mg/L). The data combine to produce a TN inlet loading of 11.3 $gN/m^2{\cdot}yr$, apportioned across the species as indicated in Figure 9.13. This is much less than the required nitrogen for even modest plant growth, indicating that the vegetative cycle must draw upon internal sources of nitrogen. There was net removal of all forms of nitrogen, summing to a 70% reduction in the load of TN. The inlet–outlet concentration reduction was less, 55%, because it does not include the contribution of rainfall nitrogen.

Since measurements were not made of vegetative nitrogen processes, assumptions must be made. The wetland was moderately well vegetated, with some open water, leading to the assumption of an annual productivity of 1,000 g $dw/m^2{\cdot}yr$ with an assumed nitrogen content of 2%. Of this, 10% was assumed to be buried as new sediments (Table 9.8B). Both nitrate and ammonia were presumed to be used to support growth, in proportion to their availability in the water. Average concentrations were used to determine the uptake ratio, although selective spatial utilization may have occurred.

This information is adequate to calculate all the average annual transfers within the wetland via mass balances. The pattern of nitrogen transfers is dominated by the vegetative cycle (Figure 9.13). Production of ammonia from decomposition of biomass is eight times higher (20.65 $gN/m^2{\cdot}yr$) than the reduction in ammonia in the water from inlet to outlet (2.64 $gN/m^2{\cdot}yr$). Nitritation is seven times higher than the reduction in the flowing ammonia load (11.32 versus 1.57 $gN/m^2{\cdot}yr$), and that high internal load of nitrite is subsequently nitrified to nitrate. Some nitrate is lost through denitrification, but more is used to support

**TABLE 9.8B**
**Assumptions for the Orlando Easterly Wetland Carbon and Oxygen Supplies**

| Assumption | | | Notes |
|---|---|---|---|
| Biomass produced = | 1,000 | g $dm/m^2{\cdot}yr$ | — |
| Carbon content = | 0.5 | — | 50% |
| Useable carbon fraction = | 0.3 | — | 30% |
| Carbon available = | 150 | $g/m^2{\cdot}yr$ | — |
| Denitrification carbon requirement = | 140 | $g/m^2{\cdot}yr$ | $1.07 \times N$ |
| Biomass N uptake = | 20 | $g/m^2{\cdot}yr$ | 2% N |
| Biomass buried = | 100 | g $dw/m^2{\cdot}yr$ | 10% |
| Nitrogen buried = | 2.0 | $g/m^2{\cdot}yr$ | 2% N |
| Oxygen needed = | 36 | $g/m^2{\cdot}yr$ | $3.43 \times$ nitritation + $1.14 \times$ nitrification, plus DO increase |
| Daily oxygen needed = | 0.16 | $g/m^2{\cdot}d$ | — |

*Note:* Biomass is the assumed source of carbon, and oxygen requirements are determined from Figure 9.14 fluxes.

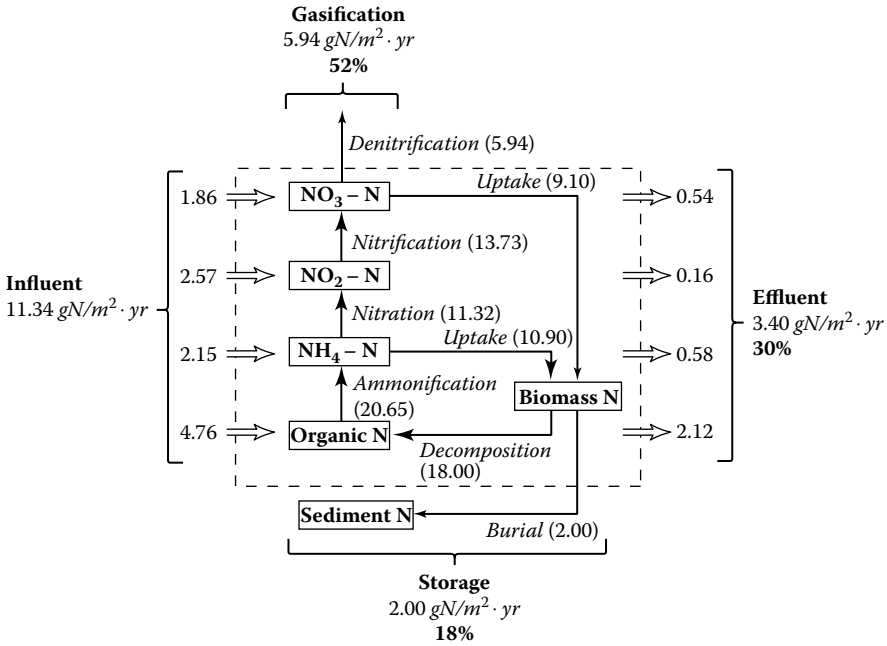



**FIGURE 9.13** Estimated annual nitrogen fluxes in the Orlando Easterly treatment wetland (gN/m²·yr). The vegetation cycle dominates this lightly loaded system.

plant growth. However, denitrification amounts to 52% of the net nitrogen input, whereas accretion of new sediments represents only 18%.

The required supplies of ancillary chemicals were present in the wetland (Table 9.8 A, B). Dissolved oxygen is present to support ammonia oxidation and the observed reaeration, which is calculated to need 0.16 gO/m²·d, well within the range of expected atmospheric reaeration (see Chapter 5). The required alkalinity is also available to support ammonia oxidation. There is no carbon in the inlet water

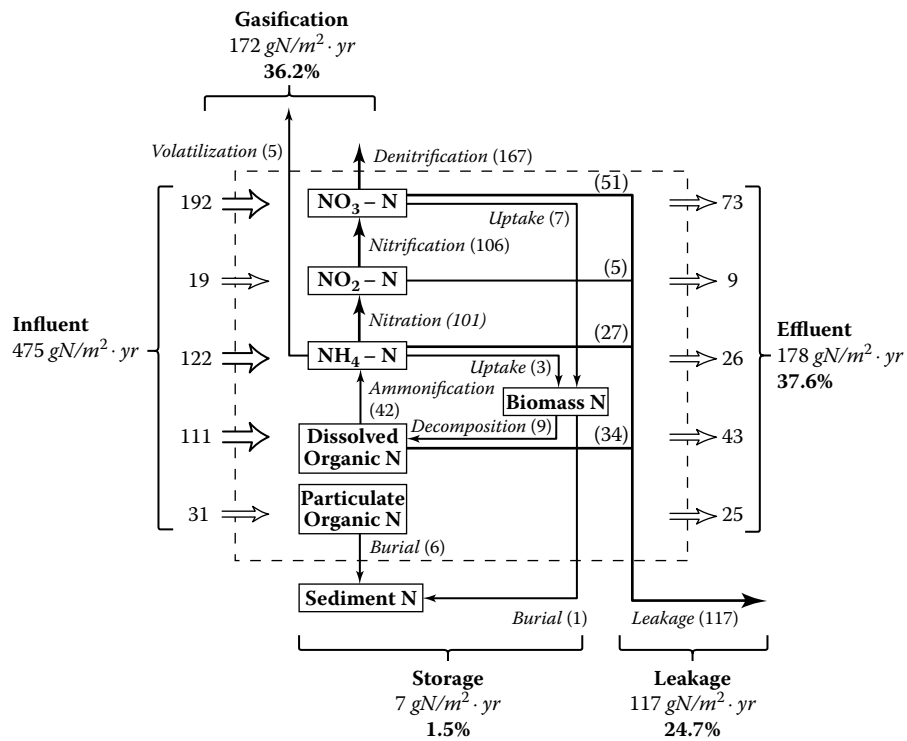



**FIGURE 9.14** Estimated annual nitrogen fluxes in the Imperial, California, demonstration wetland (gN/m²·yr). The vegetation cycle is of little importance in this system.

**TABLE 9.9A**
**Average Inlet and Outlet Concentrations for the Imperial, California, FWS Wetland for 2001–2004**

| Parameter | Inlet (mg/L) | Outlet (mg/L) | Leakage (mg/L) | Fraction |
|---|---|---|---|---|
| HLR, cm/d | 19.3 | 12.5 | 6.8 | — |
| Dissolved organic N | 1.77 | 1.01 | 1.39 | — |
| Particulate organic N | 0.46 | 0.60 | 0.53 | — |
| Ammonia N | 1.64 | 0.53 | 1.09 | 0.344 |
| Nitrite N | 0.25 | 0.18 | 0.22 | — |
| Nitrate N | 2.65 | 1.49 | 2.07 | 0.656 |
| TSS | 179 | 11 | — | — |
| $CBOD_5$ | 7.2 | 5.2 | — | — |
| DO | 8.2 | 7.2 | — | — |
| Sulfate S | 661 | 618 | 640 | — |
| Alkalinity | 239 | 203 | — | — |

*Source:* Imperial Irrigation District operating data.

to support denitrification ($CBOD_5$ = 2 mg/L), but the biomass cycle produces enough available carbon to fuel heterotrophic denitrifiers.

## Imperial, California, FWS Wetland

This FWS treatment wetland system has been in operation since 2000, and the data used here are from the four-year period 2001–2004. It consists of a 3.88-ha sedimentation basin, followed by 4.72 ha in four wetland cells in series. The system received 16,600 $m^3$/d of agricultural runoff. The cells are about 75% open water and 25% vegetated with bulrushes. Data were summarized in Tetra Tech, Inc. (TTI) (2006). The TN areal loading was over 40 times that at the Orlando Easterly Wetland.

The hydraulic loading to the system is high (19.3 cm/d), and 35% infiltrates. The incoming water has high TSS (179 mg/L, Table 9.9A), which is effectively removed in the sedimentation basin and wetland cells. However, particulate nitrogen is low, and is not reduced in the system. Oxidized and dissolved organic nitrogen dominate the inflow, which has a TN of 6.8 mg/L; the outflow has 3.8 mg/L TN (44% concentration reduction) (Table 9.9A). About 25% of the nitrogen load is infiltrated (Figure 9.14). In contrast to the Orlando system, the vegetative cycle at Imperial has almost no effect on the nitrogen budget. Vegetation was sparse, and gross uptake was estimated to be only 2% of the incoming nitrogen load.

Ammonification primarily reduces the load of dissolved organic nitrogen. Nitrification and denitrification dominate the processing matrix (Figure 9.14). The required supply of oxygen, in excess of the observed depletion of the water column DO, was 1.45 gO/$m^2$·d, which is reasonably within the range of expected atmospheric reaeration (see Chapter 5) (Table 9.9B). Sufficient alkalinity was present to support nitrification. However, there was estimated to be not enough carbon available from the decomposition of the sparse vegetation, or incoming $CBOD_5$, to support denitrification. A possible candidate mechanism was sulfur-driven autotrophic denitrification. The incoming water contained over 600 mg/L of sulfate. If only a small fraction, less than 1%, of this were reduced to sulfide in the wetland sediments, then that sulfide could have supported the balance of the observed denitrification over carbon-driven, heterotrophic denitrification.

## Musselwhite, Ontario, FWS Wetland

The Musselwhite gold mine uses FWS wetland treatment to deal with the ammonia that is produced in the gold extraction and cleanup processes. This 2.5-ha constructed wetland was operated in the unfrozen seasons, at a depth of about 30 cm and a hydraulic loading rate of 50 cm/d (Bishay and Kadlec, 2005). The site was a former forested peatland, with the trees cut down, and logs and brush left in the wetland. Marsh vegetation consisted of *Equisetum* spp., *Typha* spp., and *Carex* spp.

**TABLE 9.9B**
**Assumptions for the Imperial, California, FWS Wetland Carbon and Oxygen Supplies**

| Assumption | | | Notes |
|---|---|---|---|
| Biomass produced = | 500 | g dw/$m^2$·yr | — |
| Carbon content = | 0.5 | — | 50% |
| Useable carbon fraction = | 0.3 | — | 30% |
| Carbon available = | 75 | g/$m^2$·yr | — |
| Denitrification carbon requirement = | 179 | g/$m^2$·yr | 1.07 × N |
| Denitrification sulfide requirement = | 164 | g/$m^2$·yr | 1.69 × N excess |
| Sulfate incoming = | 47,000 | g/$m^2$·yr | — |
| Biomass N uptake = | 10 | g/$m^2$·yr | 2% N |
| Biomass buried = | 0.1 | g dw/$m^2$·yr | 10% |
| Nitrogen buried = | 1.0 | g/$m^2$·yr | 2% N |
| Oxygen needed = | 529 | g/$m^2$·yr | 3.43 × nitritation + 1.14 × nitrification, less DO reduction |
| Daily oxygen needed = | 1.45 | g/$m^2$·d | — |

*Note:* Biomass is the assumed source of carbon, and oxygen requirements are determined from Figure 9.15 fluxes.

**TABLE 9.10A**
**Average Inlet and Outlet Concentrations for the Musselwhite, Ontario, FWS Wetland for 1997–2002**

| Parameter | Inlet (mg/L) | Outlet (mg/L) | Mean (mg/L) | Fraction |
|---|---|---|---|---|
| HLR, cm/d | — | 52 | — | — |
| Dissolved organic N | 1.18 | 0.75 | 0.97 | — |
| Ammonia N | 11.61 | 3.18 | 7.40 | 0.344 |
| Nitrite N | 0.85 | 0.19 | 0.52 | — |
| Nitrate N | 5.79 | 10.10 | 7.95 | 0.656 |
| TSS | 5.0 | 5.0 | — | — |
| DO | 10.7 | 2.8 | — | — |
| Alkalinity | 124 | 86 | — | — |

*Source:* Data from Bishay and Kadlec (2005) In *Natural and Constructed Wetlands: Nutrients, Metals, and Management*. Vymazal (Ed.), Backhuys Publishers, Leiden, The Netherlands, 176–198.

Water is stored over winter in a pond, and is essentially devoid of TSS and BOD. However, partial nitritation and nitrification take place in the storage pond, leading to a mix of the nitrogen species entering the wetland (Table 9.10A, B).

The TN areal loading was over 300 times that at the Orlando Easterly wetland. Therefore, the vegetation utilization of nitrogen is of negligible consequence (Figure 9.15). There was also little organic nitrogen entering the wetland, and as a result dissolved inorganic nitrogen dominates the set of transfer processes. There was 75% reduction in the ammonia concentration, which is the regulatory parameter of interest. Because of nitrification, there was an increase in the nitrate concentration though the wetland of 80%, and these two effects partially counteract in TN reduction (25%).

Two anomalies were present concerning the supplies of ancillary chemicals. First, if nitritation and nitrification were purely heterotrophic, the conventional chemistry indicates a need for 20.2 gO/m$^2$·d, of which 4.1 was supplied by a depletion of incoming DO (Bishay and Kadlec, 2005). The net requirement of 16.1 gO/m$^2$·d is well outside the range of expectations for reaeration. Second, the carbon supply for purely heterotrophic "conventional" denitrification would be ten times higher than that estimated to be available from biomass decomposition.

An alternative possibility is that autotrophic nitrification/denitrification could have occurred. Van Loosdrecht and Jetten (1998) note that "autotrophic nitrifiers might be responsible for a range of 'strange' nitrogen conversions in wastewater treatment processes." The presence of considerable nitrite in the inlet water (13% of oxidized nitrogen), as well as ammonia, created conditions conducive for Equation 9.31. This relieves both the oxygen and carbon requirements, by about half (Bishay and Kadlec, 2005). The transfers in Figure 9.15 reflect this assumption.

## Dar es Salaam, Tanzania, HSSF Wetland

This HSSF wetland system is used to provide secondary treatment of effluent from a primary facultative pond at the University of Dar es Salaam, Tanzania (Senzia *et al.*, 2002b). The system consists of four HSSF wetland beds in parallel; each bed is 40.7 m$^2$, and the hydraulic loading was approximately 5 cm/d. Nitrogen in the pond effluent is dominated by ammonia, and by organic nitrogen (Figure 9.16). The influence of plant biomass cycling is apparent; a large fraction of the influent ammonia (32%) is uptaken by the plants; the majority of this is returned back to the system as organic nitrogen (plant biomass increases the influent organic-nitrogen loading by 46%). However, organic nitrogen undergoes ammonification and this nitrogen is returned to the ammonia pool. Nitrification and denitrification are significant, exporting 48.8% of the applied nitrogen load; however, the majority of the nitrogen present in the effluent is in the form of ammonia (88% of the effluent nitrogen), and the export of effluent nitrogen accounts for 46.4% of the influent load. Only 4.8% of the nitrogen is stored in sediments and plant detritus.

**TABLE 9.10B**
**Assumptions for the Musselwhite, Ontario, FWS Wetland Carbon and Oxygen Supplies**

| Assumption | | | Notes |
|---|---|---|---|
| Biomass produced = | 600 | g dw/m$^2$·yr | — |
| Carbon content = | 0.5 | — | 50% |
| Useable carbon fraction = | 0.3 | — | 30% |
| Carbon available = | 90 | g/m$^2$·yr | — |
| Heterotrophic denitrification supported = | 84 | g/m$^2$·yr | 1C/1.07 |
| Autotrophic denitrification = | 864 | g/m$^2$·yr | Difference |
| Biomass N uptake = | 12 | g/m$^2$·yr | 2% N |
| Biomass buried = | 60 | g dw/m$^2$·yr | 10% |
| Nitrogen buried = | 1.2 | g/m$^2$·yr | 2% N |
| Oxygen needed = | 2,167 | g/m$^2$·yr | 1.6 × nitritation + 3.0 × nitrification |
| Daily oxygen needed = | 5.9 | g/m$^2$·d | — |
| Biomass produced = | 600 | g dw/m$^2$·yr | — |

*Note:* Biomass is the assumed source of carbon, and oxygen requirements are determined from Figure 9.16 fluxes.

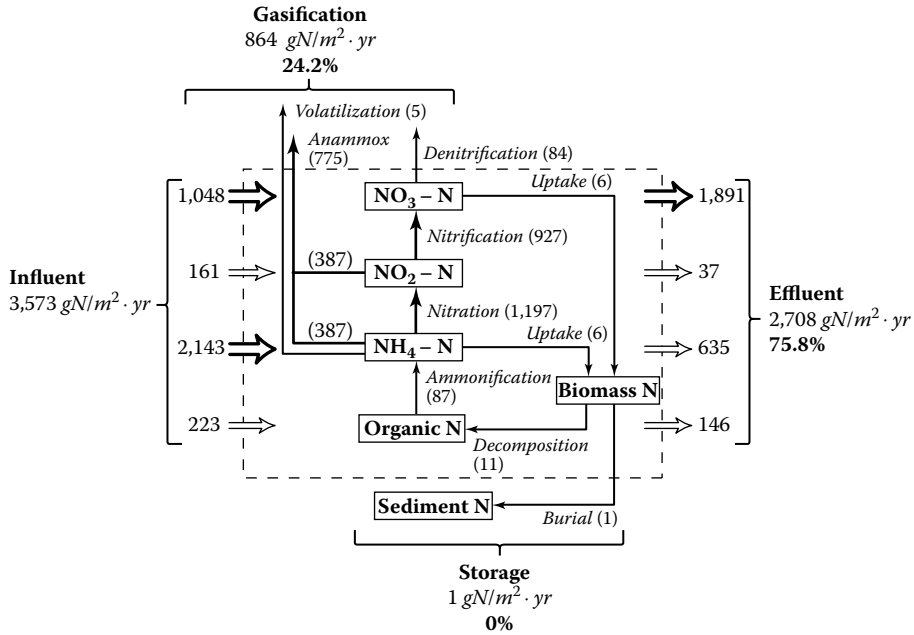



**FIGURE 9.15** Mass balance for nitrogen flows in the Musselwhite, Ontario, FWS wetland (gN/m²·yr), for an autotrophic nitrification/denitrification assumption. Base data were means of six years' measurements. The rate of denitrification, 84 gN/m²·yr, was estimated based upon carbon availability. (Adapted from Bishay and Kadlec (2005) In *Natural and Constructed Wetlands: Nutrients, Metals, and Management*. Vymazal (Ed.), Backhuys Publishers, Leiden, The Netherlands, pp. 176–198.)

Figure 9.16 is an excellent illustration of the pitfalls of using input–output analysis for specific nitrogen species. If ammonia is considered to the exclusion of other nitrogen species, one could conclude that the system is not particularly effective in ammonia-nitrogen removal (influent of 326 gN/m²·d; effluent of 217 gN/m²·d). This of course ignores the impacts of the organic nitrogen fraction and the importance of plant biomass cycling in this system. Only when all of the nitrogen species are considered in concert can an overall understanding of nitrogen removal be developed.

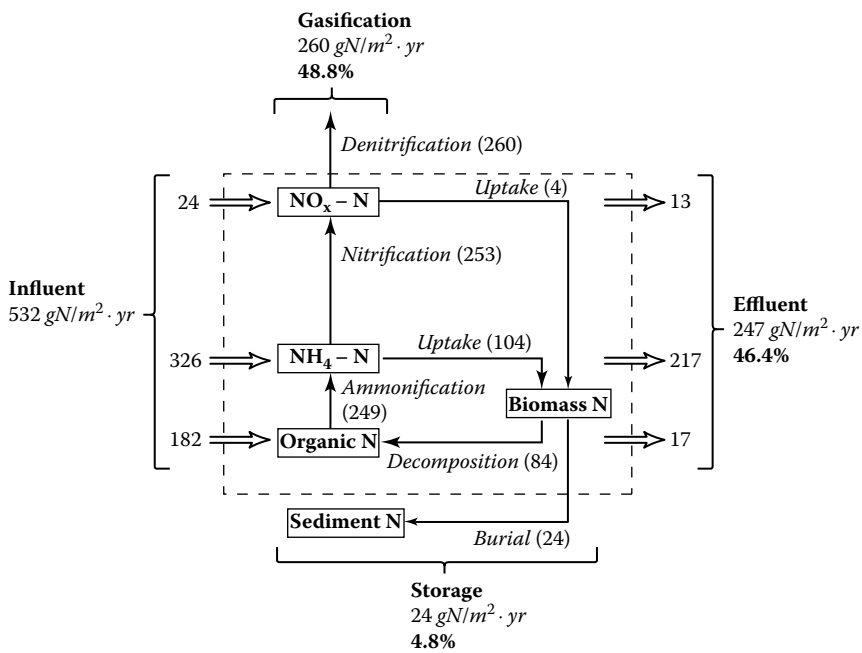



**FIGURE 9.16** Nitrogen species mass balances for a *Phragmites mauritius* HSSF wetland. (Adapted from Senzia *et al.* (2002b). *Modeling nitrogen transformation in horizontal subsurface flow constructed wetlands planted with* Phragmites mauritius. Mbwette (Ed.). Proceedings of the 8th International Conference on Wetlands Systems for Water Pollution Control, 16–19 September 2002; Comprint International Limited: University of Dar es Salaam, Tanzania, pp. 813–827.)

### Implications of the Nitrogen Mass Balance Network

A few important points emerge from this integrated view of nitrogen processing. First, the magnitude of the vegetative nitrogen cycle is by no means always trivial, because uptake can represent a good portion of the net removal for lightly loaded systems. However, net burial is only a fraction of plant uptake. Second, the influence of the biomass decay causes the true amount of ammonification to exceed the apparent rate based only on water analyses. Third, the true amount of nitrification greatly exceeds the amount based only on ammonia input–output water analyses. A sequential nitrogen kinetic model corrects for the production of ammonium from organic nitrogen, and calibrates to have higher rate constants accordingly. Finally, the rate of denitrification far exceeds the rate based only on nitrate input–output water analyses. The contribution of nitrification means that apparent denitrification is much smaller than the true value.

When microbial processes dominate, and the effects of the vegetative cycle are negligible, there are three independent mass balances that may be contrived without influences from other nitrogen species: (1) organic nitrogen, (2) TKN, and (3) TN. These are all groups of compounds, not single chemical entities. The overall reactions are:

$$\text{Organic N} \rightarrow NH_3 \tag{9.36}$$

$$\text{TKN} \rightarrow products_1 \tag{9.37}$$

$$\text{TN} \rightarrow products_2 \tag{9.38}$$

where

$products_1$ = oxidized N plus gases ($NH_3$, $N_2O$, $N_2$)
$products_2$ = gases ($NH_3$, $N_2O$, $N_2$)

Accordingly, it is reasonable to write disappearance models for these three, without including any production terms. There is, however, a background concentration of organic nitrogen ($C^*$), which influences all three rates. Nitrate, nitrite, and ammonia are all produced as well as consumed in the conversion web, and therefore reaction kinetics for these are of necessity more complex.

## 9.6 PERFORMANCE FOR ORGANIC NITROGEN

Organic nitrogen is present in domestic and municipal effluents. Wetlands typically receive these wastewaters after partial treatment, and the wetland influent then contains varying amounts of the original organic nitrogen, depending upon the type of pretreatment. Wetlands are themselves organic-rich sites, with considerable internal production of nitrogenous compounds. Incoming organic nitrogen is reduced, but not below the background concentration created by residuals and wetland return fluxes. Organic nitrogen is rarely, if ever, a regulated water quality parameter.

### Loading Considerations

Measurements of ammonification rates in natural wetlands ranged from 1 to 15 g/m²·yr (annual average 1.5) in a swamp forest in central Minnesota (Zak and Grigal, 1991) and from 4.3 to 5.9 g/m²·yr in a Minnesota bog (Urban and Eisenreich, 1988). Treatment wetlands are typically nutrient-enriched environments, and process more organic nitrogen than natural systems.

### Reduction of Organic Nitrogen in FWS Wetlands

The median net period-of-record removal rate for 60 FWS systems receiving more than 5 mg/L of organic nitrogen is 90 g/m²·yr (Table 9.11). There is, however, wide variability among systems.

As detailed in Chapter 6, it is possible to represent annual wetland performance as the effluent concentration produced ($C_o$) by a given loading rate in (LRI = HLR × $C_i$) and concentration ($C_i$). In the broad context, multiple data sets are represented by a trend that shows increasing $C_o$ with increasing LRI, with different groupings associated with each inlet

**TABLE 9.11**
**Annual Reduction of Organic Nitrogen in FWS Wetlands**

**Stipulations**

1. Data restricted to wetlands receiving inlet $C$ > 5 mg/L organic nitrogen.
2. Period of record averages are used in calculations.
3. For $k$-value calculations, the following $P$-$k$-$C^*$ parameters are selected:
   a. $C^*$ = 1.5 mg/L
   b. $P$ = 3 TIS
4. Ranges of variables:

| | HLR (cm/d) | OGN In (mg/L) | OGN Out (mg/L) |
|---|---|---|---|
| **Mean** | 6.3 | 18.2 | 8.1 |
| **Median** | 3.9 | 10.8 | 5.7 |
| **Max** | 49.9 | 69.5 | 29.6 |
| **Min** | 1.0 | 5.7 | 1.6 |

**Results (N = 60 wetlands)**

| Percentile | Load Removed (g/m²·yr) | Rate Coefficient (m/yr) |
|---|---|---|
| 0.0 | 3 | 0.6 |
| 0.1 | 28 | 5.0 |
| 0.2 | 46 | 7.9 |
| 0.3 | 55 | 10.7 |
| 0.4 | 78 | 14.6 |
| 0.5 | 90 | 17.3 |
| 0.6 | 131 | 19.7 |
| 0.7 | 180 | 27.4 |
| 0.8 | 264 | 36.2 |
| 0.9 | 395 | 61.9 |
| 1.0 | 3,461 | 262.4 |

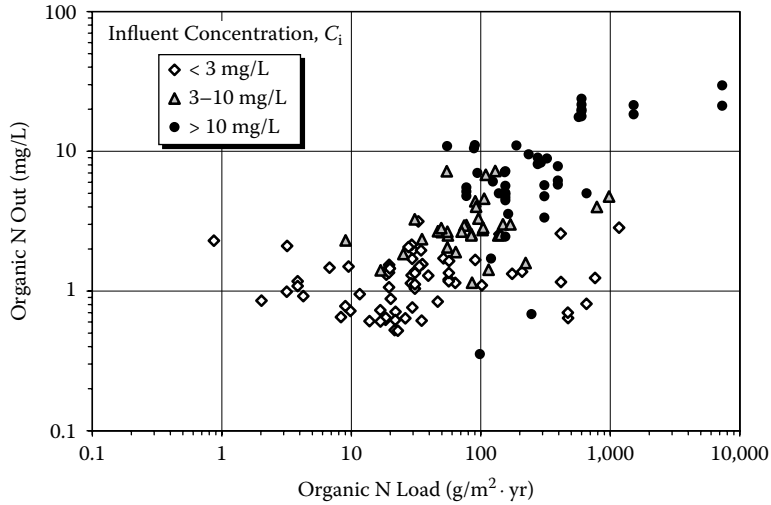



**FIGURE 9.17** Load–concentration plot for organic nitrogen in FWS wetlands. Points are separated according to the inlet concentration range. Each point represents the entire period of record (POR) for one of 147 wetlands.

concentration (Figure 9.17). The overall slope of the intersystem data is approximately 0.5 on the log–log coordinates but is close to 1.0 in the central loading region. However, if the data are sorted into different inlet concentration ranges, a different picture emerges. For inlet concentrations in the range of 0.5–2.5 mg/L, there is little change in the outlet concentrations as the organic nitrogen loading is varied. Importantly, if hydraulic loading is reduced at constant inlet concentration, there is far less effect than indicated by the 0.5 slope of the overall data trend. Loading is an insufficient design specification because hydraulic load and inlet concentration are not interchangeable factors in the load representation.

## Reduction of Organic Nitrogen in HSSF Wetlands

Many studies of HSSF wetlands have ignored the impact of organic nitrogen, even though ammonification of organic nitrogen represents a potential route of ammonia production within HSSF wetlands beds (Wallace and Knight, 2006; WERF database, 2006). Annual average effluent concentrations as a function of influent organic nitrogen loading for 123 HSSF wetlands (198 system-years of data) are summarized in Figure 9.18.

As seen in Figure 9.18, it is seen that there is a trend towards increasing effluent concentrations with increasing influent loadings of organic nitrogen, with an overall slope

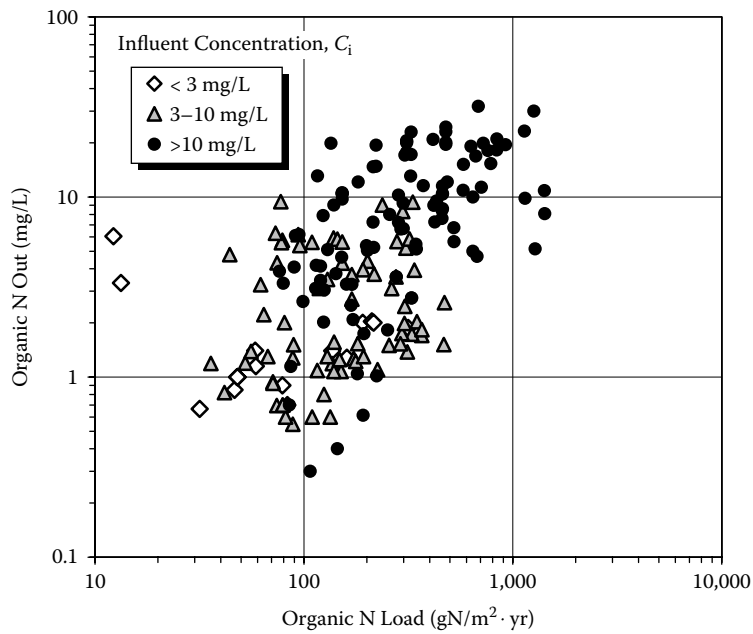



**FIGURE 9.18** Outlet organic nitrogen as a function of inlet organic nitrogen loading for HSSF wetlands. Data are annual averages for 198 wetland-years from 123 wetland cells.

## TABLE 9.12
## Annual Reduction of Organic Nitrogen in HSSF Wetlands

**Stipulations**

1. The decomposition of 2000 g/m²·yr of biomass causes production of 36 gN/m²·yr of organic nitrogen.
2. Annual averages are used in calculations.
3. For *k*-value calculations, the following *P*-*k*-*C** parameters are selected:
   a. $C^* = 1.0$ mg/L
   b. $P = 6$ TIS
4. Ranges of variables:

| | HLR (cm/d) | OGN In (mg/L) | OGN Out (mg/L) |
|---|---|---|---|
| **Mean** | 7.0 | 13.0 | 6.8 |
| **Median** | 4.7 | 11.6 | 4.3 |
| **Max** | 41.2 | 107.3 | 73.2 |
| **Min** | 0.1 | 0.0 | 0.0 |

**Results (N·*t* = 190 wetland-years)**

| Percentile | Load Removed (g/m²·yr) | Rate Coefficient (m/yr) |
|---|---|---|
| 0.05 | 27 | 1.9 |
| 0.10 | 37 | 3.8 |
| 0.20 | 57 | 6.5 |
| 0.30 | 79 | 8.8 |
| 0.40 | 95 | 12.4 |
| 0.50 | 112 | 19.6 |
| 0.60 | 137 | 25.6 |
| 0.70 | 158 | 38.0 |
| 0.80 | 194 | 72.4 |
| 0.90 | 249 | 124.2 |
| 0.95 | 305 | 168.3 |

of the intersystem data set of approximately 1.0 on log–log coordinates. However, when the influent loadings are broken down by concentration ranges, it is apparent that this relationship does not hold for systems with $C_i < 3$ mg/L, presumably because these systems are operating at an influent concentration close to the background concentration (*C**). Furthermore, there is considerable variability among systems. The median annual average removal of organic nitrogen is 112 g/m²·yr, as summarized in Table 9.12.

### Background Concentrations of Organic Nitrogen

Treatment wetlands data display decreases in organic nitrogen with contact time, which are consistent with first-order reduction kinetics, but show a nonzero background concentration. For long detention times, corresponding to large distances from the inlet, small concentrations of organic N persist. Those background concentrations typically are in the range of 0.5–2.0 mg/L, and are therefore nontrivial with respect to some regulatory requirements for TN.

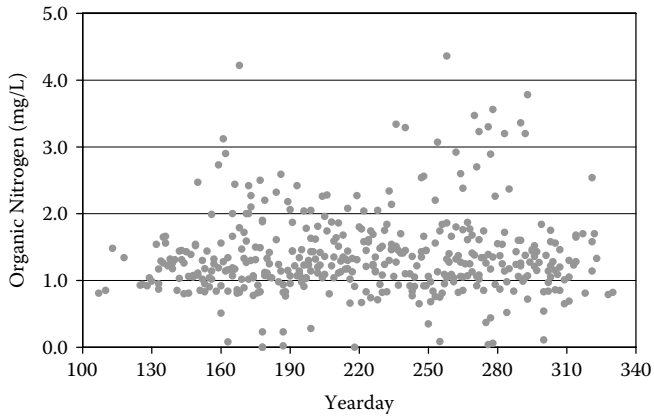



**FIGURE 9.19** Organic nitrogen in the effluent from the Estevan, Saskatchewan, constructed wetland. Data are weekly during the unfrozen season 1994–2003, with an arithmetic mean of 1.58 mg/L. (Unpublished data from town of Estevan.)

### Background Concentrations in FWS Wetlands

Because a portion of the background is due to decay processes in the wetland ecosystem, there is an effect of overall nutrient loading on the background. Lightly loaded wetlands that receive very little nitrogen or phosphorus possess lower backgrounds, such as the Orlando Easterly Wetland system in Florida (about 0.6 mg/L) or the Des Plaines, Illinois, wetlands (0.6–1.0 mg/L). Treatment wetlands that receive lagoon or secondary effluent are more heavily fertilized, and produce backgrounds of 1.5–2.0 mg/L.

There is not a large seasonality for background organic nitrogen. Wetlands operated at low hydraulic loadings have outlet concentrations approximating background. Examination of both northern and southern systems shows little seasonality, as typified by the Estevan, Saskatchewan, wetland, which operates during the unfrozen season (Figure 9.19).

### Background Concentrations in HSSF Wetlands

Analysis of $C_i$ versus $C_o$ data for HSSF wetlands suggests there is a background concentration (*C**) in the range of 1–3 mg/L (Figure 9.20); a background concentration of 1.0 mg/L has been presumptively assumed for the rate constant analysis presented in this book.

However, it should be noted that several factors influence the organic nitrogen *C** range. Plant biomass cycling will return approximately 36 g/m²·yr of organic nitrogen back to the water column (accounted for in Figure 9.20). However, if the HSSF wetland bed is insulated with a mulch layer, the presence of this mulch material can exert an additional organic nitrogen loading on the system, especially if poorly decomposed mulch materials such as wood chips or tree bark are used. Data presented in Wallace *et al*. (2001) indicates that degradation of mulch materials can lead to TKN effluent concentrations in the range of 40–60 mg/L, and this elevation can continue for two to three years. Well-decomposed mulch materials such as peat or yard waste compost

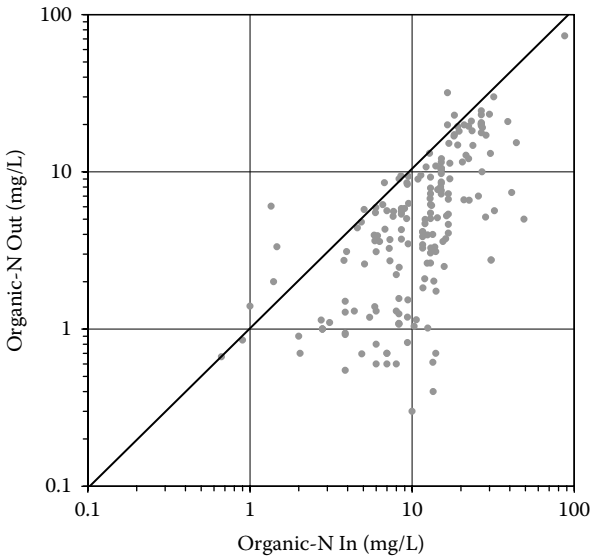



**FIGURE 9.20** Outlet organic nitrogen as a function of inlet concentration for HSSF wetlands. Data are annual averages for 193 wetland-years from 116 wetland cells.

will return much lower effluent concentrations, in the range of 10–30 mg/L TKN.

## Rates and Rate Constants

In conventional activated sludge treatment system design, ammonification is assumed to pertain to soluble organic nitrogen, and is modeled as a second-order process, first-order in soluble organic nitrogen and first-order in the biomass of heterotrophic microorganisms (U.S. EPA, 1993b). The ammonification rate increases with a doubling of the rate constant for a temperature increase of 10°C ($\theta = 1.07$) (U.S. EPA, 1993b).

The rate of ammonification in flooded soils also depends on temperature and pH (Reddy and Patrick, 1984). The ammonification rate increases with a doubling of the rate constant for a temperature increase of 10°C ($\theta = 1.07$). The rate of organic N mineralization was shown to increase with increasing temperature, from 5 to 35°C (Stanford *et al*., 1973). The θ-values are close to 1.07 in a temperature range of 15–35°C, but slightly higher ($\theta = 1.08$) at lower temperatures, 5–15°C. Mineralization essentially ceases when soil is frozen. The optimum pH range for ammonification is between 6.5 and 8.5 (Reddy and Patrick, 1984).

The organic nitrogen designation represents a large group of contributing forms and compounds. A large portion of organic nitrogen in wastewaters is likely to be particulate, although the particle size may be very small, resulting from bacterial debris and colloidal materials. A large part of the particulate organic fraction may be biodegradable (U.S. EPA, 1993b). The remaining portion comprises a potentially large number of soluble materials, ranging from the polypeptide components of humic substances to simple amino acids and urea (Fuchsman, 1980). Very few wetland studies have attempted to distinguish between dissolved and particulate forms. However, the Imperial, California, FWS project found that particulate organic nitrogen was not reduced through the train of wetland cells, whereas dissolved organic nitrogen was somewhat reduced, thus leaving a background of both particulate and dissolved forms.

### Organic Nitrogen Rate Constants in FWS Wetlands

The loss of organic nitrogen in treatment wetland environments is here assumed to follow a first-order model, although there are but few studies that document the requisite decreasing profile through the wetland. For instance, the first-order assumption was made by Gerke *et al*. (2001) for particulate organic nitrogen removal in an FWS wetland. Such profiles were determined in the Listowel, Ontario, project, and displayed virtually no seasonality or temperature effect (Figure 9.21). The Listowel profiles show a decline to

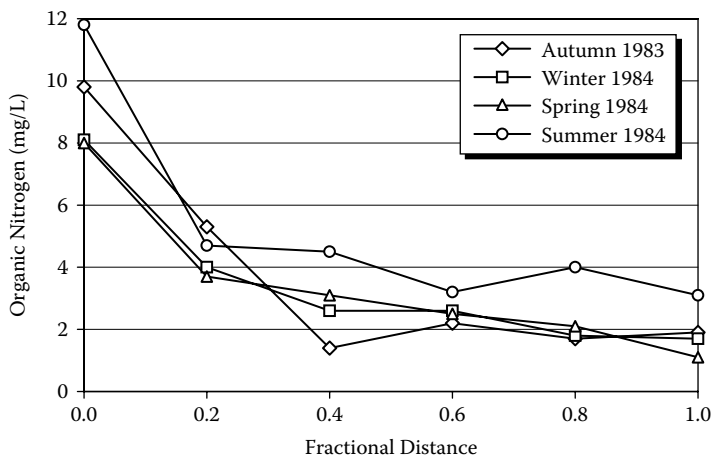



**FIGURE 9.21** Organic nitrogen profiles through the Listowel, Ontario, FWS system 4 during all seasons. Samples were taken weekly, except biweekly in winter. The flow was collected in a culvert at each measurement point. (Data from Herskowitz (1986) *Listowel Artificial Marsh Project Report*. Ontario Ministry of the Environment, Water Resources Branch, Toronto, Ontario.)

a background plateau, which supports the concept of a background concentration.

Accordingly, an area-based first-order removal rate is utilized here:

$$J_{ON} = k_{ON}(C_{ON} - C^*_{ON}) \tag{9.39}$$

where

$C_{ON}$ = wetland organic nitrogen concentration, mg/L
$C^*_{ON}$ = background wetland organic nitrogen concentration, mg/L
$J_{ON}$ = removal rate of organic nitrogen, g/m²·yr
$k_{ON}$ = removal rate constant for organic nitrogen, m/yr

The wetland environment may have actual hydraulics ranging from a few tanks in series (TIS) up to a large number, approximating plug flow, depending on design. However, organic nitrogen is expected to show weathering effects due to its complex speciation, thus reducing the effective number of TIS (see Chapter 6). Accordingly, the *P-k-C** model is chosen, with $P < N$. To compare results across systems that in general do not have known *N*-values, the value $P = 3$ is chosen here. Further, there is a fairly narrow band of *C** values, and therefore, $C^* = 1.5$ mg/L is chosen here to allow comparisons. The remaining model parameter is the *k*-value, selected to fit the model:

$$\frac{C_{ON,out} - C^*}{C_{ON,in} - C^*} = \left(1 + \frac{k_{ON}}{3q}\right)^{-3} \tag{9.40}$$

where

$q$ = wetland hydraulic loading, m/yr
$C^*$ = 1.5 mg/L (assumed)

Because of the selection of $C^* = 1.5$, parameter estimation is not reliable for low inlet concentrations, and those wetlands with $C_{ON,in} < 5$ mg/L have been excluded from calibration. Out of 147 wetlands with data for organic nitrogen (Figure 9.17), 60 systems met this criterion.

There appears to be little or no temperature dependence of organic nitrogen *k*-values. This concept is based upon intrasystem calibrations for individual wetlands. For example, Gerke *et al*. (2001) present data that indicate $\theta = 1.008$ for the Kingman, Arizona system. The Listowel systems calibrate to $\theta = 0.982$ for Equation 9.34, compared to 1.017 for the alternate assumption of plug flow (Kadlec and Reddy, 2001). This is in contrast to the strong temperature dependence observed in soils and mechanical activated sludge treatment systems.

Results of calibration of *k*-values for entire periods of record for the qualifying FWS wetland are summarized in Table 9.11. The median *k*-value for organic nitrogen is 17.3 m/yr, but the range is wide. The 10th–90th percentile range is 5.0–61.9 m/yr. Accordingly, there is a large design window that encompasses varying degrees of risk. Figure 9.17 may be used to place a proposed design hydraulic loading and inlet organic N concentration in the perspective of an existing database.

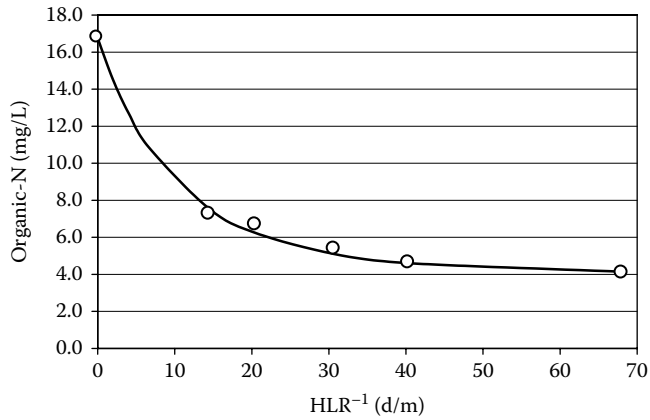



**FIGURE 9.22** Decline of organic nitrogen with detention time (inverse HLR) for side-by-side HSSF wetlands receiving dairy effluent. The line is a *P-k-C** model with $P = 6$, $C^* = 4$ mg/L, and $k = 36$ m/yr ($R^2 = 0.96$). (Data from Tanner *et al*. (1998b) *Journal of Environmental Quality*, 27(2): 448–458.)

## Organic Nitrogen Rate Constants in HSSF Wetlands

The *P-k-C** model can also be used to fit the reduction of organic nitrogen in HSSF wetlands (Figure 9.22), as the reduction appears to be first-order and decline to a nonzero background concentration (*C**).

The *P-k-C** model can be used to determine *k*-rates for organic nitrogen. Since organic nitrogen is a collection of individual nitrogenous compounds (including particulate matter) that undergo weathering in the wetland, the parameter *P* will always be less than the hydraulic parameter number of tanks in series (*N*TIS). Relatively few HSSF wetlands have been tracer tested; so the hydraulic parameter *N*TIS is not known with certainty. For a data set of 37 tracer-tested HSSF wetlands, the median value was *N*TIS = 11 (see Chapter 6). To account for weathering effects, *P*TIS = 6 has been assumed in the determination of annual rate constants. A background concentration $C^* = 1.0$ mg/L has also been assumed (see Figure 9.20).

Results of calibration for average annual *k*-rates are summarized in Table 9.12. The median *k*-value is 19.6 m/yr; but the range of *k*-values is wider than that observed in FWS wetlands. The 10th–90th percentile range is 3.8–124.2 m/yr. As a result, there is a wide range of *k*-rates that can be selected for design, with varying degrees of risk. Figure 9.19 can be used evaluate a particular design selection of *k* in the context of the existing performance database for HSSF wetlands.

There appears to be little temperature dependence or organic nitrogen *k*-values. Data from 12 HSSF wetlands yield a median value of $\theta = 1.009$, with a 10th–90th percentile range of 0.982–1.047, as indicated in Table 9.13.

**TABLE 9.13**
**Temperature Coefficients for Ammonification Rate Constants in HSSF Wetlands**

| Site | Reference | Cell | T range (°C) | Mean HLR (cm/d) | Mean $C_i$ (mg/L) | Mean $C_o$ (mg/L) | Theta |
|---|---|---|---|---|---|---|---|
| Richmond, NSW | Bavor *et al.* (1988) | Gravel | 11–24 | 3.8 | 8.29 | 1.08 | 1.009 |
| Richmond, NSW | Bavor *et al.* (1988) | *Typha* | 11–24 | 4.6 | 8.29 | 1.56 | 1.016 |
| Richmond, NSW | Bavor *et al.* (1988) | *Schoenoplectus* | 11–24 | 5.1 | 8.29 | 0.01 | 1.003 |
| Richmond, NSW | Bavor *et al.* (1988) | Mixed A | 11–24 | 4.6 | 8.29 | 1.08 | 0.982 |
| Richmond, NSW | Bavor *et al.* (1988) | Mixed B | 11–24 | 3.8 | 8.29 | 1.09 | 0.956 |
| Grand Lake, Minnesota | Unpublished data | 1 | 1–17 | 1.0 | 7.2 | 3.3 | 0.992 |
| Lincoln, Nebraska | Vanier and Dahab (1997) | *Typha, Schoenoplectus* | 4–21 | 9.5 | 11.5 | 5 | 0.982 |
| Hamilton, New Zealand | Tanner *et al.* (1998b) | L2 | 10–25 | 2.5 | 26.71 | 5 | 1.057 |
| Hamilton, New Zealand | Tanner *et al.* (1998b) | L3 | 10–25 | 3.3 | 26.71 | 5 | 1.017 |
| Hamilton, New Zealand | Tanner *et al.* (1998b) | L4 | 10–25 | 4.9 | 26.71 | 7 | 1.030 |
| Hamilton, New Zealand | Tanner *et al.* (1998b) | L5 | 10–25 | 6.9 | 26.71 | 9 | 1.047 |

| Percentile | Theta |
|---|---|
| 0.05 | 0.969 |
| 0.10 | 0.982 |
| 0.20 | 0.982 |
| 0.30 | 0.992 |
| 0.40 | 1.003 |
| 0.50 | 1.009 |
| 0.60 | 1.016 |
| 0.70 | 1.017 |
| 0.80 | 1.030 |
| 0.90 | 1.047 |
| 0.95 | 1.052 |

## 9.7 PERFORMANCE FOR TKN

The combination of ammonia and organic nitrogen, TKN, is subject to consideration as a group of compounds that are reduced in wetlands. This parameter is often regarded as representative of the total liability for ammonia nitrogen, and the presumed oxygen requirement for nitrification. Because TKN may contain a considerable proportion of ammonia, vegetation is involved in the consumption of TKN. The organic nitrogen component of TKN is added back to the water from the ecosystem decomposition processes; hence, there are important interactions with the plants (including algae) in the wetland. TKN is rarely, if ever, a regulated water quality parameter.

### Loading Considerations

Since TKN measures both organic and ammonia nitrogen, interconversions between these two species is not a concern, provided that plant uptake is accounted for (for the ammonia component). Performance data can be represented by loading analysis and the *P-k-C** model.

### Reduction of TKN in FWS Wetlands

The median net period-of-record removal rate for 101 FWS systems receiving more than 5 mg/L of TKN is 207 g/m$^2$·yr (Table 9.14). There is, however, wide variability among systems.

It is again useful to represent annual wetland performance as the effluent concentration produced ($C_o$) by a given LRI (= HLR × $C_i$) and concentration ($C_i$). In the broad context, multiple data sets are represented by a trend that shows increasing $C_o$ with increasing LRI, with different groupings associated with each inlet concentration (Figure 9.23). The overall slope of the intersystem data on the log–log coordinates varies from near zero for low inlet concentrations to about 1.0 for high inlet concentrations. As for organic nitrogen, inlet loading is an insufficient design specification because hydraulic load and inlet concentration are not interchangeable factors in the load representation.

### Reduction of TKN in HSSF Wetlands

The median annual-average removal rate for 123 HSSF wetlands (197 system-years of data) is 228 g/m$^2$·yr, as indicated in Table 9.15.

It is also useful to evaluate wetland performance ($C_o$) as a function of the inlet loading (Figure 9.23). Figure 9.24 represents data from 112 HSSF wetlands (198 system-years). In general, there is an overall upward trend of the outlet TKN concentration ($C_o$) in response to the inlet TKN loading, with a log–log slope of slightly less than 1.0. However, this apparent slope is in large measure due to the shift in inlet concentrations. When a particular inlet concentration group (like those shown on Figure 9.24) is considered, the change in outlet TKN concentration is much less, as the intersystem slope for each

**TABLE 9.14**
**Annual Reduction TKN in FWS Wetlands**

**Stipulations**

1. Data restricted to wetlands receiving inlet $C > 5$ mg/L TKN.
2. Period of record averages are used in calculations.
3. For $k$-value calculations, the following $P$-$k$-$C^*$ parameters are selected:
   a. $C^* = 1.5$ mg/L
   b. $P = 3$ TIS
4. Ranges of variables:

| | HLR (cm/d) | TKN In (mg/L) | TKN Out (mg/L) |
|---|---|---|---|
| **Mean** | 7.2 | 101.8 | 59.9 |
| **Median** | 3.9 | 32.4 | 20.0 |
| **Max** | 110.0 | 416.3 | 283.8 |
| **Min** | 0.6 | 5.1 | 1.7 |

**Results (N = 101 wetlands)**

| Percentile | Load Removed ($g/m^2 \cdot yr$) | Rate Coefficient (m/yr) |
|---|---|---|
| 0.0 | 6 | 0.2 |
| 0.1 | 42 | 4.1 |
| 0.2 | 65 | 5.2 |
| 0.3 | 92 | 6.1 |
| 0.4 | 130 | 8.5 |
| 0.5 | 207 | 9.8 |
| 0.6 | 300 | 11.3 |
| 0.7 | 508 | 13.6 |
| 0.8 | 1,115 | 20.0 |
| 0.9 | 2,203 | 35.0 |
| 1.0 | 4,683 | 153.6 |

concentration grouping is approximately 0.3. This has important design implications, because as the hydraulic loading to the wetland is decreased, the reduction in effluent concentration follows the slope of the inlet concentration group, not the overall data set. Use of the overall data set will overpredict reduction in effluent TKN concentrations as the hydraulic load is decreased.

### Reduction of TKN in VF Wetlands

Many vertical flow wetlands are designed with the express purpose of oxidizing organic and ammonia nitrogen. Effluent concentrations for TKN for vertical flow systems are summarized in Figure 9.25, which summarizes the period of record for 20 VF wetlands, annual averages for another 6 VF wetlands (17 system-years of data), plus data from intermittent sand filters that operate under similar loading and unsaturated flow conditions as VF wetlands (17 system-years of data). As Figure 9.25 illustrates, TKN loading is not an effective predictor of effluent TKN concentrations.

## Background Concentrations of TKN

Treatment wetlands data display decreases in TKN with contact time, which are consistent with first-order reduction kinetics; but show a nonzero background concentration for long detention. This is consistent with the observed small background concentrations of organic N. As shall be discussed in this chapter, there is a zero background for ammonia, so background TKN is the same as background organic nitrogen, typically in the range of 0.5–2.0 mg/L for both FWS and HSSF wetland systems. For rate analysis, a background concentration ($C^*$) value of 1.5 mg/L was assumed for FWS wetlands (Table 9.14), and a value of 1.0 mg/L was assumed for HSSF wetlands (Table 9.15).

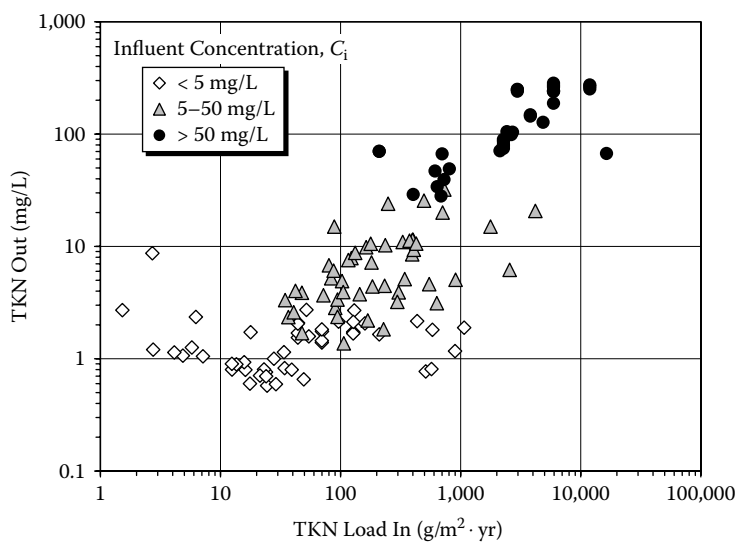



**FIGURE 9.23** Load–concentration plot for total Kjeldahl nitrogen in FWS wetlands. Points are separated according to the inlet concentration range. Each point represents the period of record (POR) for one of 135 wetlands.

## TABLE 9.15
## Annual Reduction of Total Kjeldahl Nitrogen in HSSF Wetlands

**Stipulations**

1. The decomposition of 2,000 g/m²·yr of biomass causes production of 36 gN/m²·yr of organic nitrogen.
2. Annual averages are used in calculations.
3. For *k*-value calculations, the following *P*-*k*-*C** parameters are selected:
   a. $C^* = 1.0$ mg/L
   b. $P = 6$ TIS
4. Ranges of variables:

| | HLR (cm/d) | TKN In (mg/L) | TKN Out (mg/L) |
|---|---|---|---|
| **Mean** | 7.6 | 49.3 | 32.5 |
| **Median** | 4.9 | 34.8 | 23.2 |
| **Max** | 41.2 | 226.0 | 189.5 |
| **Min** | 1.1 | 2.1 | 0.4 |

**Results (N = 123; N·*t* = 197 wetland-years)**

| Percentile | Load Removed (g/m²·yr) | Rate Coefficient (m/yr) |
|---|---|---|
| 0.05 | −180 | −18.5 |
| 0.10 | 41 | 1.8 |
| 0.20 | 109 | 3.4 |
| 0.30 | 151 | 4.8 |
| 0.40 | 185 | 7.1 |
| 0.50 | 228 | 9.1 |
| 0.60 | 287 | 11.6 |
| 0.70 | 361 | 14.6 |
| 0.80 | 453 | 19.0 |
| 0.90 | 585 | 37.5 |
| 0.95 | 1,761 | 144.2 |

### Rates and Rate Constants

In conventional activated sludge treatment system design, removal of TKN is not directly modeled, but results from ammonification of the organic component and nitrification of the ammonia component. The loss of organic nitrogen in treatment wetland environments is here assumed to follow a first-order model, based upon studies that document the requisite decreasing profile through the wetland.

Profiles along the length of the Kingman, Arizona, FWS system show such decreases, but removal is different in warm and cold seasons (Figure 9.26). Accordingly, an area-based first-order removal rate is utilized here:

$$J_{TKN} = k_{TKN}(C_{TKN} - C^*_{TKN}) \tag{9.41}$$

where

$C_{TKN}$ = wetland TKN concentration, mg/L
$C^*_{TKN}$ = background wetland TKN concentration, mg/L
$J_{TKN}$ = removal rate of TKN, g/m²·yr
$k_{TKN}$ = removal rate constant for TKN, m/yr

The wetland environment may have actual hydraulics ranging from a few TIS up to a large number, approximating plug flow, depending on wetland configuration. Organic nitrogen is expected to show weathering effects as discussed above. Ammonia is less liable to experience weathering, because it exists primarily in dissolved form, typically with only small contributions of particulate (sorbed) forms. Speculatively, the effective number of TIS (see Chapter 6) should be less than the tracer TIS, but by a slightly lesser margin than for

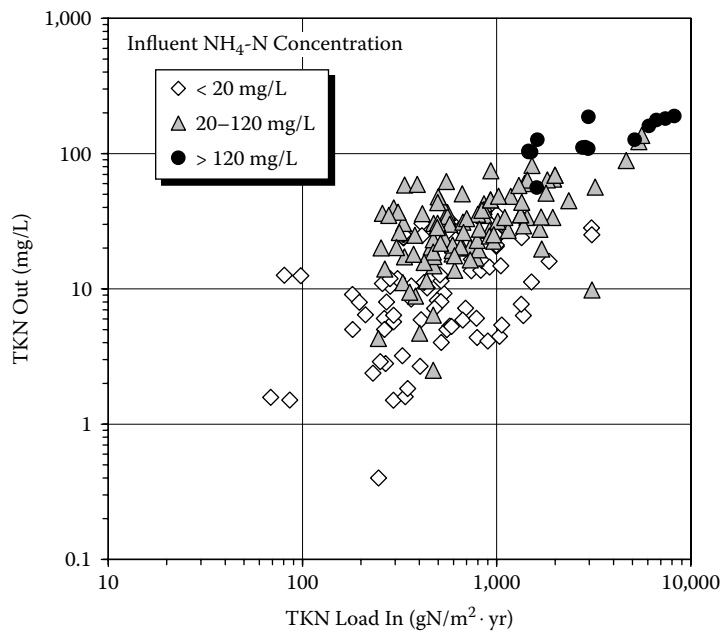



**FIGURE 9.24** Load response data for total Kjeldahl nitrogen in HSSF wetlands. Annual average information from 112 wetlands and 198 wetland-years is shown.

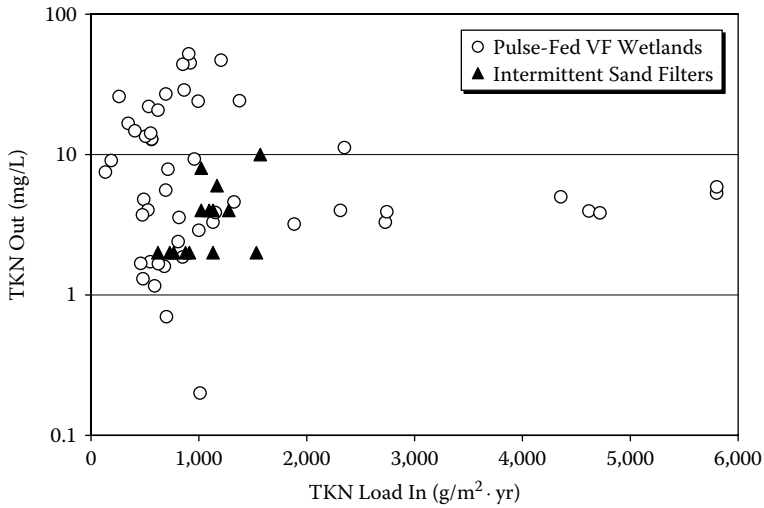



**FIGURE 9.25** Concentration–loading chart for TKN in pulse-fed vertical flow wetlands and intermittent sand filters. Data includes period-of-record performance for 20 vertical flow wetlands, annual average reductions for another 6 vertical flow wetlands (17 system-years of data), and annual average reductions for three intermittent sand filters (17 system-years of data) that were operated under similar loading regimes. TKN loading is not an effective predictor of effluent TKN concentrations.

organic nitrogen. Accordingly, the *P-k-C** model is chosen, with $P < N$.

## TKN Rate Constants for FWS Wetlands

To compare results across systems that in general do not have known *N*-values, the value $P = 3$ is chosen here. The value $C^* = 1.5$ mg/L is retained based upon organic nitrogen considerations. The remaining model parameter is the *k*-value, selected to fit the model:

$$\frac{C_{TKN,out} - 1.5}{C_{TKN,in} - 1.5} = \left(1 + \frac{k_{TKN}}{3q}\right)^{-3} \tag{9.42}$$

Because of the selection of $C^* = 1.5$, parameter estimation is not reliable for low inlet concentrations, and wetlands with $C_{TKN,in} < 5$ mg/L have been excluded from calibration. Out of 157 wetlands with data for TKN (Figure 9.23), 101 met this criterion. The median annual rate constant was $k_{TKN} = 9.8$ m/yr (Table 9.14). The 10th–90th percentile range is 4.1–35.0 m/yr. There is a significant temperature dependence of TKN *k*-values. Even on an average annual basis, temperature or season may be an important determinant of the rate constant, and is thus responsible for the some of the intersystem variability in annual *k*-values. Accordingly, it is necessary to examine intra-annual effects.

### *Microbially Dominated Wetlands*

When the TKN loading to the wetland exceeds the growth requirements of the plants and algae by a considerable margin, the removal of TKN is very likely to be microbially mediated. The loading limit for bacterial conversion to predominate is approximately 120 gN/m²·yr (Kadlec, 2005d).

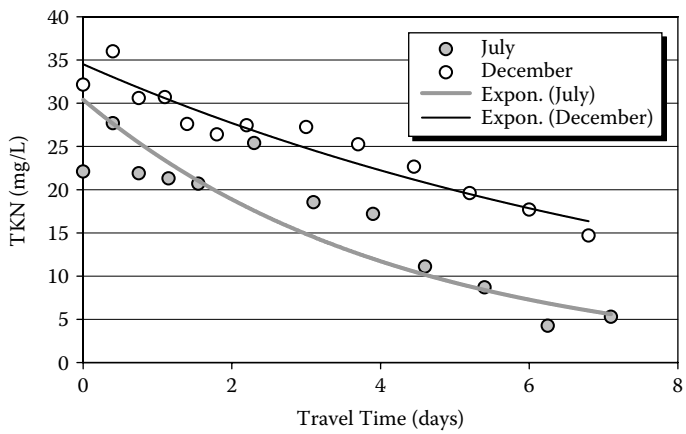



**FIGURE 9.26** Longitudinal profiles of TKN at the Kingman, Arizona, FWS wetland. (Data from Gerke *et al.* (2001) *Water Research*, 35(16): 3857–3866.)

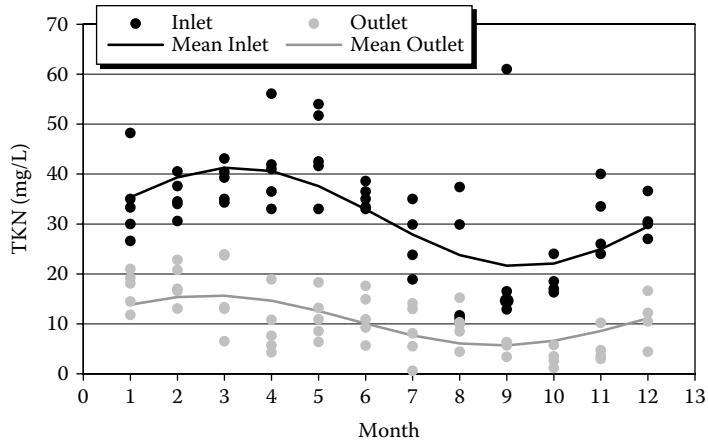



**FIGURE 9.27** Folded inlet and outlet time series for TKN for the Kingman, Arizona, FWS wetland. (Unpublished data from city of Kingman.)

| Cyclic parameters | Inlet | Outlet |
|---|---|---|
| $C_{mean}$ (mg/L) | 31.4 | 10.7 |
| $A$ | 0.32 | 0.47 |
| $t_{max}$ (days) | 83 | 66 |
| $t_{min}$ (days) | 265 | 249 |

There is typically a monotonic decline in TKN along the flow path of a wetland (see Figure 9.26). Sampling along the flow direction results in variability from at least two sources: (1) spatial selection of the sampling points, and (2) temporal variability in input flows and concentrations that may propagate in the flow direction. Nevertheless, there is a clear downward trend, as TKN is removed from the water during travel through the wetland. Rates of decline are faster in summer than in winter, implying that a temperature effect is present in these microbially dominated systems. In many wetland systems, there are annual trends in input concentrations that often follow a sinusoidal tend, reflecting changes in the pretreatment and inlet water quality for that pretreatment wetland (Figure 9.27). Under these circumstances, it is not appropriate to use percentage reductions as a measure of performance, because of the confounding effects of seasonal flows, concentrations, and microbial activity. Accordingly, the first-order model is here utilized, together with a temperature coefficient (θ), which are capable of accounting for these effects (see Chapter 6).

Results of calibration of $k$-values for entire periods of record for representative wetlands are summarized in Table 9.16. Monthly averages were used to avoid synoptic error (transit time offset). Calibrations were performed for best estimates of the internal hydraulics for each wetland. Therefore, $P$-values range from 2 (New Hanover, measured $P = N = 2$) to near plug flow conditions, based upon system geometry. In most cases, the $C^* = 1.5$ was used, excepting three cases in which slightly different $C^*$ were indicated by data. The median $k_{20}$-value for TKN is 21.0 m/yr, but the range is wide.

Temperature coefficients had a median value of 1.036, indicating a relatively strong thermal effect on the suite of microbial processes that contribute to TKN reduction.

The example systems in Table 9.16 do not display any limitations due to the supplies of oxygen. The theoretical oxygen demands for full nitrification of the removed TKN are in the range of 0–7.1 g/m²·d, which is within the feasible range of reaeration combined with inlet dissolved oxygen. There was generally some BOD entering these example systems, with a median of 1.5 times the entering TKN. This potential carbonaceous oxygen demand does not contribute to an extreme need for DO in the example systems, although it may contribute to less than optimal nitrification. The role of open water in providing the oxygen for nitrification is not clear in this intersystem comparison of rate constants for TKN, because of confusion with other factors.

### *Agronomic Wetlands (Lightly Loaded Systems)*

When the TKN loading to the wetland is less than the growth requirements of the plants and algae by a considerable margin, the removal of TKN is very likely to be mediated by the growth and decay of biomass. As a rough guideline, this situation occurs for TKN loading less than approximately 120 gN/m²·yr (Kadlec, 2005d). This occurs for almost half (41%) of the 135 wetlands displayed in Figure 9.23. It is important to note that low inlet TKN load very often means very low inlet TKN concentration, close to background; consequently, there is no ability to obtain meaningful calibrations of TKN rate constants.

Uptake presumably occurs for the ammonia component of TKN, and release may be considered to add to the organic component. Because plant uptake rates do not correspond to the annual cycle of water temperatures, TKN removal in agronomic wetlands cannot be characterized by modified Arrhenius θ-factors. For example, the Estevan, Saskatchewan, system had modest hydraulic loadings coupled with low

**TABLE 9.16**
**Dependence of TKN Rate Constants on Temperature for FWS Systems**

| Site | Location | TKN In (mg/L) | TKN Out (mg/L) | TKN Load ($g/m^2 \cdot yr$) | $P$ (TIS) | $C^*$ (mg/L) | $k_{20}$ (m/yr) | θ | Estimated Open Water (%) | DO (mg/L) | Annual T (°C) | TKN Theoretical $O_2$ Demand ($g/m^2 \cdot d$) | BOD/TKN In |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Texel | The Netherlands | 3.2 | 1.8 | 36–1,279 | ∞ | 1.8 | 118 | 1.075 | 50 | 4.5–16 | 10.0 | 0.2–7.1 | — |
| Imperial | California | 3.3 | 2.1 | 368 | ∞ | 1.25 | 81 | 1.036 | 77 | 7.5–10.1 | 20.7 | 1.7 | 1.9 |
| Brawley | California | 4.7 | 1.8 | 206 | ∞ | 1.25 | 70 | 1.029 | 75 | 8.2–10.9 | 20.7 | 1.6 | 1.4 |
| Eskilstuna | Sweden | 4.9 | 3.6 | 205 | 6 | 1.5 | 21 | 1.011 | 80 | — | 6.5 | 0.7 | 1.3 |
| Brighton | Ontario | 9.9 | 11.8 | 172 | 3 | 1.5 | –4.4 | 1.045 | 20 | — | 9.4 | –0.4 | 0.5 |
| Listowel 4 | Ontario | 18.5 | 8.8 | 132 | 10 | 1.5 | 4 | 0.969 | 5 | 0.5–5 | 8.0 | 0.9 | 3.0 |
| Listowel 5 | Ontario | 18.5 | 10.6 | 178 | 3 | 1.5 | 8 | 1.011 | 5 | 0.5–3.3 | 7.9 | 1.0 | 3.0 |
| Linköping | Sweden | 18.6 | 10.8 | 288 | 3 | 1.5 | 17 | 1.037 | 10 | — | 6.7 | 1.5 | — |
| Kingman | Arizona | 33.0 | 11.0 | 501 | 20 | 1.5 | 25 | 1.022 | 4 | 2.8–7.3 | 11.7 | 4.2 | 1.5 |
| Warangal | India | 36.5 | 4.0 | 570 | 3 | 1.5 | 66 | 1.081 | — | 4 | 23.1 | 6.4 | 4.4 |
| New Hanover | North Carolina | 132.0 | 66.0 | 716 | 2 | 1.5 | 4.5 | 1.078 | 10 | 3.3–4.2 | 19.4 | 4.5 | 0.4 |
| **Median** | | | | | | **1.5** | **21.0** | **1.036** | | | **10.0** | **1.56** | **1.47** |

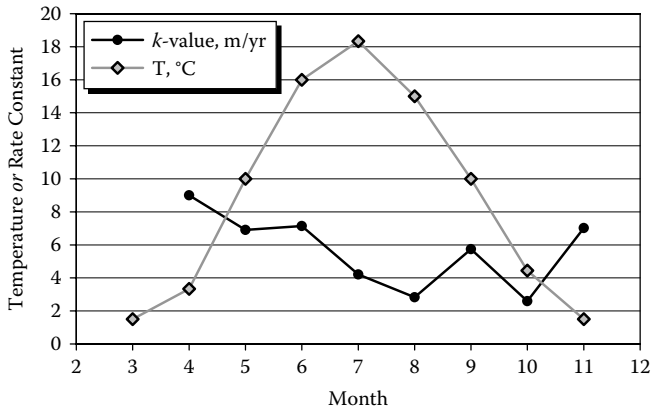



**FIGURE 9.28** The absence of a relationship between temperature and total Kjeldahl nitrogen rate constants for the Estevan, Saskatchewan, FWS system. Nine years' data are represented, for which the TKN loading was 47 g/m²·yr, well below the agronomic rate of 120 g/m²·yr.

incoming TKN from lagoon pretreatment. This system was operated seasonally (May–October) for nine years, during which time the TKN loading averaged 47 g/m²·yr. Although estimated temperatures ranged from less than 2°C to about 20°C, there was no corresponding change in monthly *k*-values for TKN (Figure 9.28).

For agronomic wetlands, there may still be seasonality, out of synchronization with temperature. The appropriate method of dealing with this situation is via monthly *k*-values, according to (see Chapter 6):

$$k = k_j \tag{9.43}$$

where

$j$ = month number = 1, 2, 3, ... 12

For the nine-year period of record for Estevan, the mean monthly results are:

| | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| $k_{TKN,j}$ | = 9.0 | 6.9 | 7.1 | 4.2 | 2.8 | 5.7 | 2.6 | 7.0 m/yr |
| $j$ | = 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 |

(9.44)

In any case, it is prudent to examine the actual seasonal progression of *k*-values, to ascertain whether a θ-value approach is warranted or not. Additionally, if there is not a temperature correlation, the $R^2$ for a θ-factor regression will be very low.

### TKN Rate Constants for HSSF Wetlands

Ranges in annual average *k*-rates for HSSF wetlands are summarized in Table 9.15, with the assumptions of *P*TIS = 6; *C** = 1.0 mg/L. The median *k*-rate is 9.1 m/yr. However, there is wide variability in the data, and the 10th–90th percentile range is 1.8–37.5 m/yr.

There appears to be little, if any, temperature dependence on TKN removal in HSSF wetlands. Data from 9 HSSF wetlands are presented in Table 9.17. The median θ-factor was 1.001; and the 10th–90th percentile range is 0.951–1.011.

**TABLE 9.17**
**Temperature Coefficients for TKN Removal Rate Constants in HSSF Wetlands**

| Site | Reference | Cell | T range (°C) | Mean HLR (cm/d) | Mean $C_i$ (mg/L) | Mean $C_o$ (mg/L) | Theta |
|---|---|---|---|---|---|---|---|
| Richmond, NSW | Bavor *et al.* (1988) | Gravel | 11–24 | 3.8 | 43.5 | 19.9 | 1.001 |
| Richmond, NSW | Bavor *et al.* (1988) | *Typha* | 11–24 | 4.6 | 43.5 | 20.4 | 1.001 |
| Richmond, NSW | Bavor *et al.* (1988) | *Schoenoplectus* | 11–24 | 5.1 | 43.5 | 19.4 | 1.006 |
| Richmond, NSW | Bavor *et al.* (1988) | Mixed A | 11–24 | 4.6 | 43.5 | 16.4 | 1.005 |
| Richmond, NSW | Bavor *et al.* (1988) | Mixed B | 11–24 | 3.8 | 43.5 | 13.7 | 0.998 |
| Benton, Kentucky | TVA unpublished data | 3 | 5–25 | 8.4 | 14.3 | 10.2 | 0.954 |
| Grand Lake, Minnesota | Unpublished data | 1 | 1–17 | 1.0 | 58.4 | 41.2 | 1.008 |
| Lincoln, Nebraska | Vanier and Dahab (1997) | *Typha, Schoenoplectus* | 4–21 | 9.5 | 31.8 | 21.0 | 1.024 |
| Waipoua, New Zealand | Unpublished data | 1 | 11–21 | 0.4 | 74.9 | 61.7 | 0.940 |
| Richmond, NSW | Bavor *et al.* (1988) | Gravel | 11–24 | 3.8 | 43.5 | 19.9 | 1.001 |
| Richmond, NSW | Bavor *et al.* (1988) | *Typha* | 11–24 | 4.6 | 43.5 | 20.4 | 1.001 |

| Percentile | Theta |
|---|---|
| 0.10 | 0.951 |
| 0.20 | 0.980 |
| 0.30 | 0.999 |
| 0.40 | 1.001 |
| 0.50 | 1.001 |
| 0.60 | 1.004 |
| 0.70 | 1.006 |
| 0.80 | 1.007 |
| 0.90 | 1.011 |

## 9.8 PERFORMANCE FOR TOTAL NITROGEN

TN, which is defined as the combination of organic, ammonia, and oxidized nitrogen, is subject to consideration as a group of compounds that are reduced in wetlands. This grouping is known to possess sequential conversions, including primarily ammonification followed by nitrification, followed by denitrification, all proceeding at varying rates. TN in the water is augmented by releases from decaying vegetation and microbial biomass. As a consequence, the rate of decline of TN along the flow path is expected to be determined in part by the speciation of the incoming nitrogen. TN removal rates for nitrified influents are anticipated to be highest, because the precursor conversions of organic and ammonia nitrogen have already occurred in pretreatment.

### Loading Considerations

Reductions in TN in treatment wetlands systems can be represented by loading analysis and the *P-k-C** model.

### Reduction of Total Nitrogen in FWS Wetlands

The median net period-of-record removal rate for 116 FWS systems receiving more than 5 mg/L TN is 129 g/m²·yr (Table 9.18). There is, however, wide variability among systems.

It is again useful to represent annual wetland performance as the effluent concentration produced ($C_o$) by a given inlet loading rate (LRI = HLR × $C_i$) and concentration ($C_i$). In the broad context, multiple data sets are represented by a trend that shows increasing $C_o$ with increasing LRI, with different groupings associated with each inlet concentration (Figure 9.29). The overall slope of the intersystem data on the log–log coordinates varies from near zero for low inlet concentrations to about 1.0 for high inlet concentrations.

**TABLE 9.18**
**Annual Reduction of Total Nitrogen in FWS Wetlands**

**Stipulations**

1. Data restricted to wetlands receiving inlet $C > 5$ mg/L TN.
2. Period of record averages are used in calculations.
3. For *k*-value calculations, the following *P-k-C** parameters are selected:
   a. $C^* = 1.5$ mg/L
   b. $P = 3$ TIS
4. Ranges of variables:

| | HLR (cm/d) | TN In (mg/L) | TN Out (mg/L) |
|---|---|---|---|
| **Mean** | 8.9 | 83.8 | 49.5 |
| **Median** | 4.3 | 17.4 | 10.2 |
| **Max** | 123.0 | 416.6 | 284.6 |
| **Min** | 0.2 | 5.2 | 1.7 |

**Results (N = 116 wetlands)**

| Percentile | Load Removed (g/m²·yr) | Rate Coefficient (m/yr) |
|---|---|---|
| 0.0 | 3 | 0.2 |
| 0.1 | 42 | 4.0 |
| 0.2 | 67 | 5.3 |
| 0.3 | 76 | 6.6 |
| 0.4 | 103 | 8.7 |
| 0.5 | 129 | 12.6 |
| 0.6 | 214 | 17.1 |
| 0.7 | 375 | 24.2 |
| 0.8 | 550 | 29.6 |
| 0.9 | 1,973 | 39.2 |
| 1.0 | 7,504 | 109.0 |

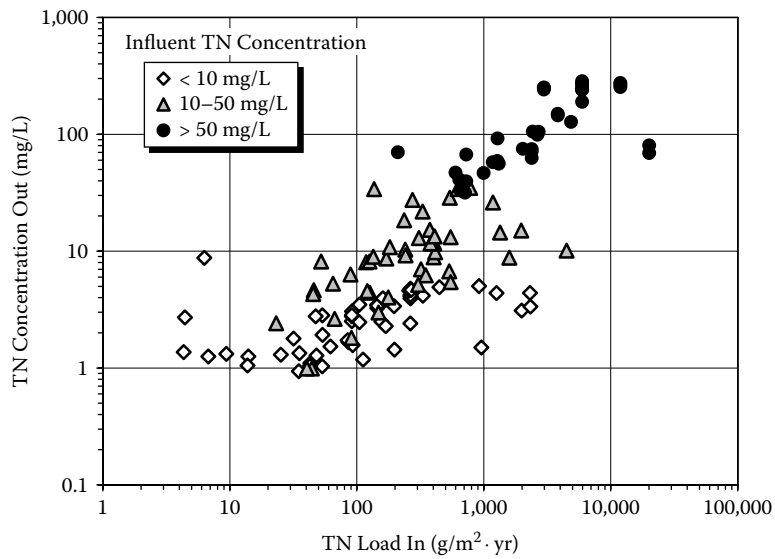



**FIGURE 9.29** Load–concentration plot for total nitrogen in FWS wetlands. Points are separated according to the inlet concentration range. Each point represents the entire period of record (POR) for one of 141 wetlands.

As for TKN and organic nitrogen, inlet loading is an insufficient design specification, because hydraulic load and inlet concentration are not interchangeable factors in the load representation.

## Reduction of Total Nitrogen in HSSF Wetlands

The median annual-average removal rate for 123 HSSF wetlands (197 system-years of data) is 273 g/m²·yr, as indicated in Table 9.19.

It is also useful to evaluate wetland performance ($C_o$) as a function of the inlet loading (Figure 9.30). Figure 9.30 represents data from 112 HSSF wetlands (198 system-years). In general, there is an overall upward trend of the outlet TN concentration ($C_o$) in response to the inlet TN loading, with a log–log slope of slightly less than 1.0. However, this apparent slope is in large measure due to the shift in inlet concentrations. When a particular inlet concentration group (like those shown on Figure 9.30) is considered, the change in outlet TN concentration is much less, as the intersystem slope for each concentration grouping is approximately 0.3. This has important design implications, because as the hydraulic loading to the wetland is decreased, the reduction in effluent concentration follows the slope of the inlet concentration group, not the overall data set. Use of the overall data set will overpredict the reductions in effluent TN concentrations as the hydraulic loading is decreased.

## Reduction of Total Nitrogen in VF Wetlands

Vertical flow wetlands typically transform organic and ammonia nitrogen to oxidized forms (nitrate and nitrite). As a result, effluents from VF wetlands are typically dominated by oxidized forms of nitrogen and the overall reduction of TN may be low (although the chemical form of nitrogen exiting the wetland may be very different than the chemical form entering the wetland).

**TABLE 9.19**
**Annual Reduction of Total Nitrogen in HSSF Wetlands**

**Stipulations**

1. The decomposition of 2,000 g/m²·yr of biomass causes production of 36 gN/m²·yr of organic nitrogen.
2. Annual averages are used in calculations.
3. For *k*-value calculations, the following *P*-*k*-*C** parameters are selected:
   a. $C^* = 1.0$ mg/L
   b. $P = 6$ TIS
4. Ranges of variables:

| | HLR (cm/d) | TN In (mg/L) | TN Out (mg/L) |
|---|---|---|---|
| **Mean** | 7.6 | 54.1 | 36.0 |
| **Median** | 4.9 | 41.4 | 25.6 |
| **Max** | 41.2 | 250.6 | 190.6 |
| **Min** | 1.1 | 6.8 | 3.4 |

**Results (N = 123; N·*t* = 198 wetland-years)**

| Percentile | Load Removed (g/m²·yr) | Rate Coefficient (m/yr) |
|---|---|---|
| 0.05 | −135.9 | −2.2 |
| 0.10 | 61.6 | 1.9 |
| 0.20 | 120.9 | 3.3 |
| 0.30 | 161.1 | 4.7 |
| 0.40 | 209.0 | 6.8 |
| 0.50 | 272.8 | 8.4 |
| 0.60 | 371.7 | 11.2 |
| 0.70 | 443.8 | 14.2 |
| 0.80 | 561.7 | 18.1 |
| 0.90 | 713.2 | 30.5 |
| 0.95 | 2,680.2 | 100.3 |

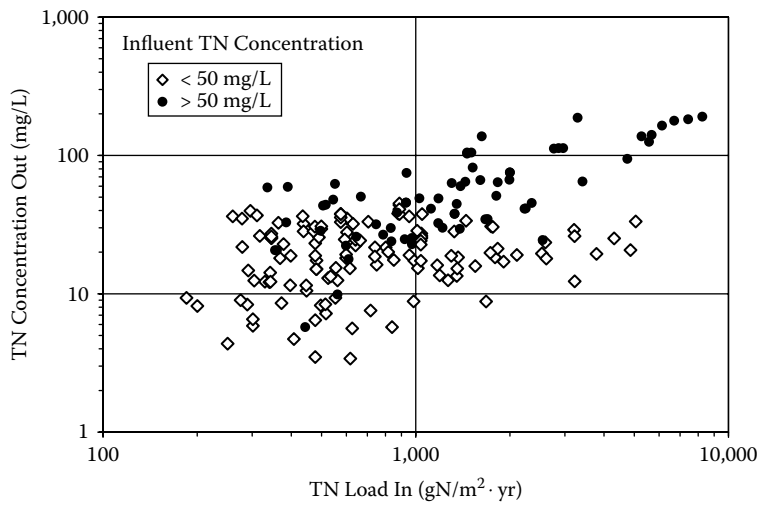



**FIGURE 9.30** Load response data for total nitrogen HSSF wetlands. Annual average information from 112 wetlands and 198 system-years is shown.

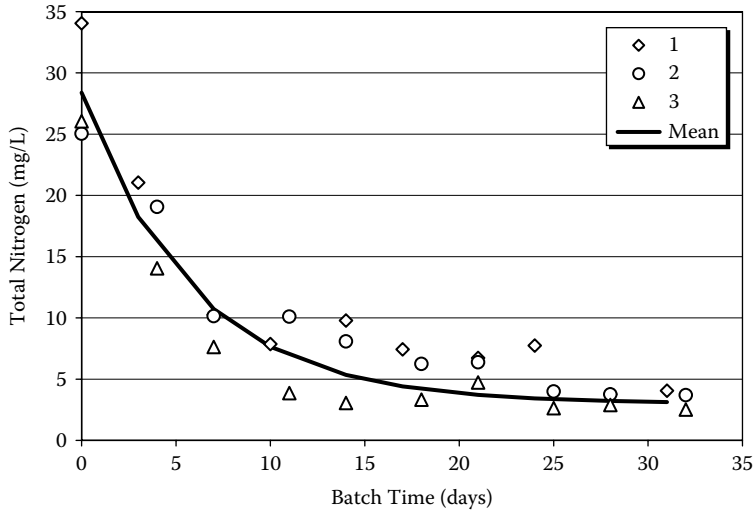



**FIGURE 9.31** Reduction of total nitrogen in the batch treatment FWS wetland systems at Humboldt, Saskatchewan. (Data from Lakhsman (1981) *A demonstration project at Humboldt to provide tertiary treatment to the municipal effluent using aquatic plants.* SRC Publication No. E-820-4-E-81: Saskatchewan Research Council. Graph from Kadlec and Knight (1996). *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

### Background Concentrations of Total Nitrogen

Treatment wetlands data display decreases in TN with contact time, which are consistent with first-order reduction kinetics; but show a nonzero background concentration for long detention. This is true for the time series of TN values when the system is operated in batch mode (Figure 9.31), and is also true for side-by-side systems of different detention times operated in flow through mode (Figure 9.32).

These nonzero plateaus of TN concentration at long detention times are consistent with the observed small background concentrations of organic nitrogen, and reflected in TKN. As shall be discussed in the following text, there are zero background concentrations for ammonia and oxidized nitrogen, so background TN is essentially the same as background organic nitrogen, typically in the range of 0.5–2.5 mg/L for all types of treatment wetlands.

### Rates and Rate Constants

In conventional activated sludge treatment system design, removal of TN is not directly modeled, but results from ammonification of the organic component, nitrification of the ammonia component, and denitrification of the oxidized nitrogen component. It is presumed in this section that TN removal in wetland treatment systems is an integrative measure of individual nitrogen transformations which can be approximated by area-based first-order rate expressions. This is supported by data such as that in Figures 9.31 and 9.32. Therefore, an area-based first-order removal rate is utilized here:

$$J_{TN} = k_{TN}(C_{TN} - C^*_{TN}) \tag{9.45}$$

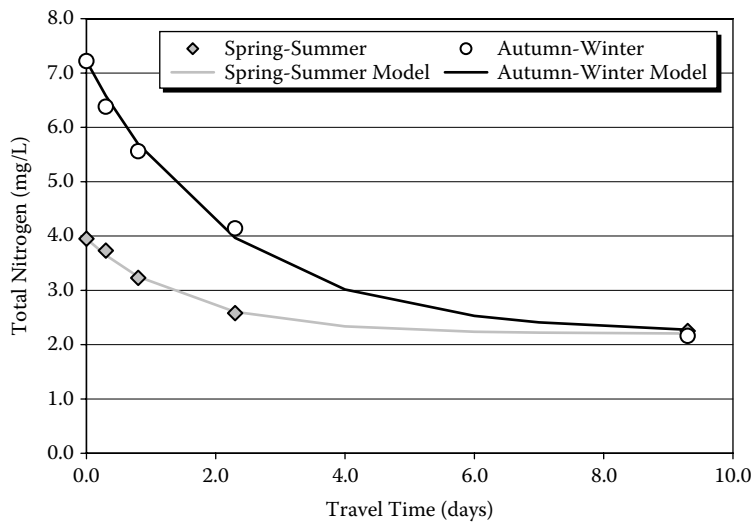



**FIGURE 9.32** Total nitrogen as a function of nominal detention time for the Texel wetlands. Replicate channels were operated side-by-side with different flow rates. (Data from Toet (2003) A Treatment Wetland Used for Polishing Tertiary Effluent from a Sewage Treatment Plant: Performances and Processes. Ph.D. Thesis, University of Utrecht, The Netherlands.)

where

$C_{TN}$ = wetland total nitrogen concentration, mg/L
$C^*_{TN}$ = background wetland total nitrogen concentration, mg/L
$J_{TN}$ = removal rate of total nitrogen, g/m²·yr
$k_{TN}$ = removal rate constant for total nitrogen, m/yr

The wetland environment may have actual hydraulics ranging from a few TIS up to a large number, approximating plug flow, depending on wetland configuration. Organic nitrogen is expected to show weathering effects as discussed in the previous text. Ammonia and nitrate are less liable to experience weathering, because these exist primarily in dissolved form, typically with only small contributions of particulate (sorbed) forms. Speculatively, the effective number of TIS (see Chapter 6) will be less than the tracer TIS, but by a slightly lesser margin than for organic nitrogen and TKN. Accordingly, the *P-k-C** model is chosen, with $P < N$.

## Total Nitrogen Rate Constants for FWS Wetlands

Results across systems for the assumed value $P = 3$ are summarized in this section. The value $C^* = 1.5$ mg/L is retained based upon organic nitrogen considerations. The remaining model parameter is the *k*-value, selected to fit the model:

$$\frac{C_{TN,out} - 1.5}{C_{TN,in} - 1.5} = \left(1 + \frac{k_{TN}}{3q}\right)^{-3} \tag{9.46}$$

Because of the selection of $C^* = 1.5$, parameter estimation is not reliable for low inlet concentrations, and those wetland with $C_{TN,in} < 5$ mg/L have been excluded from calibration. Out of 141 wetlands with data for TN (Figure 9.29), 116 met this criterion. The median annual rate constant was $k_{TN}$ = 12.6 m/yr (Table 9.18). The 10th–90th percentile range is 4.0–39.2 m/yr. There is a significant temperature dependence of TN *k*-values. Even on an average annual basis, temperature or season may be an important determinant of the rate constant, and is thus responsible for the some of the intersystem variability in annual *k*-values. Accordingly, it is necessary to examine intra-annual effects.

### *Microbially Dominated Wetlands*

There are typically clear downward trends in concentration along the flow path, as TN is removed from the water during travel through the wetland. Summer rates of decline are greater than in winter, implying that a temperature effect is present in these microbially dominated systems. It is not appropriate to use percentage reductions as a measure of seasonal performance, because of the confounding effects of seasonal flows, concentrations, and microbial activity. In many wetland systems, there are annual trends in input concentrations that often follow a sinusoidal tend, reflecting changes in the pretreatment and inlet water quality for that pretreatment wetland (Figure 9.33). For instance, a pretreatment plant conducting partial nitrification and partial denitrification will be more effective in summer, leading to less ammonia and less nitrate in the warmer months. A temperature coefficient (θ), is capable of accounting for these effects (see Chapter 6).

Results of calibration of *k*-values for entire periods of record for representative wetlands are summarized in Table 9.20. Monthly averages were used to avoid synoptic

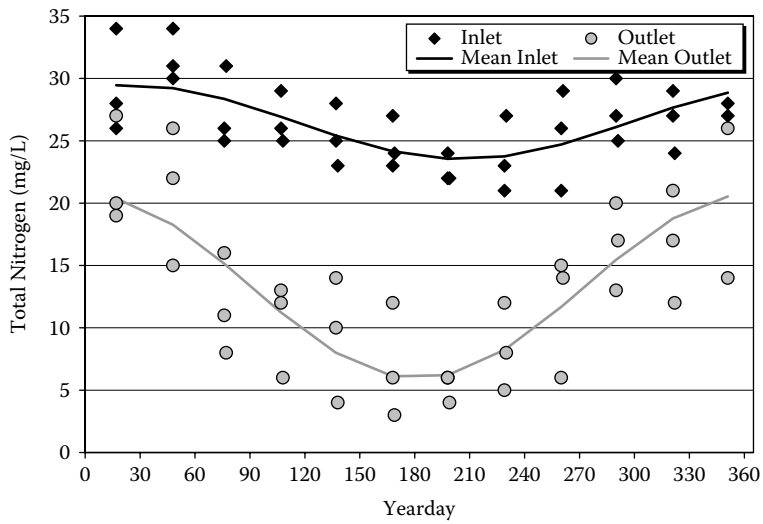



**FIGURE 9.33** Folded time series of inlet and outlet total nitrogen for the Linköping, Sweden, FWS wetlands. (From unpublished data, courtesy of K. Tonderski.)

| Cyclic parameters | Inlet | Outlet |
|---|---|---|
| $C_{mean}$ (mg/L) | 26.5 | 13.3 |
| *A* | 0.11 | 0.56 |
| $t_{max}$ (days) | 24 | 364 |
| $t_{min}$ (days) | 207 | 182 |

**TABLE 9.20**
**Dependence of Total Nitrogen Rate Constants on Temperature for FWS Systems**

| Site | Location | System | Years | Estimated Open Water (%) | TN In (mg/L) | TN Out (mg/L) | TN Load ($g/m^2 \cdot yr$) | DO (mg/L) | Annual (°C) | $P$ (TIS) | $C^*$ (mg/L) | $k_{20}$ (m/yr) | Theta |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Texel | The Netherlands | Pilot | 1 | 50 | 5.6 | 2.2 | 63–2,242 | 4.5–16 | 10.0 | ∞ | 2.2 | 115 | 1.050 |
| Tres Rios | Arizona | H1 pre | 2 | 25 | 5.6 | 2.6 | 268 | 1.3–3.1 | 18.9 | 6 | 0.8 | 49 | 1.061 |
| Tres Rios | Arizona | H2 pre | 2 | 25 | 5.6 | 2.4 | 264 | 1.4–3.1 | 18.9 | 6 | 0.8 | 43 | 1.084 |
| Tres Rios | Arizona | H1 post | 2 | 85 | 6.5 | 4.4 | 270 | 3 | 19.9 | 6 | 0.8 | 13 | 1.130 |
| Tres Rios | Arizona | H2 post | 2 | 90 | 6.5 | 4.8 | 308 | 3.0–4.2 | 20.7 | 6 | 0.8 | 11 | 1.103 |
| Imperial | California | All | 4 | 77 | 6.6 | 3.8 | 709 | 7.5–10.1 | 20.7 | ∞ | 1.25 | 76 | 1.048 |
| Brawley | California | All | 4 | 75 | 7.6 | 2.3 | 257 | 8.2–10.9 | 20.7 | ∞ | 1.25 | 52 | 1.080 |
| Lakeland | Florida | Cell 1 | 7 | 5 | 11.0 | 4.0 | 172 | 2.5–7.3 | 22.9 | 3 | 1.0 | 18 | 1.079 |
| Brighton | Ontario | All | 4 | 20 | 13.0 | 10.0 | 219 | — | 9.4 | 3 | 1.5 | 7 | 1.047 |
| Listowel | Ontario | 4 | 4 | 5 | 19.0 | 9.0 | 135 | 0.5–5 | 8.0 | 10 | 1.5 | 4 | 0.978 |
| Listowel | Ontario | 5 | 4 | 5 | 19.0 | 11.0 | 173 | 0.5–3.3 | 7.9 | 3 | 1.5 | 8 | 1.012 |
| Eskilstuna | Sweden | Ekeby | 3 | 80 | 20.0 | 15.0 | 1,175 | — | 6.5 | 6 | 1.5 | 25 | 1.049 |
| Linköping | Sweden | All | 3 | 10 | 27.0 | 13.0 | 410 | — | 6.7 | 3 | 1.5 | 42 | 1.081 |
| Kingman | Arizona | All | 5 | 4 | 33.0 | 13.0 | 519 | 2.8–7.3 | 11.7 | 20 | 1.5 | 25 | 1.050 |
| Warangal | India | All | 2 | — | 37.0 | 5.0 | 570 | 4 | 23.1 | 3 | 1.5 | 53 | 1.024 |
| Richmond | Australia | Pilot | 2 | 0 | 44.0 | 26.1 | 1,253 | 1.0–1.4 | 17.0 | 3 | 1.5 | 15 | 0.953 |
| New Hanover | North Carolina | Pilot | 4 | 10 | 135.0 | 64.0 | 750 | 3.3–4.2 | 19.4 | 2 | 1.5 | 5 | 1.082 |
| Hamilton (meat) | New Zealand | Pilot | 2 | 0 | 160.0 | 96.0 | 3,380 | 0.3–7.3 | 15.0 | 3 | 1.5 | 18 | 1.087 |
| **Median** | | | | | | | | | **18.0** | | **1.5** | **21.5** | **1.056** |

error (transit time offset). Calibrations were performed for best estimates of the internal hydraulics for each wetland. Therefore, *P*-values range from 2 (New Hanover, measured $P = N = 2$) to near plug flow conditions, based upon system geometry. The value $C^* = 1.5$ was used when data did not allow determination of a calibrated value. However, calibrated values were with a narrow range, 0.8–2.2 mg/L. The median $k_{20}$-value for TN is 21.5 m/yr, but the range is wide.

Temperature coefficients had a median value of 1.056, indicating a relatively strong thermal effect on the suite of microbial processes that contribute to TN reduction.

The wetlands in Table 9.20 are all continuous flow systems. Four storm water wetlands in Sweden were used to model TN disappearance (Arheimer and Wittgren, 1994). Nitrate comprised about half of the total, which was in the range $5 \le TN \le 20$ mg/L. The calibration spanned two calendar years. The model was a temperature sensitive, first-order areal model with a zero background concentration:

$$J_{TN} = \left[ r\, T_{10} \right] C_{TN} \tag{9.47}$$

where

$C_{TN}$ = total nitrogen concentration, mg/L = g/m$^3$
$J_{TN}$ = total nitrogen removal flux, g/m$^2$·d
$r$ = retention calibration factor, m/°C·d
$T_{10}$ = mean temperature for the last ten days, °C

The product $[rT_{10}]$ is equal to the first-order irreversible rate constant ($k$) for TN reduction. The value of $r = 0.0023$ m/°C·d calibrated data from the four wetlands over two years with $R^2 = 0.92$. Over the range $5 \le T \le 25$°C, the equivalent values are $k_{20} = 16.2$ m/yr and $\theta = 1.081$.

### *Agronomic Wetlands (Lightly Loaded Systems)*

When the TN loading to the wetland is less than the growth requirements of the plants and algae by a considerable margin, the removal of TN is very likely to be mediated by the growth and decay of biomass. As a rough guideline, this situation occurs for TN loadings less than approximately 120 gN/m$^2$·yr (Kadlec, 2005d). This occurs for over one-quarter (28%) of the 141 wetlands displayed in Figure 9.29. It is important to note that low inlet TN load very often means very low inlet TN concentration, close to background; and consequently, there is no ability to obtain meaningful calibrations of TN rate constants.

Uptake presumably occurs for both the ammonia and nitrate components of TN, and release may be considered to add to the organic component. As for TKN, plant uptake rates do not correspond to the annual cycle of water temperatures, and hence TN removal in agronomic wetlands cannot be characterized by modified Arrhenius θ-factors. The contrast between agronomic and microbial control of the TN rate constants is illustrated for New Hanover, North Carolina, and Listowel, Ontario, in Figure 9.34. The New Hanover TN loading was high, and the rate constants are seasonally synchronized with water temperature. But for Listowel, there is a high uptake in the spring growth period, which occurs at moderately cool temperatures, and no correspondence between monthly rate constants and temperature is present.

For the agronomic Listowel wetland, it would be appropriate to utilize monthly $k$-values:

| | | | | | | |
|---|---|---|---|---|---|---|
| $k_{TN,j}$ | = 4.2 | 2.4 | 4.0 | 11.9 | 24.3 | 13.7 m/yr |
| $j$ | = 1 | 2 | 3 | 4 | 5 | 6 |
| $k_{TN,j}$ | = 7.1 | 4.4 | 7.0 | 11.6 | 14.0 | 6.9 m/yr |
| $j$ | = 7 | 8 | 9 | 10 | 11 | 12 |

(9.48)

In any case, it is prudent to examine the actual seasonal progression of $k$-values, to ascertain whether a $\theta$-value approach is warranted or not. Additionally, if there is not a temperature correlation, the $R^2$ for a θ-factor regression will be very low.

### *Depth Effects*

The parameters of first-order models are referred to as "rate constants," but there is no *a priori* reason to believe that these very empirical "constants" do not in fact depend upon

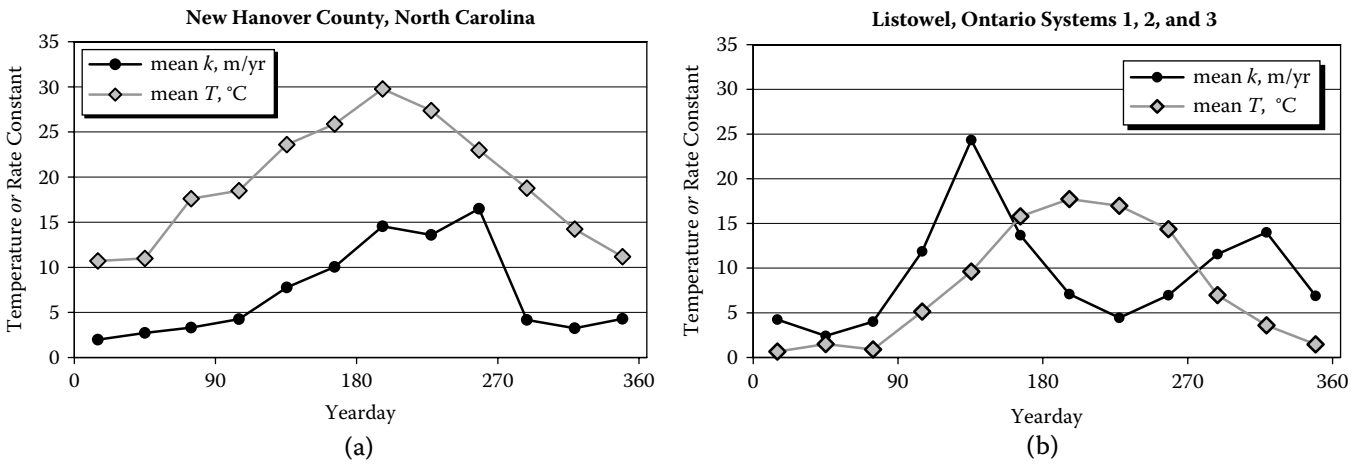



**FIGURE 9.34** The relationships between temperature and total nitrogen rate constants for two FWS wetlands. The Listowel total nitrogen loading was 111 g/m$^2$·yr, and New Hanover was 750 g/m$^2$·yr. Data for four years are represented in each case.

other operational characteristics of the wetland. The design variable of depth is indirectly involved in sizing computations using areal rate constants, but is directly involved if volumetric rate constants are employed. If $k$-values change with depth, then that effect must be accounted in design. The relation $k = (\varepsilon h)k_V$ requires that both $k$ and $k_V$ cannot be independent of depth.

If $k_V$ is constant with respect to depth, then $k$ is proportional to depth. That condition implies the removal of TN to be uniformly distributed vertically throughout the water column. If $k$ is constant, $k_V$ is inversely proportional to depth. That condition corresponds to removal apportioned to wetland surface area. Neither ideal extreme is likely to be present in a treatment wetland, but data often show FWS wetlands to behave with constant $k$, meaning that $k_V$ increases with decreasing depth. For instance, the reduction of TN in 17 side-by-side wetland cells at Jackson Bottoms, Oregon (SRI, 1990), shows $k_V$ inversely proportional to depth (Figure 9.35). The values $P = \infty$ (very long slender wetlands) and $C^* = 1.5$ mg/L were presumed in calibration. The implications for design are very important. *If the volumetric model is utilized in FWS calculations, there appears to be the option of increasing performance by increasing the water depth, and hence increasing the nominal detention time. However, that advantage is lost if the volumetric rate "constant" decreases with increasing depth, as indicated in Figure 9.35.*

## Total Nitrogen Rate Constants for HSSF Wetlands

Ranges in annual average $k$-rates for HSSF wetlands are summarized in Table 9.19, with the assumptions of $P$TIS = 6 and $C^*$ = 1.0 mg/L. The median $k$-rate is 8.4 m/yr. However, there is wide variability in the data, and the 10th–90th percentile range is 1.9–30.5 m/yr.

There appears to be little, if any, temperature dependence on TN removal in HSSF wetlands. Period of record data from 16 HSSF wetlands are presented in Table 9.21. The median $\theta$-factor was 1.005; and the 10th–90th percentile range is 0.990–1.029.

### Intrasystem Variability

In some treatment systems, TN must be reduced to regulatory limits. Regardless of the design method for nitrogen removal, the method should ensure regulatory compliance. Models represent only the seasonal trends of effluent TN concentrations, leaving a considerable amount of probabilistic scatter in performance (see Chapter 6). Monthly limits are the most common averaging period for regulatory compliance. Therefore, it is useful to examine the variability of monthly average outlet concentrations, here represented by a fractional addition to the trend value ($\Psi$):

$$C_{trend} = C_{avg}\left[1 + A\cos\left(\omega(t - t_{max})\right)\right] + E \tag{6.1}$$

$$\Psi = \frac{E}{C_{trend}} \tag{6.60}$$

where

$A$ = fractional amplitude of the seasonal cycle
$C$ = instantaneous monthly outlet concentration, mg/L
$C_{avg}$ = period of record average outlet concentration, mg/L
$C_{trend}$ = cyclic mean concentration, mg/L
$E$ = random portion of the outlet concentration, mg/L
$t$ = time of the year, Julian day
$t_{max}$ = time of the year for the maximum outlet concentration, Julian day
$\Psi$ = fractional addition to the trend value, dimensionless

The multiplier on the trend value is $(1 + \Psi)$. The set of monthly averages for a wetland over some period of record will yield a distribution of $\Psi$-values. Because we are interested in preventing or controlling exceedances, the upper percentile points of that distribution are useful in design. For instance, the 90th percentile represents the fractional addition to the trend that may be expected to occur one time out of ten during the period of record. Note that these $\Psi$-values are not

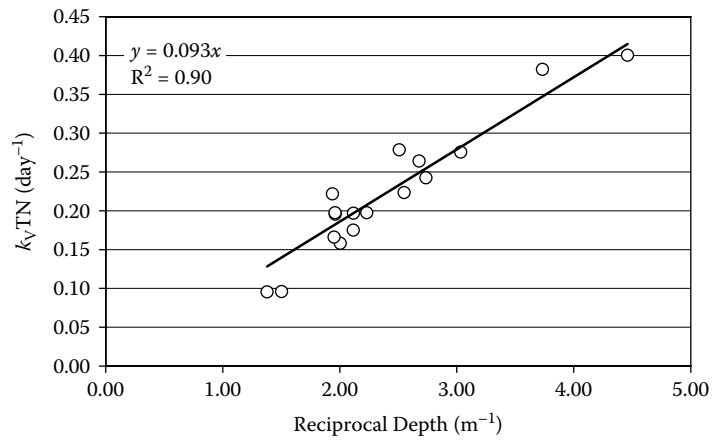



**FIGURE 9.35** Depth effect on the Jackson Bottoms, Oregon, FWS total nitrogen volumetric rate "constant."

**TABLE 9.21**
**Temperature Coefficients for Total Nitrogen Removal Rate Constants in HSSF Wetlands**

| Site | Reference | Cell | T range (°C) | Mean HLR (cm/d) | Mean $C_i$ (mg/L) | Mean $C_o$ (mg/L) | Theta |
|---|---|---|---|---|---|---|---|
| Richmond, NSW | Bavor *et al.* (1988) | Gravel | 11–24 | 3.8 | 44.1 | 21.6 | 1.002 |
| Richmond, NSW | Bavor *et al.* (1988) | *Typha* | 11–24 | 4.6 | 44.1 | 21.7 | 1.001 |
| Richmond, NSW | Bavor *et al.* (1988) | *Schoenoplectus* | 11–24 | 5.1 | 44.1 | 20.1 | 1.007 |
| Richmond, NSW | Bavor *et al.* (1988) | Mixed A | 11–24 | 4.6 | 44.1 | 18.6 | 1.006 |
| Richmond, NSW | Bavor *et al.* (1988) | Mixed B | 11–24 | 3.8 | 44.1 | 15.3 | 0.996 |
| Hardin, Kentucky | TVA, unpublished | 1 | 6–27 | 9.7 | 20.2 | 13.2 | 0.956 |
| Grand Lake, Minnesota | Unpublished | 1 | 1–17 | 1.0 | 59.4 | 41.2 | 1.013 |
| NERCC, Minnesota | Unpublished | 1 | 1–16 | 1.4 | 87.6 | 57.1 | 1.017 |
| NERCC, Minnesota | Unpublished | 2 | 1–16 | 1.4 | 87.6 | 59.8 | 1.031 |
| Lincoln, Nebraska | Vanier and Dahab (1997) | *Typha, Schoenoplectus* | 4–21 | 9.5 | 34.8 | 23.0 | 1.026 |
| North Yorkshire 1, U.K. | CWA database (2006) | 1 | 4–15 | 4.5 | 36.4 | 31.3 | 0.995 |
| Hamilton, New Zealand | Tanner *et al.* (1998b) | L1 | 10–25 | 1.5 | 55.9 | 18.6 | 1.062 |
| Hamilton, New Zealand | Tanner *et al.* (1998b) | L2 | 10–25 | 2.5 | 55.9 | 20.7 | 1.003 |
| Hamilton, New Zealand | Tanner *et al.* (1998b) | L3 | 10–25 | 3.3 | 55.9 | 24.1 | 1.000 |
| Hamilton, New Zealand | Tanner *et al.* (1998b) | L4 | 10–25 | 4.9 | 55.9 | 28.1 | 1.007 |
| Hamilton, New Zealand | Tanner *et al.* (1998b) | L5 | 10–25 | 6.9 | 55.9 | 31.0 | 0.985 |

| Percentile | Theta |
|---|---|
| 0.05 | 0.978 |
| 0.10 | 0.990 |
| 0.20 | 0.996 |
| 0.30 | 1.001 |
| 0.40 | 1.002 |
| 0.50 | 1.005 |
| 0.60 | 1.007 |
| 0.70 | 1.010 |
| 0.80 | 1.017 |
| 0.90 | 1.029 |
| 0.95 | 1.039 |

quite compliance percentiles, because of the finite number of months involved in a calibration set.

Table 9.22 lists percentiles of the monthly Ψ-distributions for some representative FWS wetlands. These data indicate that the median of the 95th percentile is an additional 70% above the trend, and that the median of the 90th percentile is an additional 55%.

To incorporate this variability into design, the wetland designer must oversize the wetland. Suppose it is necessary to meet a monthly limit of 3.0 mg/L of TN at least nine times out of ten (90th percentile). The designer should proceed in three steps:

1. Use seasonal variations, either temperature-driven for microbial wetlands or month-driven for agronomic wetlands, to establish the bottleneck period, which is likely to be winter unless there are large reductions of inflow rates and concentrations in winter. The month with the largest outlet TN is selected as the controlling month.
2. Apply a multiplier to account for the random part of the effluent concentration distribution during the controlling month. For instance, this

**TABLE 9.22**
**Trend Multipliers for Effluent Total Nitrogen Concentrations in FWS Wetlands**

| System | Years of Data | Excursion Frequency 50% | 80% | 90% | 95% |
|---|---|---|---|---|---|
| Brawley, California | 4 | 0.95 | 1.48 | 1.79 | 2.03 |
| Brighton, Ontario | 4 | 1.05 | 1.18 | 1.21 | 1.24 |
| Kingman, Arizona | 5 | 0.97 | 1.60 | 1.79 | 2.56 |
| Lakeland 1, Florida | 7 | 0.83 | 1.24 | 1.49 | 1.60 |
| Linköping, Sweden | 3 | 0.98 | 1.21 | 1.28 | 1.35 |
| Listowel 3, Ontario | 4 | 0.86 | 1.29 | 1.65 | 1.72 |
| Listowel 4, Ontario | 4 | 0.94 | 1.25 | 1.62 | 1.89 |
| New Hanover, North Carolina | 4 | 0.97 | 1.20 | 1.43 | 1.71 |
| Orlando Easterly Wetland, Florida | 9 | 0.99 | 1.20 | 1.36 | 1.46 |
| Tres Rios H1, Arizona | 6 | 0.96 | 1.42 | 1.65 | 1.70 |
| **Median** | | **0.96** | **1.25** | **1.55** | **1.70** |
| **Mean** | | **0.95** | **1.31** | **1.53** | **1.73** |

*Note:* Data are approximately monthly. The multiplier on the trend concentration is $(1 + \Psi)$; see Equation 6.61. For instance, one out of ten months, we can expect a total nitrogen concentration 1.55 times the long-term mean value based on the median of the 10 wetlands.

could be the median multiplier of 1.70 from Table 9.22.

3. Increase the wetland size so that the calculated seasonally variable effluent concentration does not exceed 3/(1.70) = 1.76 mg/L during the bottleneck period.

The reader should note that this correction is not a safety factor. It is known from the nature of existing data sets that excursions will occur. It becomes necessary to plan for those excursions, so that potential exceedances occur at less than the predetermined frequency. Wetland data may sometimes contain sufficient intrasystem variability to place the 100th percentile above the median inlet concentration. Thus, it may not be possible to design a wetland to totally avoid the possibility of monthly exceedances.

## 9.9 PERFORMANCE FOR AMMONIA

Ammonia is an intermediate in the sequential processing of nitrogen in treatment wetlands, which is produced by ammonification of organic nitrogen, and oxidized by aerobic and possibly anaerobic processes. Because of toxicity of un-ionized ammonia in receiving aquatic ecosystems, this nitrogen species is often singled out for regulation. The fraction of unionized ammonia depends upon water temperature as well as total dissolved ammonia, and hence regulation may be seasonal, with lower concentration limits in summer months.

Percent removal is an inadequate measure of ammonia performance. Indeed, negative removals may result, for two reasons. First, because ammonia is an intermediate in the processing sequence, production may exceed removal. This effect may be seen in the transect studies for the FWS wetland at Listowel, Ontario, in which ammonification produces ammonia in the inlet zones (Figure 9.36). Second, the analytical limits for the determination of ammonia in the laboratory may skew the difference between values at or below the selected detection limit, especially for low influent concentrations. As a result, 30 of 208 FWS wetlands reporting ammonia data show negative removals, and even higher fractions of systems in smaller databases have been reported to have negative removals (Reed *et al.*, 1995).

### Reduction of Ammonia in FWS Wetlands

Reductions of ammonia in treatment wetlands systems can be represented by loading analysis and the *P-k-C** model. The median net period-of-record removal rate for 118 FWS systems receiving more than 1 mg/L ammonia nitrogen is 127 g/m$^2$·yr (Table 9.23). There is, however, wide variability among systems.

The load response graph for ammonia reflects the effluent ammonia concentration produced ($C_o$) by a given TKN LRI (= HLR × $C_i$) and ammonia concentration ($C_i$). Multiple data sets are represented by a trend that shows increasing $C_o$ with increasing LRI, with different groupings associated with each inlet concentration (Figure 9.37). The overall slope of the intersystem data on the log–log coordinates is about 1.0 for all inlet concentrations. There is not an apparent lower limit to exit ammonia concentrations, indicating a near-zero background for this species. As for TKN and TN, inlet loading is an insufficient design specification, because hydraulic load and inlet concentration are not interchangeable factors in the load representation.

### Implied Oxygen Supply in FWS Wetlands

Because all mechanisms of ammonia reduction require oxygen to varying degrees, it is useful to speculate on the amount consumed in the FWS wetland database. Most studies of ammonia removal in constructed wetlands assume the occurrence of the "classical" sequence of autotrophic nitrification followed by respiratory denitrification. The nitrification step requires 4.3 gO/gN, as well as an additional 0.3 gO/gN to supply the organisms.

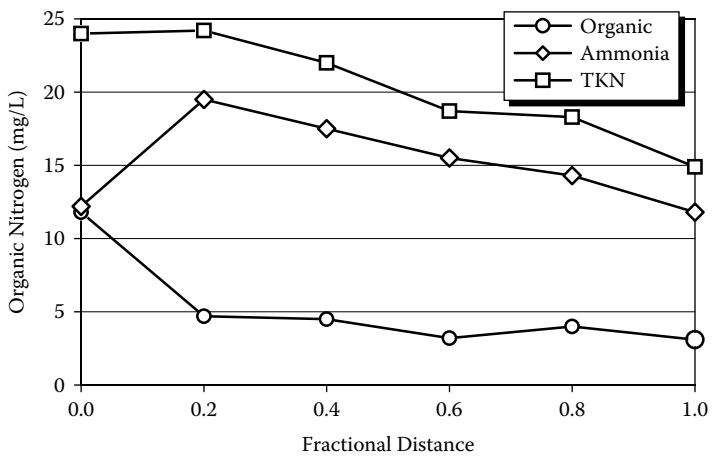



**FIGURE 9.36** Profiles of major dissolved nitrogen species in Listowel, Ontario, FWS Wetland 4 in Summer 1984. Lines are model calibration, symbols denote data points which are averages of biweekly data over three months. (From Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

## TABLE 9.23
## Annual Reduction of Ammonia Nitrogen in FWS Wetlands

**Stipulations**

1. Data restricted to wetlands receiving inlet $C > 1$ mg/L ammonia nitrogen.
2. Period of record averages are used in calculations.
3. For $k$-value calculations, the following $P$-$k$-$C^*$ parameters are selected:
   a. $C^* = 0$ mg/L
   b. $P = 3$ TIS
4. Sequential conversion of organic to ammonia nitrogen is accounted.
5. An annual vegetative nitrogen uptake of 40 g/m²·yr is apportioned to ammonia and nitrate according to their mean concentrations.
6. 90% of the vegetative uptake is recycled, and 10% permanently buried.

| | HLR (cm/d) | Org-N In (mg/L) | Org-N Out (mg/L) | $NH_4$–N In (mg/L) | $NH_4$–N Out (mg/L) |
|---|---|---|---|---|---|
| **Mean** | 7.3 | 10.5 | 4.9 | 75.3 | 45.5 |
| **Median** | 4.1 | 5.7 | 2.7 | 15.5 | 7.3 |
| **Max** | 110.0 | 69.5 | 29.6 | 405.5 | 279.3 |
| **Min** | 0.3 | 0.0 | 0.0 | 1.2 | 0.1 |

**Results (N = 118 wetlands)**

| Percentile | Load Removed (g/m²·yr) | Rate Coefficient (m/yr) |
|---|---|---|
| 0.0 | 7 | 0.4 |
| 0.1 | 37 | 4.7 |
| 0.2 | 51 | 7.0 |
| 0.3 | 68 | 8.7 |
| 0.4 | 85 | 11.1 |
| 0.5 | 127 | 14.7 |
| 0.6 | 233 | 26.1 |
| 0.7 | 387 | 45.1 |
| 0.8 | 823 | 59.2 |
| 0.9 | 1,941 | 85.6 |
| 1.0 | 4,680 | 258.5 |

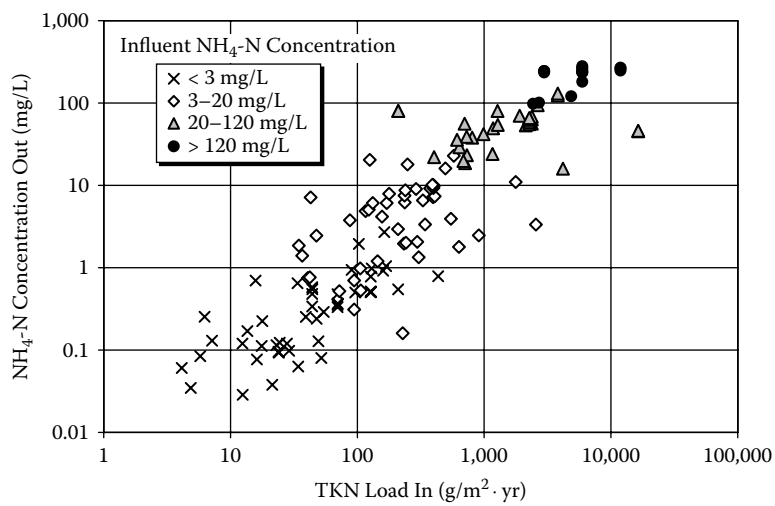



**FIGURE 9.37** Load–concentration plot for ammonia nitrogen in FWS wetlands. Inlet TKN loading is used, because ammonification can add to the production of ammonia. Points are separated according to the inlet ammonia concentration range. Each point represents the entire period of record (POR) for one of 149 wetlands.

**TABLE 9.24**
**Oxygen Usage in FWS Wetlands as a Function of Assumed Stoichiometry of BOD Reduction and Ammonia Removal**

| Percentile | BODLR[a] (gO/m²·d) | Internal ALR[b] (gN/m²·d) | External ALR[b] (gN/m²·d) | Maximum[c] O Usage (gO/m²·d) | Intermediate[c] O Usage (gO/m²·d) | Minimum[c] O Usage (gO/m²·d) |
|---|---|---|---|---|---|---|
| 0.05 | 0.00 | 0.02 | –0.31 | –0.05 | –0.02 | –0.08 |
| 0.10 | 0.01 | 0.05 | –0.19 | 0.09 | 0.04 | –0.03 |
| 0.20 | 0.05 | 0.05 | –0.10 | 0.35 | 0.17 | 0.01 |
| 0.30 | 0.16 | 0.07 | –0.06 | 0.62 | 0.33 | 0.04 |
| 0.40 | 0.37 | 0.09 | –0.03 | 0.98 | 0.52 | 0.10 |
| 0.50 | 0.86 | 0.12 | 0.02 | 1.47 | 0.91 | 0.19 |
| 0.60 | 1.12 | 0.16 | 0.20 | 3.65 | 1.89 | 0.53 |
| 0.70 | 1.93 | 0.25 | 0.48 | 6.92 | 3.92 | 1.02 |
| 0.80 | 2.80 | 0.34 | 1.14 | 16.72 | 7.59 | 3.39 |
| 0.90 | 5.17 | 0.67 | 4.88 | 43.14 | 17.22 | 8.79 |
| 0.95 | 24.11 | 0.86 | 6.90 | 50.30 | 27.44 | 12.10 |

*Note:* Both internal (from organic N) and external ammonia loads are considered. The Anammox route requires half the ammonia to be converted to nitrite, which needs approximately 1.7 gO/gN. BOD may be reduced by anaerobic or aerobic processes. The data represent period of record data from 95 wetlands.

[a] BODLR = BOD load removed.

[b] ALR = ammonia load removed.

[c] The maximum case assumes 1.5 gO/gBOD and 4.6 gO/g$NH_4$–N.
The intermediate case assumes 1.0 gO/gBOD and 1.7 gO/g$NH_4$–N.
The minimum case assumes 0.0 gO/gBOD and 1.7 gO/g$NH_4$–N.

As noted in Chapter 8, there is similar doubt that BOD reduction is solely due to microbial processes utilizing free oxygen, since anaerobic processes (such as fermentation) will also remove BOD. More accurate estimation would allow for settlement of solid BOD, and consumption in denitrification (Cooper, 1999). Nevertheless, if an upper limit to the necessary oxygen is sought, then CBOD reduction via aerobic degradation must be added to the requirement, and included in the overall oxygen demand.

***Note:*** *The assumption of "classical" nitrification/denitrification coupled with assumption of aerobic degradation of BOD will always result in the highest estimate of the implied oxygen supply.*

The use of "alternate" nitrogen chemistry, coupled with different ranges of BOD degradation (aerobic or anaerobic) will result in estimates of the implied oxygen supply considerably lower than that associated with "classical" nitrification and denitrification. Table 9.24 shows the distribution of implied oxygen supply to FWS wetlands under several different stoichiometric assumptions.

As seen in Table 9.24, FWS wetlands are relatively oxygen-transfer limited systems, with a median implied transfer rate of 1.47 gO/m²·d for the most optimistic stoichiometric assumption (1.5 gO/gBOD; 4.6 gO/g$NH_4$–N). However, there is a wide range in the data set; the 10th and 90th percentiles are 0.09–43.14 gO/m²·d for the most optimistic stoichiometry. Values at the higher probability levels are presumably increased by factors such as wind mixing.

## Reduction of Ammonia in HSSF Wetlands

The median annual-average removal rate for HSSF wetlands (213 system-years of data) is 208 g/m²·yr, as indicated in Table 9.25.

It is also useful to evaluate wetland performance ($C_o$) as a function of the inlet loading (see Figure 9.38). Figure 9.38 represents data from 112 HSSF wetlands (198 system-years). In general, there is an overall upward trend of the outlet ammonia concentration ($C_o$) in response to the inlet TKN loading, with a log–log slope of approximately 0.5. However, this apparent slope is due in a large measure due to the shift in inlet concentrations. When a particular inlet concentration group (like those shown in Figure 9.38) is considered, the change in outlet ammonia concentration is much less, as the intersystem slope for each concentration grouping is approximately 0.25. This has important design implications, because as the hydraulic loading to the wetland is decreased, the reduction in effluent concentration follows the slope of the inlet concentration group, not the overall data set. Use of the overall data set will overpredict the reductions in effluent ammonia concentrations as the hydraulic loading is decreased.

### Plant Uptake of Ammonia

"Conventional wisdom" is that nitrification is the dominant mechanism for ammonia reduction in HSSF wetlands (U.S. EPA, 2000a; Crites *et al.*, 2006). However, plant uptake can have an effect in some cases. To place this in perspective, the growth of biomass during the year is assigned to be

**TABLE 9.25**
**Annual Implied Nitrification of Ammonia Nitrogen in HSSF Wetlands**

**Stipulations**

1. The decomposition of 2,000 g/m$^2$·yr of biomass causes production of 36 gN/m$^2$·yr of organic nitrogen.
2. Annual averages are used in calculations.
3. For *k*-value calculations, the following *P-k-C** parameters are selected:
   a. $C^* = 0$ mg/L
   b. $P = 6$ TIS
4. Ranges of variables:

| | HLR (cm/d) | $NH_4$–N In (mg/L) | $NH_4$–N Out (mg/L) |
|---|---|---|---|
| **Mean** | 7.0 | 39.7 | 28.3 |
| **Median** | 4.7 | 24.0 | 16.1 |
| **Max** | 41.2 | 229.5 | 182.0 |
| **Min** | 0.1 | 0.3 | 0.1 |

**Results (N·*t* = 213 wetland-years)**

| Percentile | Load Removed (g/m$^2$·yr) | Rate Coefficient (m/yr) |
|---|---|---|
| 0.05 | −2 | 0.0 |
| 0.10 | 19 | 0.4 |
| 0.20 | 69 | 2.7 |
| 0.30 | 123 | 5.2 |
| 0.40 | 161 | 8.4 |
| 0.50 | 208 | 11.4 |
| 0.60 | 258 | 15.0 |
| 0.70 | 333 | 18.8 |
| 0.80 | 438 | 29.8 |
| 0.90 | 573 | 63.3 |
| 0.95 | 770 | 133.3 |

2,000 g/m$^2$·yr (this has been shown to be a representative value in Chapter 3). At a tissue nitrogen of 2%, 40 gN/m$^2$·yr are used, of which the large majority (approximately 90%) is eventually returned via decomposition. A large HSSF database has been examined for the proportions of plant uptake and nitrification. In this analysis, an allowance has been made for ammonification of organic nitrogen as an internal source of ammonia. It is found that in 25% of the 117 planted HSSF wetlands surveyed, plant uptake is at least a quarter of the annual nitrification rate. Thus, the designer needs to be aware that at low ammonia loading rates (the approximate critical level has earlier been suggested to be 120 gN/m$^2$·yr), plant uptake may be a factor in the processing of ammonia.

## Implied Oxygen Supply in HSSF Wetlands

Because all mechanisms of ammonia reduction require oxygen to varying degrees, it is useful to speculate on the amount consumed in the HSSF wetlands in the database. Most studies of ammonia removal in constructed wetlands assume the occurrence of the "classical" sequence of autotrophic nitrification followed by respiratory denitrification. The nitrification step requires 4.3 gO/gN, as well as an additional 0.3 gO/gN to supply the microorganisms.

The result is an "implied oxygen requirement" of the decrease in CBOD (aerobically) plus 4.6 times the nitrification rate. However, a number of authors have used the decrease in ammonia rather than the mass balance nitrification rate (Cooper, 1999; Platzer, 1999; Noorvee *et al*., 2005b). This estimate is high because of the omission of plant uptake of ammonia, but low because of the internal ammonia production assignable to the loss of organic nitrogen. Here, the mass balance approach is used to estimate nitrification, which corrects for these effects. However, the artifact of alternative mechanisms of ammonia processing remains. The assumption of "classical" nitrification/denitrification, coupled with

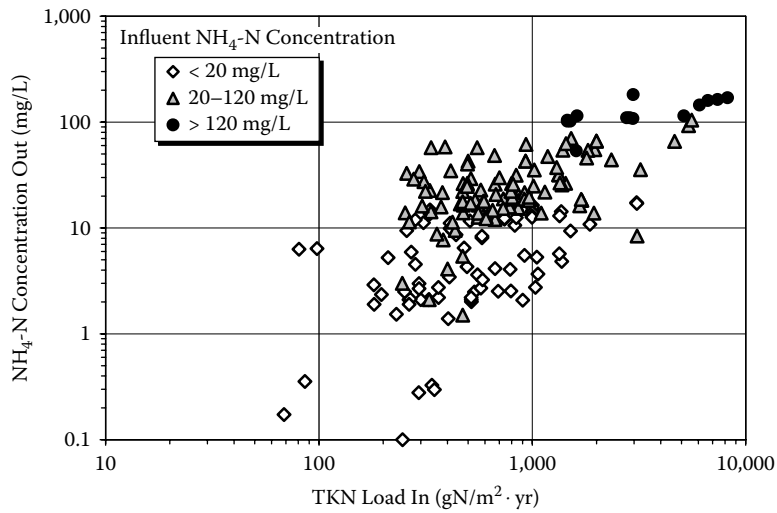



**FIGURE 9.38** Load response data for HSSF wetlands. The loading includes organic nitrogen, because of the potential for ammonia production via ammonification. Annual average information from 112 wetlands and 198 wetland-years is shown.

assumption of aerobic degradation of BOD, will always result in the highest estimate of the implied oxygen supply.

However, nitrification–denitrification and organic carbon removal are closely coupled in treatment wetlands and nitrogen transformation intermediates (such as $NO_2^-$ and $NO_3^-$) rarely accumulate in HSSF wetland beds. There is increasing evidence that in such oxygen-limited environments, nitrification, denitrification, and other microbial processes (e.g., methane oxidation) may be much more closely coupled (also described as integrated or simultaneous processes) and may include a range of alternative and co-metabolic pathways. Examples of potential alternative pathways with reduced overall oxygen requirements that have relevance to treatment wetlands have been discussed above, and include: oxygen-limited autotrophic nitrification–denitrification, and anaerobic ammonium oxidation (anammox). There is also the possibility of oxidation of ammonium by heterotrophs deriving energy from organic substrates.

The use of "alternate" nitrogen chemistry, coupled with different ranges of BOD degradation (aerobic or anaerobic), will result in estimates of the implied oxygen supply considerably lower than that associated with "classical" nitrification and denitrification. Table 9.26 shows the distribution of implied oxygen supply to HSSF wetlands under several different stoichiometric assumptions.

As seen in Table 9.26, HSSF wetlands are relatively oxygen-transfer limited systems, with a median implied transfer rate of 6.3 gO/m$^2$·d for the most optimistic stoichiometric assumption (1.5 gO/g BOD; 4.6 gO/g$NH_4$–N). However, there is a wide range in the data set; the 10th and 90th percentiles are 2.1–21.1 gO/m$^2$·d for the most optimistic stoichiometry, which is probably an overestimate in the case of HSSF wetlands. The use of "alternate" nitrogen stoichiometries results in much lower implied oxygen transfers.

## Oxygen Transfer—Plants or Atmospheric Diffusion?

Nothing has been more controversial in the literature of wetland ammonia removal than the ongoing discussions of the role of plants in oxygen supply. One side of this issue is represented by the idea that physical oxygen transfer is negligible, and that plant oxygenation is responsible for the entirety of ammonia oxidation. The logical extension of this point of view is that the more roots, the more oxygen transfer. Accordingly, sources such as U.S. EPA (1993f), Reed *et al*. (1995), and Crites *et al*. (2006) have promulgated a universal root-volumetric oxygen transfer rate of 7.5 gO/m$^3$·d, based upon data from the Santee, California, test facility. For a 45-cm rooting depth, and 100% coverage of the wetland, this amounts to an areal delivery of 3.4 gO/m$^2$·d. This concept places a premium on both deep, extensive rooting and healthy, complete coverage. The Santee study (Gersberg *et al*., 1986) found that *Phragmites* and bulrushes "remained very healthy throughout our study," but cattails "showed a marked yellowing, and most had died after six months."

McGechan *et al*. (2005a, b) modeled six horizontal layers in a HSSF system in West Harwood, Scotland, and allowed for both atmospheric aeration and plant aeration. Although it was possible to reproduce the longitudinal profiles of nitrogen species, they could not accurately partition the oxygen supply between the two routes. The implied oxygen supply was 8 gO/m$^2$·d.

Table 9.27 shows the results of side-by-side studies with planted and unplanted HSSF wetlands. On average, the

**TABLE 9.26**
**Oxygen Usage in HSSF Wetlands as a Function of Assumed Stoichiometry of BOD Reduction and Ammonia Removal**

| Percentile | BODLR[a] (gO/m$^2$·d) | Internal ALR[b] (gN/m$^2$·d) | External ALR[b] (gN/m$^2$·d) | Maximum[c] O Usage (gO/m$^2$·d) | Intermediate[c] O Usage (gO/m$^2$·d) | Minimum[c] O Usage (gO/m$^2$·d) |
|---|---|---|---|---|---|---|
| 0.05 | 0.36 | –0.01 | –0.14 | 0.9 | 0.6 | 0.0 |
| 0.10 | 0.72 | 0.05 | –0.04 | 2.1 | 1.2 | 0.2 |
| 0.20 | 1.11 | 0.11 | 0.01 | 3.6 | 1.7 | 0.4 |
| 0.30 | 1.40 | 0.15 | 0.14 | 4.2 | 2.2 | 0.6 |
| 0.40 | 1.88 | 0.20 | 0.24 | 5.2 | 2.7 | 0.8 |
| 0.50 | 2.17 | 0.23 | 0.28 | 6.3 | 3.2 | 1.0 |
| 0.60 | 2.93 | 0.29 | 0.38 | 8.5 | 4.4 | 1.2 |
| 0.70 | 4.22 | 0.35 | 0.57 | 10.6 | 5.4 | 1.5 |
| 0.80 | 5.28 | 0.47 | 0.73 | 12.8 | 7.5 | 2.0 |
| 0.90 | 10.58 | 0.63 | 1.09 | 21.1 | 12.8 | 2.6 |
| 0.95 | 18.54 | 0.78 | 1.52 | 38.2 | 23.0 | 3.6 |

*Note:* Both internal (from organic N) and external ammonia loads are considered. The Anammox route requires half the ammonia to be converted to nitrite, which needs approximately1.7 gO/gN. BOD may be reduced by anaerobic or aerobic processes. The data represent 85 wetlands and 168 wetland-years of data.

[a] BODLR = BOD load removed.

[b] ALR = ammonia load removed.

[c] The maximum case assumes 1.5 gO/gBOD and 4.6 gO/g$NH_4$–N.
The intermediate case assumes 1.0 gO/gBOD and 1.7 gO/g$NH_4$–N.
The minimum case assumes 0.0 gO/gBOD and 1.7 gO/g$NH_4$–N.

## TABLE 9.27
## Maximum Implied Oxygen Requirements for Side-by-Side Studies of HSSF Wetlands with and without Plants

| Reference | Site Name | System Name | Cell | Year | BODLR ($g/m^2 \cdot d$) | Nitrification ($g/m^2 \cdot d$) | Total Oxygen Implied | Unplanted/Planted Ratio |
|---|---|---|---|---|---|---|---|---|
| Theis and Young (2000) | Minoa, New York | E2.1A | U | 1996–97 | 4.4 | 2.2 | 6.6 | — |
| Theis and Young (2000) | Minoa, New York | E1.2A | S | 1996–97 | 4.4 | 1.9 | 6.3 | 1.045 |
| Theis and Young (2000) | Minoa, New York | E1.1A | P | 1996–97 | 4.4 | 2.5 | 6.9 | 0.956 |
| Theis and Young (2000) | Minoa, New York | E2.2A | P | 1996–97 | 4.4 | 1.9 | 6.3 | 1.058 |
| Akratos and Tsihrintzis (2007) | Thrace, Greece | MG-Z | Control | 2004–06 | 3.3 | 0.99 | 4.3 | — |
| Akratos and Tsihrintzis (2007) | Thrace, Greece | MG-C | Cattail | 2004–06 | 3.4 | 2.03 | 5.4 | 0.789 |
| Akratos and Tsihrintzis (2007) | Thrace, Greece | MG-R | Reed | 2004–06 | 3.2 | 1.65 | 4.9 | 0.872 |
| Bavor *et al.* (1988) | Richmond, NSW | — | Gravel | 1984–86 | 2.4 | 5.4 | 7.7 | — |
| Bavor *et al.* (1988) | Richmond, NSW | — | *Typha* | 1984–86 | 2.1 | 4.9 | 6.9 | 1.116 |
| Bavor *et al.* (1988) | Richmond, NSW | — | *Schoenoplectus* | 1984–86 | 2.4 | 5.6 | 8.0 | 0.974 |
| Bavor *et al.* (1988) | Richmond, NSW | — | Mix A | 1984–86 | 2.2 | 5.7 | 7.9 | 0.981 |
| Bavor *et al.* (1988) | Richmond, NSW | — | Mix B | 1984–86 | 1.8 | 5.2 | 7.0 | 1.104 |
| Tanner (1996) | Hamilton, New Zealand | 1 | Unplanted | 1990–91 | 1.6 | 1.1 | 2.7 | — |
| Tanner (1996) | Hamilton, New Zealand | 1 | Bulrushes | 1990–91 | 2.2 | 2.5 | 4.7 | 0.581 |
| Tanner (1996) | Hamilton, New Zealand | 2 | Unplanted | 1990–91 | 1.6 | 1.2 | 2.8 | — |
| Tanner (1996) | Hamilton, New Zealand | 2 | Bulrushes | 1990–91 | 1.5 | 1.6 | 3.0 | 0.926 |
| Tanner (1996) | Hamilton, New Zealand | 3 | Unplanted | 1990–91 | 1.0 | 0.6 | 1.6 | — |
| Tanner (1996) | Hamilton, New Zealand | 3 | Bulrushes | 1990–91 | 1.1 | 1.2 | 2.3 | 0.703 |
| Tanner (1996) | Hamilton, New Zealand | 4 | Unplanted | 1990–91 | 0.9 | 0.6 | 1.5 | — |
| Tanner (1996) | Hamilton, New Zealand | 4 | Bulrushes | 1990–91 | 1.0 | 1.0 | 2.0 | 0.787 |
| Van Oostrom and Cooper (1990) | Hamilton Horotiu | — | Gravel | 1988–89 | 8.8 | 5.8 | 14.7 | — |
| Van Oostrom and Cooper (1990) | Hamilton Horotiu | — | *Schoenoplectus* | 1988–89 | 10.2 | 6.8 | 17.0 | 0.862 |
| Van Oostrom and Cooper (1990) | Hamilton Horotiu | — | *Glyceria* | 1988–89 | 9.4 | 5.8 | 15.2 | 0.963 |
| Van Oostrom and Cooper (1990) | Hamilton Horotiu | — | Gravel | 1989–90 | 5.1 | 3.5 | 8.6 | — |
| Van Oostrom and Cooper (1990) | Hamilton Horotiu | — | *Schopenoplectus* | 1989–90 | 5.3 | 3.8 | 9.2 | 0.938 |
| Van Oostrom and Cooper (1990) | Hamilton Horotiu | — | *Glyceria* | 1989–90 | 5.0 | 3.7 | 8.7 | 0.991 |
| Gersberg *et al.* (1984) | Santee, California | G | — | — | 3.8 | 1.0 | 4.8 | — |
| Gersberg *et al.* (1984) | Santee, California | CT | — | — | 4.5 | 1.9 | 6.4 | 0.750 |
| Gersberg *et al.* (1984) | Santee, California | BR | — | — | 5.3 | 5.4 | 10.7 | 0.449 |
| Gersberg *et al.* (1984) | Santee, California | Phrag | — | — | 4.5 | 4.5 | 9.0 | 0.532 |
| **Mean** | | | | | **3.69** | **3.09** | **6.78** | **0.869** |
| **Standard deviation** | | | | | **2.47** | **1.99** | **3.96** | **0.188** |

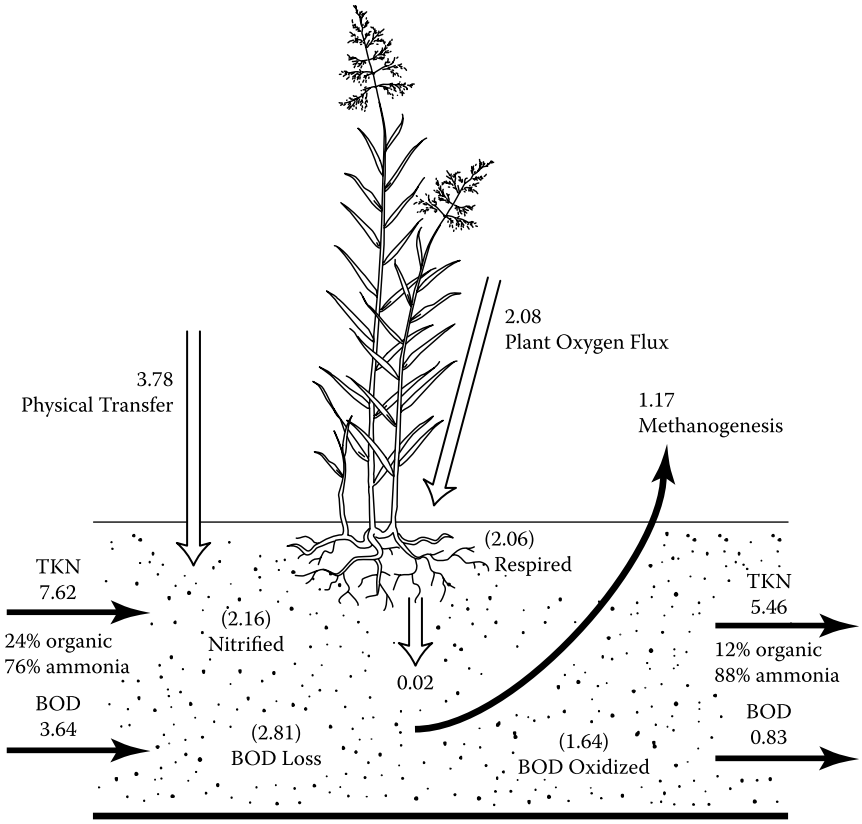



**FIGURE 9.39** Fluxes of oxygen sources and sinks in a HSSF wetland, the latter expressed as oxygen equivalents. See text for explanation. Reinterpretation of Brix (1990) data from Kalø, Denmark.

unvegetated systems have implied oxygen transfers that are 87% of those observed for vegetated systems. The Gersberg *et al.* (1986) data stand out as the most influenced by plants, with 45–75% of the transfer ascribable to plants, depending upon which plant is considered. These studies show a mean implied oxygen transfer of 6.8 gO/m$^2$·d, apportioned equally to BOD and ammonia reduction.

A widely quoted study of Brix (1990) produced field measurements of oxygen transfer in the reed bed at Kalø, Denmark, and sought to explain the fate of that oxygen via independent measurements. Figure 9.39 shows a reinterpretation of that work. Field and laboratory measurements showed that about twice as much oxygen was supplied physically as via plants, and that of the plant oxygen flux, almost none was excess reaching the water column. Brix (1990) considered that BOD was the only sink, but here it is presumed that nitrification of ammonia is also a factor. The oxygen equivalent of BOD is assumed to be 1.0 gO/gBOD, and 4.6 gO/gBOD for TKN.

Field measurements of methane emission suggested that nearly half of the BOD loss was via anoxic routes (Brix, 1990) rather than oxidation, and here methanogenesis is used to close the oxygen supply–demand balance. These studies were conducted in spring, and it is presumed that any nitrogen required for early plant growth came from translocation. This study demonstrated that net plant oxygen transfer (from the roots to the water column) is a minor contributor in HSSF wetlands, and that the implied oxygen requirement (based on classical nitrification and aerobic degradation of BOD) is likely to be an overestimate because of observed anaerobic BOD reduction processes. *The reinterpretation here suggests that ammonification of organic nitrogen, and the implied oxygen requirement for that extra ammonia, are of equal or greater importance than the apparent change in influent–effluent ammonia concentrations.*

Wu *et al.* (2001) measured oxygen fluxes in sealed HSSF wetland mesocosms, both vegetated and unvegetated. Results were consistent with the "layer" concept of Figure 9.40, meaning that ammonia diffusion to the top aerobic zone was suggested to be the controlling mechanism. Importantly, there was but slight increase in oxygen utilization by vegetated systems, of 0.45 gO/m$^2$·d, for both 10 and 50 mg/L of ammonia nitrogen. The utilization was 6.0 and 7.5 gO/m$^2$·d, respectively. Larger, field mesocosm results confirmed the ammonia removal rate to be strongly increasing with ammonia concentration.

## Reduction of Ammonia in VF Wetlands

Based on the information currently available, vertical flow wetlands are effective in oxidizing organic and ammonia-nitrogen. However, these removals are highly dependent on the mass loading of the wetland, and the operational regime (loading and resting periods) employed in the operation and

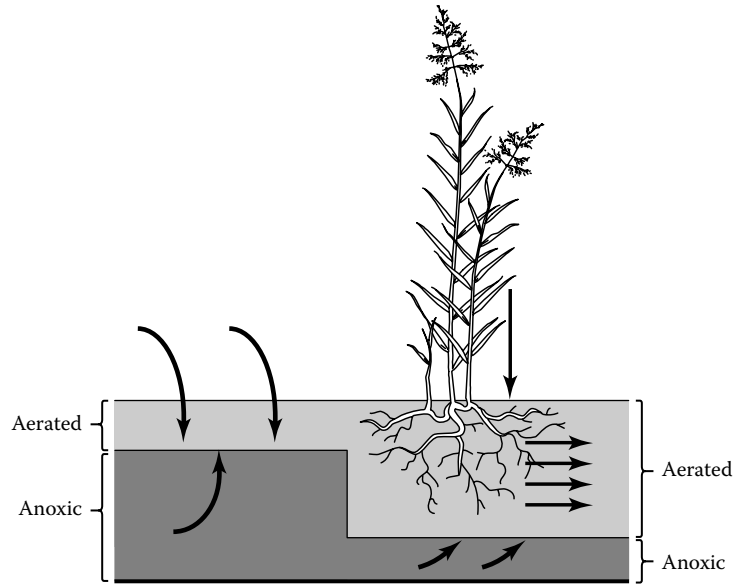



**FIGURE 9.40** Simplified model of movement of oxygen into the HSSF bed.

maintenance of the system. It is also apparent that the cation exchange capacity (CEC) of the bed materials can play a large role in retention of ammonia and subsequent processing within the VF wetland bed (Johns *et al*., 1998; Gisvold *et al*., 2000; Austin *et al*., 2006b).

### Implied Oxygen Supply in VF Wetlands

The TN entering vertical flow wetlands can be compared to the TN leaving the system. Presumably, the reduction can be contributed to the oxidation and reduction of nitrogen within the wetland. The different chemical stoichiometries currently known provide a bound on the maximum and minimum oxygen transfers that occur in pulse-loaded vertical flow beds (Table 9.28). It should be noted that pulse-loaded vertical flow wetlands appear to provide oxygen transfers considerably higher than FWS or HSSF wetlands (Tables 9.24 and 9.26).

Oxygen transfer is a key issue in the design of vertical flow wetland beds (Johansen and Brix, 1996; Cooper, 1999; Cooper *et al*., 1999), and is discussed in more detail in Chapters 20 and 21.

### Background Concentrations of Ammonia

There is not an apparent lower limit to exit ammonia concentrations in Figures 9.37 and 9.38, indicating a near-zero background for this species. This result has also been observed in natural treatment wetlands. For instance, transect water chemistry has been acquired at the Houghton Lake treatment wetland over a 28-year period, including measurements of ammonium nitrogen. These data show an exponential decrease in $NH_4$–N concentrations with distance from the discharge, thus supporting a first-order model. Summer season operation shows very low background concentrations (approximately 0.05 mg/L), both for the wetland prior to wastewater discharges and for the unaffected zones of the present-day wetland. Wastewater concentrations are much higher (approximately 10 mg/L), therefore it is accurate to represent the data with a first-order model with a zero background. Time series data for batch wetlands also support the near-zero background (Figure 9.41). The detection limit for ammonia is typically 0.05 mg/L, and thus the zero background usually represents some value less than the detection limit. The assumption of $C^* = 0$ mg/L appears to approximate operating data from all types of treatment wetlands.

### Rates and Rate Constants

In conventional activated sludge treatment system design, removal of ammonia is often modeled with a Monod formulation, with a half-saturation constant of 1.0 mg/L (U.S. EPA, 1993b):

$$J_{AN} = \frac{k' C_{AN}}{K + C_{AN}} \tag{9.49}$$

where

$C_{AN}$ = ammonia nitrogen concentration, mg/L (= g/m$^3$)
$J_{AN}$ = ammonia nitrogen removal flux, g/m$^2$·yr
$k'$ = Monod rate constant, m/yr
$K$ = half-saturation constant, mg/L

This formulation has been adopted in some analyses of ammonia in wetlands (Langergraber, 2001), but the half-saturation constant is so low that near-zero-order behavior should be seen for $C_{AN} >> 1$. Instead, for FWS wetlands, there is a linear correlation between the annual removal flux and the annual mean ammonia concentration for 98 wetlands

**TABLE 9.28**
**Oxygen Usage in VF Wetlands as a Function of Assumed Stoichiometry of BOD Reduction and Ammonia Removal**

| Percentile | BODLR[a] (gO/m²·d) | Internal ALR[b] (gN/m²·d) | External ALR[b] (gN/m²·d) | Maximum[c] O Usage (gO/m²·d) | Intermediate[c] O Usage (gO/m²·d) | Minimum[c] O Usage (gO/m²·d) |
|---|---|---|---|---|---|---|
| 0.05 | 2.4 | –0.2 | 0.1 | 14.2 | 7.6 | 1.0 |
| 0.10 | 2.6 | –0.1 | 0.2 | 19.5 | 10.4 | 1.4 |
| 0.20 | 3.6 | 0.2 | 0.6 | 20.7 | 11.5 | 2.0 |
| 0.30 | 6.5 | 0.3 | 0.8 | 22.6 | 12.4 | 2.3 |
| 0.40 | 8.4 | 0.4 | 1.0 | 22.9 | 12.6 | 2.6 |
| 0.50 | 9.4 | 0.5 | 1.3 | 24.7 | 13.4 | 3.5 |
| 0.60 | 10.4 | 0.6 | 1.9 | 29.3 | 14.7 | 4.1 |
| 0.70 | 11.3 | 0.6 | 2.0 | 30.7 | 16.5 | 5.1 |
| 0.80 | 12.7 | 0.8 | 2.7 | 39.9 | 20.0 | 9.1 |
| 0.90 | 17.7 | 1.3 | 8.4 | 51.2 | 21.9 | 14.1 |
| 0.95 | 19.5 | 1.8 | 10.1 | 55.6 | 27.4 | 16.8 |

*Note:* Both internal (from organic N) and external ammonia loads are considered. The Anammox route requires half the ammonia to be converted to nitrite, which needs 3.43 gO/gN. BOD may be reduced by anaerobic or aerobic processes. The data represent 22 wetlands and 34 wetland-years of data.

[a] BODLR = BOD load removed.

[b] ALR = ammonia load removed.

[c] The maximum case assumes 1.5 gO/gBOD and 4.6 gO/g$NH_4$–N.
The intermediate case assumes 1.0 gO/gBOD and 1.7 gO/g$NH_4$–N.
The minimum case assumes 0.0 gO/gBOD and 1.7 gO/g$NH_4$–N.

for $C_{AN} > 1$, with $R^2 = 0.61$ and a $k$-value of $k_{AN} = 9.2$ m/yr. This result should not be surprising, because it has been noted that both ammonia volatilization and plant uptake may be involved in wetland ammonia removal, as well as microbial processes. Therefore, the area-based first-order expression is preferred to model the disappearance of ammonia nitrogen (nitrification and other processes combined) in wetland treatment systems:

$$J_{AN} = k_{AN} C_{AN} \tag{9.50}$$

where

$k_{AN}$ = removal rate constant for ammonia N, m/yr

## Ammonia Rate Constants for FWS Wetlands

Results across systems for the value $P = 3$ are given here. The value $C^* = 0.0$ mg/L is used, and the remaining model parameter is the $k$-value, selected to fit the model:

$$\frac{C_{AN,\,out}}{C_{AN,\,in}} = \left(1 + \frac{k_{AN}}{3q}\right)^{-3} \tag{9.51}$$

Parameter estimation is not reliable for low inlet concentrations, and those wetlands with $C_{AN,in} < 1$ mg/L have been excluded from calibration. Out of 147 wetlands with data for all species of nitrogen (see Figure 9.37), 118 met this criterion.

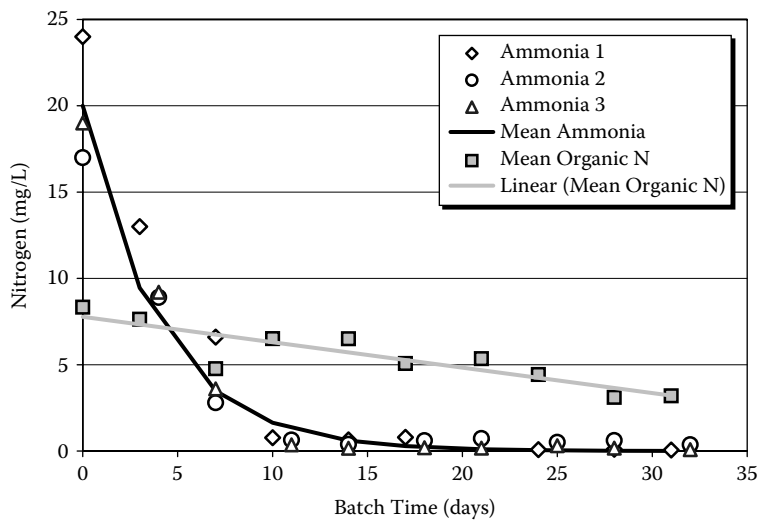



**FIGURE 9.41** Reduction of ammonia nitrogen in the batch treatment FWS wetland systems at Humboldt, Saskatchewan. (Data from Lakhsman (1981) *A demonstration project at Humboldt to provide tertiary treatment to the municipal effluent using aquatic plants*. SRC Publication No. E-820-4-E-81. 74 pp. Saskatchewan Research Council.)

Calibration included ammonification (production) and nitrification (destruction), as well as return of organic nitrogen from the decomposition of biomass. The median annual rate constant was $k_{AN}$ = 14.7 m/yr (Table 9.23). The 10th–90th percentile range is 4.7–85.6 m/yr. There is a significant temperature dependence of ammonia *k*-values. Even on an average annual basis, temperature or season may be an important determinant of the rate constant, and these factors are thus responsible for some of the intersystem variability in annual *k*-values. Accordingly, it is necessary to examine intra-annual effects.

### *Microbially Dominated Wetlands*

Ammonia nitrogen is a nutrient source to support growth in the wetland. As for TKN and TN, when the ammonia loading to the wetland exceeds the growth requirements of the plants, bacteria, and algae by a considerable margin, the removal of ammonia is very likely to be microbially mediated. The loading limit for bacteria to predominate is approximately 120 gN/m$^2$·yr (Kadlec, 2005d). That means that slightly under half (47%) of the 118 FWS wetlands qualified as microbially controlled. In this ammonia data set, some wetlands are presumed to derive growth nitrogen from nitrate as well as ammonia.

In those cases where there is a significant contribution of ammonification to the ammonia loading, a sequential correction has been made to the kinetic scheme. However, there are many situations in which there is little ammonification, because incoming and outgoing organic nitrogen are low compared to ammonia. In the latter cases, the concentrations of ammonia alone suffice to estimate rate constants.

Results of calibration of $k_{AN}$-values for entire periods of record for representative wetlands are summarized in Table 9.29 Monthly averages were used to avoid synoptic error (transit time offset). Calibrations were performed for best estimates of the internal hydraulics for each wetland. Therefore, *P*-values range from 2 (New Hanover, measured $P = N = 2$) to near plug flow conditions, based upon system geometry. The median $k_{20}$-value for ammonia is 14.2 m/yr, but the range is wide.

There are strong seasonal effects on the ammonia rate constant, which for microbially mediated systems is in synchrony with the water temperature. A temperature coefficient ($\theta$), is capable of accounting for these effects (see Chapter 6). Temperature coefficients had a median value of 1.049, indicating a relatively strong thermal effect on the suite of microbial processes that contribute to ammonia reduction.

Some of the effects of other environmental factors may be sought from the data in Table 9.29. The maximum implied oxygen requirement for complete nitrification has been calculated as 4.6 times the ammonia load removal (theoretical oxygen). This theoretical amount should be supplied by incoming DO together with reaeration. The Musselwhite system stands out as an anomaly, because the ammonia disappearance by traditional nitrification (19.7 gO/m$^2$·d) requires too much oxygen (see Chapter 5). Accordingly, Bishay and Kadlec (2005) have speculated on other mechanisms of ammonia removal in that wetland. Wetlands receiving very high ammonia also would require considerable theoretical oxygen (4–6 gO/m$^2$·d), which is at the upper, doubtful end of the supply range. Those wetlands may be losing small amounts of ammonia to volatilization (see Poach *et al*., 2002, 2004, and the discussion earlier in this chapter).

### *Open Water Zones*

U.S. EPA (2000a) speculates that oxygenation is better provided by zones of open water, containing submerged vegetation if possible. Such zones are then incorporated as a necessary part of the wetland system design. We here observe that such reasoning would apply only to those FWS wetlands (about half) that are under microbial control of ammonia removal. In Table 9.29, there are five wetlands that had large fractions of open water (Pontotoc (2), Brawley, Oxelösund, Hassleholm). These display a wide range of $k_{AN20}$ values, from quite low (6.9 m/yr) to quite high (140 m/yr). Consequently, there is nothing in this detailed data analysis to support the open water advantage concept. Kadlec (2005e) has drawn a similar conclusion for a wider set of FWS treatment wetlands. Open water zones are likely to foster a greater component of algal uptake for agronomic (lightly loaded) wetlands, in which ammonia removal is controlled by vegetation and algae.

Both algae and macrophytes are effective nitrogen cyclers, with the former dominating lagoon treatment systems, and the latter dominating FWS wetlands. Ammonia areal loadings to ponds are typically in the range defined here for microbial control. Algal biomass densities are typically lower than those for emergent macrophytes, but turnover times are much faster. A source of information on ammonia loss rates may be found in the literature on facultative ponds. For instance, Pano and Middlebrooks (1982) analyzed data from several ponds, and calibrated a first-order areal, well-mixed model to some of the data; and verified the model with the remainder of the data. The precise mechanism for ammonium reduction was not studied, but the model was chosen based on previous modeling of ammonia stripping ponds. Predictions were for $k_{AN20}$ are 5–13 m/yr, for pH in the range of 7.0–7.5, which is the typical range for FWS treatment wetlands. The temperature coefficients derived from the Pano and Middlebrooks (1982) model are in the range $\theta$ = 1.045–1.069, for pH in the range of 7.0–7.5.

The Pano and Middlebrooks (1982) model remains as the benchmark reference at the present time (Abis, 2002; Mara, 2003). The same model was adopted by Soares *et al*. (1996), who found $k_{AN}$ = 14 m/yr at 21°C, and 20 m/yr at 22°C, both for pH = 7.5. Zimmo *et al*. (2004) measured the individual processes in ponds, and found sedimentation (most), microbial (some), and volatilization (least) losses all played a part in removal. Azov and Tregubova (1995) found evidence for major amounts of sequential nitritation and nitrification, but little denitrification. Therefore, pond data probably represent mixed control mechanisms.

Because these pond areal $k_{AN}$ values correspond to the central tendency of the FWS wetland values, there appears

## TABLE 9.29
## Dependence of Ammonia Rate Constants on Temperature for FWS Systems

| Name | Location | Cells | Influent Type | Years | HLR In (cm/d) | $NH_4$–N In (mg/L) | $NH_4$–N Out (mg/L) | ALI[a] ($g/m^2 \cdot yr$) | *P* (TIS) | Temp (°C) | $k_{AN20}$ (m/yr) | Theta | Theoretical Oxygen ($g/m^2 \cdot d$) | Vegetation |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Brawley | California | All | River | 4 | 8.9 | 4.1 | 0.1 | 131 | ∞ | 21.0 | 140.0 | 1.072 | 1.59 | Open |
| Hassleholm | Sweden | All | Secondary | 5 | 4.8 | 7.2 | 4.6 | 126 | 3 | 6.0 | 43.3 | 1.069 | 0.56 | Open |
| Columbia | Missouri | All | Secondary | 3 | 13.4 | 7.8 | 5.9 | 373 | 3 | 14.5 | 10.0 | 1.044 | 1.14 | — |
| Brighton | Ontario | All | Lagoon | 4 | 5.0 | 11.0 | 8.8 | 190 | 3 | 9.4 | 12.5 | 1.053 | 0.48 | — |
| Musselwhite | Ontario | All | Mine | 4 | 51.8 | 12.9 | 3.5 | 2,144 | ∞ | 11.0 | 245.0 | 1.033 | 19.70 | — |
| Oxelösund | Sweden | All | Secondary | 5 | 2.1 | 18.0 | 11.9 | 140 | 3 | 6.0 | 8.1 | 1.056 | 0.59 | Open |
| Sacramento | California | Cell 7 | Secondary | 2 | 6.2 | 18.2 | 10.3 | 411 | ∞ | 15.9 | 9.2 | 1.109 | 2.25 | — |
| Warangal | India | All | Primary | 2 | 4.3 | 19.9 | 3.6 | 311 | 3 | 23.1 | 66.0 | 1.027 | 3.20 | — |
| Linköping | Sweden | All | Secondary | 3 | 4.2 | 26.5 | 9.1 | 255 | 3 | 8.6 | 17.8 | 1.032 | 2.12 | — |
| Saginaw | Michigan | All | Leachate | 2 | 1.6 | 29.5 | 5.0 | 183 | 3 | 19.3 | 28.5 | 1.045 | 1.92 | — |
| Pontotoc | Mississippi | 2 | Animal | 2 | 1.5 | 112.2 | 38.9 | 630 | 3 | 20.11 | 6.9 | 1.018 | 5.18 | Open |
| Pontotoc | Mississippi | 1 | Animal | 2 | 1.3 | 112.7 | 35.8 | 540 | 3 | 18.12 | 6.8 | 1.005 | 4.65 | Open |
| New Hanover | North Carolina | All | Leachate | 4 | 1.5 | 117.4 | 56.9 | 639 | 2 | 19.4 | 4.9 | 1.090 | 4.15 | — |
| Duplin County | North Carolina | All | Animal | 7 | 1.1 | 120.2 | 20.8 | 538 | 3 | 14.0 | 15.9 | 1.063 | 5.60 | — |
| **Median** | | | | | | | | **342** | | | **14.2** | **1.049** | | |

[a] ALI = influent ammonia loading rate

**TABLE 9.30**
**Annual Rate Constants for Microbial FWS Wetlands with Appreciable Components of Submerged Aquatic Vegetation**

| Name | Location | Vegetation | Nonemergent Coverage (%) | T (°C) | TN In (mg/L) | TN Out (mg/L) | ALI[a] (g/m²·yr) | *P*TIS | $k_{AN}$ (m/yr) |
|---|---|---|---|---|---|---|---|---|---|
| Snogeröd | Sweden | *Elodea canadensis* | 100 | 8 | 9.8 | 7.5 | 260[b] | ∞ | 52 |
| Durham | New Hampshire | *Elodea nuttallii* | 100 | 17 | 20 | 11 | 220[b] | 1 | 110 |
| Texel | The Netherlands | *Elodea nuttallii* *Ceratophyllum demersum* | 50 | 10 | 1.73 | 1.35 | 490 | 12 | 71 |
| Texel | The Netherlands | *Elodea nuttallii* *Ceratophyllum demersum* | 50 | 10 | 1.73 | 1.54 | 450 | 12 | 42 |
| Arcata | California | *Potamogeton pectinatus* | 73 | 13 | 7.4 | 6.3 | 313 | 3 | 7 |
| Arcata | California | *Potamogeton pectinatus* | 83 | 13 | 6.3 | 5 | 349 | 3 | 13 |
| Arcata | California | *Potamogeton pectinatus* | 78 | 13 | 5 | 4.1 | 162 | 3 | 6 |

[a] ALI = influent ammonia loading rate.
[b] Total nitrogen loading.

to be neither an advantage nor a disadvantage to unvegetated open water zones in wetlands designed for microbial ammonia reduction. However, submerged aquatic vegetation (SAV) zones may provide ammonia removal benefit, presumably due to subsurface oxygenation by these submerged photosynthesizers. On an annual basis, system data provided higher than the global median microbial ammonia rate constants in New Hampshire (Bishop and Eighmy, 1989), Sweden (Gumbricht, 1993a), and the Netherlands (Toet, 2003); but less than the than the global median microbial ammonia rate constants at Arcata, California (U.S. EPA, 1999) (Table 9.30). Effective strategies for maintaining SAV communities remain elusive, because they may easily be shaded out by floating plants (*Lemna*), overrun by aggressive semifloating species (*Hydrilla*), damaged by dry-out or extreme wind (hurricanes), or killed by high turbidity.

### *Agronomic Wetlands (Lightly Loaded)*

When the ammonia loading to the wetland is less than the growth requirements of the plants and algae by a considerable margin, the removal of ammonia is very likely to be mediated by the growth and decay of biomass, including plants and algae. As stated above, a rough guideline is TKN loading less than approximately 120 gN/m²·yr (Kadlec, 2005d). If it is presumed that ammonia is the preferred form of nutrient nitrogen, this criterion may also be applied for ammonia.

Plant uptake rates do not correspond to the annual cycle of water temperatures, and hence ammonia removal in agronomic wetlands cannot be characterized by modified Arrhenius $\theta$-factors. The contrast between water temperature sequences and ammonia rate constants is illustrated for Estevan, Saskatchewan, in Figure 9.42. The ammonia loading to this wetland was 28 g/m²·yr, average during the growing season. No correspondence between monthly rate constants and temperature is present. For this agronomic wetland, it would be appropriate to utilize monthly *k*-values:

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| $k_{TN,j}$ | = 17.1 | 12.5 | 11.7 | 9.0 | 12.0 | 14.6 | 9.1 m/yr |
| $j$ | = 5 | 6 | 7 | 8 | 9 | 10 | 11 |

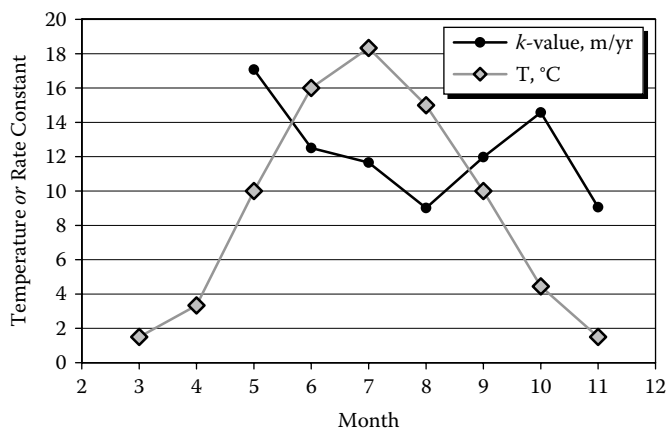



**FIGURE 9.42** Seasonal ammonia rate constants for the Estevan, Saskatchewan, FWS treatment wetland. This system was operated seasonally (May–November) for nine years.

**TABLE 9.31**
**Partitioning of Annual Ammonia Removal Rate Constants in FWS Wetlands According to the Control Mechanism**

| | Agronomic Wetlands (N = 73) | | Microbial Wetlands (N = 74) | |
|---|---|---|---|---|
| Percentile | Inlet Load ($g/m^2 \cdot yr$) | Rate Coefficient, $k_{AN}$ (m/yr) | Inlet Load ($g/m^2 \cdot yr$) | Rate Coefficient, $k_{AN}$ (m/yr) |
| 0.0 | 0.3 | 0.4 | 123 | 3.5 |
| 0.1 | 1.9 | 10 | 170 | 4.6 |
| 0.2 | 3.6 | 29 | 259 | 5.7 |
| 0.3 | 7.3 | 45 | 367 | 7.4 |
| 0.4 | 15.4 | 52 | 465 | 9.1 |
| 0.5 | 25.2 | 67 | 920 | 11.0 |
| 0.6 | 38.8 | 87 | 1,648 | 13.4 |
| 0.7 | 40.2 | 120 | 1,984 | 16.1 |
| 0.8 | 70.0 | 197 | 2,896 | 27.8 |
| 0.9 | 81.6 | 319 | 5,791 | 74.2 |
| 1.0 | 108.0 | 1,230 | 11,583 | 429 |

Similar patterns occur for other agronomic systems, such as Listowel, Ontario, 1, 2, and 3; and Arcata, California, enhancement wetlands.

It is noteworthy that first-order areal ammonia removal rate constants are higher for agronomic systems than for microbial systems. The median annual $k_{AN}$ for agronomic wetlands is 67 m/yr ($N = 73$), while for microbial wetlands it is 11 m/yr ($N = 74$). Stated another way, 80% of agronomic wetlands have annual $k_{AN} > 30$ m/yr, whereas 80% of microbial wetlands have annual $k_{AN} < 30$ m/yr (Table 9.31). Therefore, the designer must be cognizant of the potential for a change in the type of ammonia removal as prospective loadings are varied, and the system shifts from plant-dominated (agronomic) to microbially dominated.

It is also important to recognize that agronomic control does not mean that the ultimate removal of ammonia is to the biomass compartment of the wetland. It has been estimated that 90% of uptake is recycled to the water column, presumably during periods of warmer temperatures. It may be speculated that the return of nitrogen occurs from solids (phytomass, necromass) that are also the sites of microbial attachment. Thus, ammonification and nitrification are presumed to be occurring at exactly the locations where TKN is generated, and no mass transfer step is involved to slow the processes. Further, agronomic ammonia application rates do not imply large oxygen transfer rates, and atmospheric reaeration is probably adequate in almost all cases. In their review of nitrogen transformations in flooded soils, Reddy and Patrick (1984) summarized estimated first-order nitrification measurements from soil studies. The first-order, volume-based nitrification rates they summarized ranged from 0.003 to 3.1 $d^{-1}$, with a mean of 0.29 $d^{-1}$. Assuming an effective nitrification depth of 30 cm in a typical wetland treatment system, these values are equivalent to area-based first-order nitrification rate constants between 0.33 and 339 m/yr with an average of 32 m/yr. It is likely that these values represent typical nitrification rate constants in systems that are not limited by a shortage of oxygen.

### *Depth Effects*

Ammonia removal is controlled by processes that are apportioned to FWS wetland surface area. Plant biomass is an areal parameter of the wetland, for both floating and rooted species, and hence cycling is area-dependent. Volatilization, if any, is water surface-specific. Microbial attachment sites are associated with root mats, sediment–water interfaces, and litter layers. They are also associated with plant stems and leaves, which may be distributed throughout the water column. It is therefore expected that increasing water depths will not provide proportionally more removal activity.

Figure 9.43 illustrates this effect for the side-by-side tests at Arcata, California (Gearheart *et al*., 1983). Values of $k_{V20}$ are inversely proportional to depth, meaning that doubling the depth halves the $k_V$-value. Then according to the definitions, the areal rate constant does not depend on depth. If the volumetric model is utilized in FWS calculations, there appears to be the option of increasing performance by increasing the water depth, and hence increasing the nominal detention time. However, that advantage is lost if the volumetric rate "constant" decreases with increasing depth, as indicated in Figure 9.43.

## Ammonia Rate Constants for HSSF Wetlands

It should be noted that organic nitrogen is an important source of ammonia due to mineralization (ammonification). The HSSF wetland is typically an oxygen-limited system, and ammonification of organic nitrogen can lead to increases in ammonia concentrations within the wetland bed (Figure 9.44).

HSSF wetlands are most commonly used for secondary treatment, with a primary treatment device such as an Imhoff tank (Imhoff and Fair, 1929) or a septic tank (U.S. EPA, 1980) upstream of the wetland. HSSF wetlands are typically chosen because (if properly designed and operated), wastewater is not exposed during the treatment process, minimizing potential public health issues. Devices such as

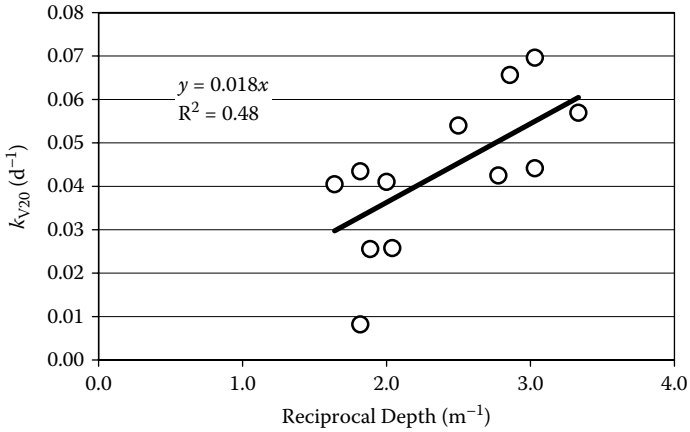



**FIGURE 9.43** The dependence of volumetric ammonia rate "constants" on reciprocal depth for the Arcata, California, pilot FWS wetlands. Deeper water results in lower volumetric rate constants. (Data from Gearheart *et al*. (1983) *City of Arcata Marsh Pilot Project, effluent quality results—system design and management*. Final Report to the North Coast Regional Water Quality Board and State Water Resources Board.)

Imhoff tanks or septic tanks have anaerobic environments. In these primary treatment devices, there is usually a partial conversion of organic nitrogen to ammonia (ammonification) plus the ammonia originally present in the wastewater influent. As a result, HSSF wetlands that are designed for secondary treatment typically receive their influent nitrogen in the form of ammonia and organic nitrogen. Although the organic nitrogen fraction cannot be ignored in these cases, ammonification often does not interfere with a monotonic decreasing trend in ammonia along the flow direction (Figure 9.45).

Most HSSF wetlands are dominated by microbial processing of ammonia (Figure 9.38), and the various alternative mechanisms typically require oxygen. In passive, steady flow HSSF wetlands with no mechanical aeration, the required oxygen supply is from the air, because wetland influents typical are devoid of dissolved oxygen. The routes by which oxygen can gain entry to the water column are: (1) atmospheric diffusion and (2) plant oxygen flows (see Figure 9.40).

In the former case, oxygen diffuses from air into the water in the top bed layer. Ammonia must diffuse upward into this layer so that microbes can have both ingredients to perform nitrification. In the latter case, the oxygenated zone consists of micro zones near roots, with oxygen supplied through the plant roots. The amount of this plant aeration flux is contingent upon the root density and volume, as well as plant physiology, which dictates how much oxygen can be delivered to protect the roots from anaerobiasis. If the amount of ammonia arriving is not overwhelming, both mechanisms offer the ability to supply nitrifiers and other N-processing bacteria.

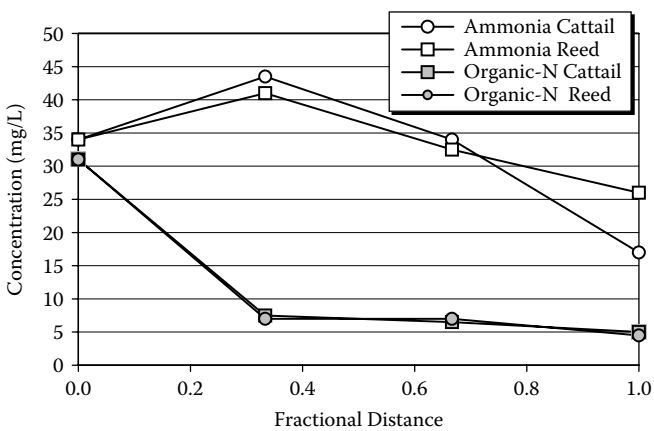



**FIGURE 9.44** Profiles of ammonia and organic nitrogen through the HSSF wetlands at Thrace, Greece. (Data from Akratos and Tsihrintzis (2007) *Ecological Engineering*, 29(2): 173–191.)

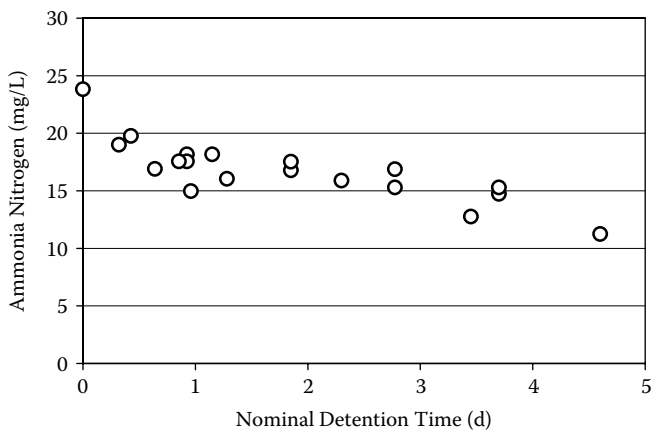



**FIGURE 9.45** Decrease of ammonia along profiles through the three cells of the Minoa, New York, HSSF wetland on February 15, 1996. The cells had different hydraulic loadings, and measurements were taken at the quarter points of the flow path. (Data from Theis and Young (2000) *Subsurface flow wetland for wastewater treatment at Minoa*. Final Report to the New York State Energy Research and Development Authority, Albany, New York.)

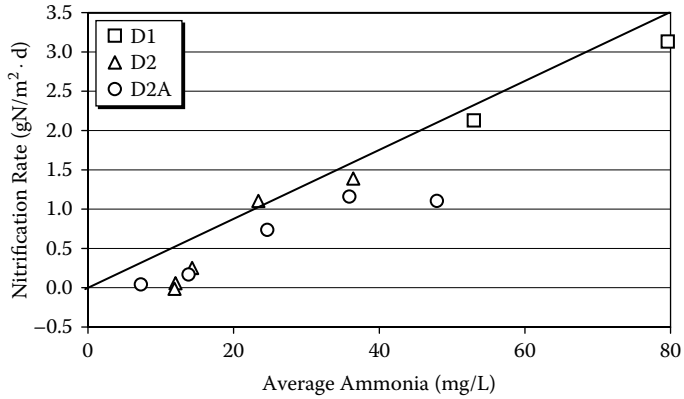



**FIGURE 9.46** Dependence of nitrification rate on ammonia concentration for dairy effluents for HSSF wetlands in New Zealand. The range of inlet ammonia is restricted to $C_i < 80$ mg/L. For higher inlet concentrations, the apparent proportionality does not hold. (Data from Tanner *et al*. (2002a) *Ecological Engineering*, 18(4): 499–520.)

The usual model for the ammonia diffusion is a mass transfer coefficient times a concentration driving force (see Chapter 6), and thus an ammonia concentration effect is anticipated. Accordingly, a first-order areal model is expected to be operative. Rate constants are expected to be mass transfer coefficients, and thus depend upon water velocity, but only mildly upon temperature. There is ample precedent for mass transfer control of gas–liquid reactions in more controlled environments (Danckwerts, 1970).

A test of the idea of first-order removal is available from the data of Tanner *et al*. (2002a). Wetlands were operated in sequence, with each train of cells receiving a different strength of dairy wastewater. As a check of the first-order idea, Equation 9.50 is used as the basis for a plot of data. The rate of nitrification is found to be linear in the ammonia concentration for these wetland mesocosms (Figure 9.46, $k$ = 13 m/yr, $R^2$ = 0.89). However, as the ammonia concentration extends to high values, in excess of 100 mg/L, the first-order concept no longer is evidenced by the data. At higher concentrations, the oxygen supply rate becomes limiting, rather than ammonia mass transfer to the reaction zone.

It is possible that, under conditions of heavy loading, the ability of the plants to defend against anoxia at their roots can be exceeded by the ammonia and BOD supply. Under that condition, plants may reduce their active root mass so that they can maintain oxic conditions near those that remain. Likewise, physical aeration may be limited by the air-side mass transfer process, meaning that all the ammonia diffusing to the surface will not be consumed.

Confirmation of these concepts is found in the data from several studies that have focused upon vertical profiles of redox, ammonia, and oxidation products (oxidized nitrogen species). The top of the bed is typically at higher redox potential, contains less ammonia, and contains more oxidized nitrogen (Figure 9.47). Similar results were found at Minoa, New York (Theis and Young, 2000), Richmond, New South Wales (Bavor *et al*., 1988), and West Harwood, Scotland (McGechan *et al*., 2005b).

Ranges in annual average $k$-rates for HSSF wetlands are summarized in Table 9.25, with the assumptions of $P$TIS = 6 and $C^*$ = 0 mg/L. The median $k$-rate is 11.4 m/yr. However, there is wide variability in the data, and the 10th–90th percentile range is 0.4–63.3 m/yr.

Temperature dependence ($\theta$) is summarized based on the period of record for 18 HSSF wetlands in Table 9.32. The overall data set shows little dependence on temperature (median $\theta$ = 1.014). However, the reader is cautioned that the range in $\theta$-values is wide, and that individual systems demonstrated to have a strong dependence on temperature. As a result, use of a median $\theta$-factor may not be appropriate if regulatory compliance for ammonia is a project objective. In Table 9.32, the 10th–90th percentile range is 0.976–1.082.

### *Effect of Media Size on Ammonia Removal*

Based on the data that is currently available, it appears that ammonia removal is reduced as the size of the bed media increases. There are several factors that could contribute to these observations: (1) coarse bed media has less surface area (per unit volume) than finer materials, and hence less opportunity for ammonia-oxidizing biofilms; and (2) plant root growth and root penetration are inhibited in coarse bed materials (Greenway, 2002). Effect of media size on ammonia removal rates is summarized in Table 9.33.

### *Mulch Effects*

In cold-climate applications, mulch is often used to insulate HSSF wetlands (Henneck *et al*., 2001; Wallace *et al*., 2001) as a means to close the energy balance without freezing the wetland (Kadlec *et al*., 2003; Wallace and Nivala, 2005). The mulch layer reduces atmospheric exchanges, including the diffusion of oxygen.

This mulch material can degrade and exert an additional nitrogen loading on the system (Wallace *et al*., 2001). Organic nitrogen leached from the mulch layer undergoes ammonification in the water column, imposing an additional ammonia loading on the system. The results of this mulch

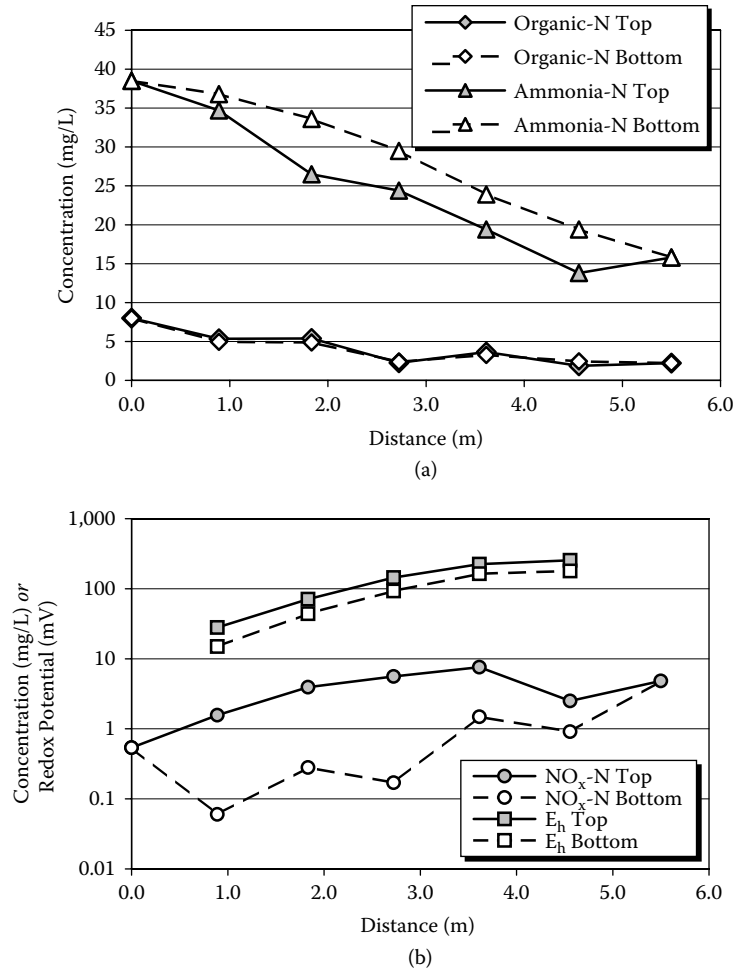



**FIGURE 9.47** Longitudinal profiles in the Lismore, Australia, HSSF wetland. Samples were taken at the top and bottom of the gravel bed. (Data from Bayley *et al.* (2003) *Water Science Technology*, 48(5): 175–182.)

loading are exacerbated under situations where the mulch is a poorly decomposed material (Wallace and Knight, 2006) and the wetland experiences low hydraulic loadings. Under these conditions, the system is unlikely to respond with a *k*-rate as good as the values summarized in Table 9.25, at least for the first few years of operation, based on 21 system-years of start-up data (Wallace *et al*., 2001).

### Intrasystem Variability

Ammonia nitrogen is very frequently the target of regulatory limits. Those limits properly seek to avoid the consequences of eutrophication and possible aquatic toxicity caused by ammonia, and may be imposed at varying frequencies of measurement. Wetland design must account for both the seasonal variability in the ammonia trends, and the stochastic variation that is superimposed upon it. The general concepts have been discussed in the section on TN, and are embodied in Equations 9.40 and 9.41. Seasonal trends may or may not follow water temperatures, since agronomic (lightly loaded) systems will be heavily influenced by plant biomass cycling.

### Variability in FWS Wetlands

An example of such trends and variability is given in Figure 9.48 for ammonia nitrogen at the Columbia, Missouri, FWS wetland facility. The measurement frequency was daily on weekdays. There is a pronounced annual trend, which is well represented by a cosine function. A good deterministic model would also reproduce this trend. The highest ammonia trend concentrations occur in the winter, and this cold period therefore would control design sizing, although ammonia was a design criterion for Columbia. There is a large amount of scatter of the daily concentrations about the trend line, with values both above and below the trend, as there should be for any good data fitting procedure. These daily excursions are not in general excused from regulation. The distribution of fractional departures from the trend is nearly a normal distribution for Columbia (Figure 9.48). The high concentration end of the distribution is of regulatory interest, and it is therefore useful to look at the high percentile points. The 80th, 90th, and 95th percentiles are at daily fractional excursions of 0.23, 0.33, and 0.72 of the trend, respectively.

**TABLE 9.32**
**Temperature Coefficients for Nitrification Rate Constants in HSSF Wetlands**

| Site | Reference | Cell | T Range (°C) | Mean HLR (cm/d) | Mean $C_i$ (mg/L) | Mean $C_o$ (mg/L) | Theta |
|---|---|---|---|---|---|---|---|
| Richmond, NSW | Bavor *et al.* (1988) | Gravel | 11–24 | 3.8 | 35.19 | 18.78 | 0.997 |
| Richmond, NSW | Bavor *et al.* (1988) | *Typha* | 11–24 | 4.6 | 35.19 | 18.80 | 0.998 |
| Richmond, NSW | Bavor *et al.* (1988) | *Schoenoplectus* | 11–24 | 5.1 | 35.19 | 19.43 | 1.007 |
| Richmond, NSW | Bavor *et al.* (1988) | Mixed A | 11–24 | 4.6 | 35.19 | 15.29 | 1.016 |
| Richmond, NSW | Bavor *et al.* (1988) | Mixed B | 11–24 | 3.8 | 35.19 | 12.64 | 1.012 |
| Grand Lake, Minnesota | Unpublished | 1 | 1–17 | 1.0 | 50.7 | 40.2 | 1.074 |
| NERCC, Minnesota | Unpublished | 1 | 1–16 | 1.4 | 78.2 | 55.3 | 1.017 |
| NERCC, Minnesota | Unpublished | 2 | 1–16 | 1.4 | 78.2 | 59.3 | 1.023 |
| Lincoln, Nebraska | Vanier and Dahab (1997) | *Typha, Schoenoplectus* | 4–21 | 9.5 | 20.4 | 15.7 | 1.057 |
| Middleton, United Kingdom | CWA (2006) | 1 | 4–17 | 15.6 | 2.70 | 0.64 | 1.002 |
| West Harwood, Scotland | McGechan *et al.* (2005b) | 1 | 5–17 | 5.2 | — | — | 1.032 |
| Lake Capri, Missouri | Regmi *et al.* (2003) | Non-vegetated | 2–24 | 2.3 | 18.9 | 17.0 | 1.100 |
| Lake Capri, Missouri | Regmi *et al.* (2003) | Vegetated | 2–24 | 2.3 | 18.9 | 10.4 | 1.110 |
| Hamilton, New Zealand | Tanner *et al.* (1998b) | L1 | 10–25 | 1.5 | 29.19 | 18.6 | 1.062 |
| Hamilton, New Zealand | Tanner *et al.* (1998b) | L2 | 10–25 | 2.5 | 29.19 | 15.9 | 0.976 |
| Hamilton, New Zealand | Tanner *et al.* (1998b) | L3 | 10–25 | 3.3 | 29.19 | 19.1 | 0.983 |
| Hamilton, New Zealand | Tanner *et al.* (1998b) | L4 | 10–25 | 4.9 | 29.19 | 21.2 | 0.975 |
| Hamilton, New Zealand | Tanner *et al.* (1998b) | L5 | 10–25 | 6.9 | 29.19 | 22.3 | 0.911 |

| Percentile | Theta |
|---|---|
| 0.05 | 0.965 |
| 0.10 | 0.976 |
| 0.20 | 0.989 |
| 0.30 | 0.998 |
| 0.40 | 1.006 |
| 0.50 | 1.014 |
| 0.60 | 1.018 |
| 0.70 | 1.031 |
| 0.80 | 1.060 |
| 0.90 | 1.082 |
| 0.95 | 1.102 |

Monthly limits are a quite common averaging period. Therefore, it is useful to examine the variability of monthly average outlet concentrations, represented by a fractional addition to the trend value ($\Psi$), which corresponds to a multiplier on the trend value of $(1 + \Psi)$. Table 9.34 lists percentiles of the monthly ammonia $\Psi$-distributions for some representative FWS wetlands. These data indicate that the median of the 95th percentile is an additional 110% above the trend, and that the median of the 90th percentile is an additional 87%. To incorporate this variability into design, the wetland designer must oversize the wetland, according to the steps outlined in the section on TN. Because deterministic equations ($k$-rate or other) represent the central estimate of performance, failure to incorporate variability means that 50% of the anticipated concentrations will be above the calculated values. If the design calculation is set at the regulatory limit, exceedance frequencies of 50% are expected to result. The risk of exceedance is lowered by increasing wetland size to the selected percentile of the $\Psi$-distribution.

**TABLE 9.33**
**Effect of Gravel Size on Nitrogen Rate Constants for HSSF Wetlands**

| Source | Parameter | $k_A$ (m/yr) Coarse | $k_A$ (m/yr) Fine |
|---|---|---|---|
| García *et al*. (2003b) (deep) | Ammonia | 3.4 | 3.9 |
| García *et al*. (2003b) (shallow) | Ammonia | 5.5 | 8.4 |
| Akratos and Tsihrintzis (2007) | TKN | 5.1 | 6.7 |
| Akratos and Tsihrintzis (2007) | Ammonia | 4.9 | 6.3 |
| Akratos and Tsihrintzis (2007) | Nitrification | 9.3 | 12.3 |

## Variability in HSSF Wetlands

Similar to FWS wetlands, HSSF wetlands display seasonal trends. Although the central tendency of the available data set indicates minimal dependence on water temperature (Table 9.32), some HSSF wetlands are clearly temperature-dependent for reasons that are not fully understood. It is also reasonable to expect that HSSF wetlands will exhibit

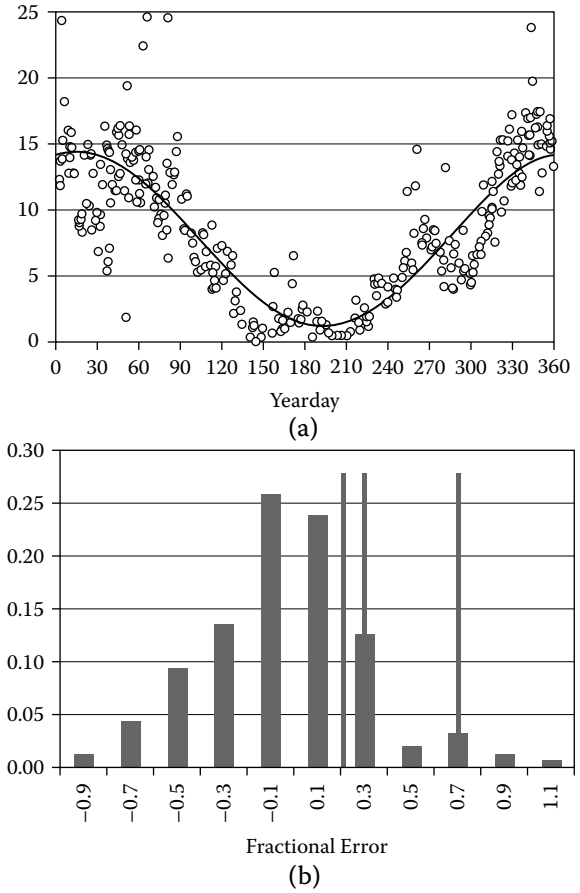



**FIGURE 9.48** Distribution of daily ammonia concentrations leaving the Columbia, Missouri, FWS treatment wetland over a 20-month period (upper panel). A seasonal cosine trend is apparent, which forms the basis for computing fractional errors for each point. The distribution of errors is nearly normal (lower panel). Lines represent the 80th, 90th, and 95th percentiles at 0.23, 0.33, and 0.72, respectively.

seasonal differences due to changes in flow and loading, as well as the impact of plant biomass cycling of nitrogen (Figure 9.49).

Table 9.35 lists percentiles of the monthly ammonia Ψ-distributions for some representative HSSF wetlands. These data indicate that the median of the 95th percentile is an additional 195% above the trend, and that the median of the 90th percentile is an additional 176%. To incorporate this variability into design, the wetland designer must oversize the wetland, according to the steps outlined in the section on TN. Because deterministic equations (*k*-rate or other) represent the central estimate of performance, failure to incorporate variability means that 50% of the anticipated concentrations will be above the calculated values. If the design calculation is set at the regulatory limit, exceedance frequencies of 50% are expected to result. The risk of exceedance is lowered by increasing wetland size to the selected percentile of the Ψ-distribution.

**TABLE 9.34**
**Trend Multipliers for Effluent Ammonia Nitrogen Concentrations in FWS Wetlands**

| System | Years of Data | Excurslon Frequency 50% | 80% | 90% | 95% |
|---|---|---|---|---|---|
| Columbia, Missouri | 3 | 0.97 | 1.37 | 1.70 | 1.80 |
| Duplin County, North Carolina | 7 | 0.29 | 1.40 | 1.71 | 2.58 |
| Oxelösund, Sweden | 5 | 1.00 | 1.29 | 1.38 | 1.41 |
| Hassleholm, Sweden | 5 | 0.91 | 1.40 | 1.54 | 1.67 |
| Brighton, California | 4 | 0.95 | 1.30 | 1.46 | 1.54 |
| New Hanover, North Carolina | 4 | 0.94 | 1.18 | 1.34 | 1.68 |
| Linköping, Sweden | 3 | 0.88 | 1.49 | 1.73 | 1.93 |
| Musselwhite, Ontario | 6 | 0.91 | 1.53 | 2.44 | 2.91 |
| Augusta, Georgia | 6 | 0.72 | 1.33 | 2.35 | 2.59 |
| Titusville, Florida | 7 | 0.86 | 1.51 | 2.30 | 2.66 |
| Listowel 1, Ontario | 4 | 0.95 | 1.53 | 1.91 | 2.10 |
| Listowel 2, Ontario | 4 | 0.85 | 1.57 | 1.99 | 2.68 |
| Listowel 3, Ontario | 4 | 0.96 | 1.56 | 1.91 | 2.73 |
| Listowel 4, Ontario | 4 | 0.96 | 1.60 | 1.98 | 2.08 |
| Listowel 5, Ontario | 4 | 0.89 | 1.45 | 1.65 | 1.80 |
| Brawley, California | 4 | 0.83 | 1.27 | 1.87 | 2.80 |
| Imperial, California | 4 | 0.90 | 1.83 | 2.91 | 3.29 |
| **Median** | | **0.91** | **1.45** | **1.87** | **2.10** |
| **Mean** | | **0.88** | **1.45** | **1.89** | **2.25** |

*Note:* Trend multiplier is $(1 + \Psi)$; see Equation 6.61.

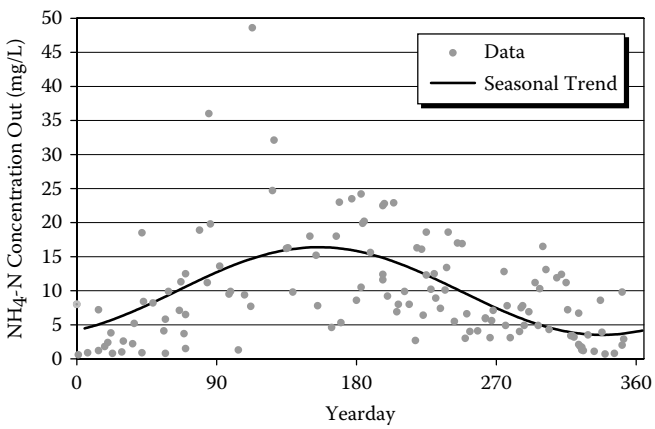



**FIGURE 9.49** Seasonal changes in effluent ammonia concentrations from a HSSF wetland, Staffordshire 3, England. For this example system, a cosine trend is observed, with $A = 0.65$, $t_{max} = 156$ days, and $R^2 = 0.32$. Sampling frequency was weekly, and 2.6 years of system performance is represented. (Other systems will exhibit different seasonal changes, based on temperature, climate, latitude, and influent loadings.) (Data from CWA database (2006) *Constructed Wetlands Interactive Database*, *Version 9.02*. Compiled by Job and Cooper. United Kingdom Constructed Wetland Association (CWA): Gloucestershire, United Kingdom.)

**TABLE 9.35**
**Trend Multipliers for Effluent Ammonia Concentrations in HSSF Wetlands**

| System | Years of Data | Excursion Frequency 80% | 90% | 95% | 99% |
|---|---|---|---|---|---|
| Cumbria, England | 9 | 1.12 | 2.40 | 4.79 | 7.84 |
| Leicestershire 2, England | 5 | 1.74 | 2.35 | 2.75 | 3.17 |
| Ola, Arkansas | 15 | 1.45 | 1.78 | 2.21 | 3.71 |
| Dierks, Arkansas | 12 | 1.70 | 2.18 | 2.34 | 2.73 |
| Las Animas, Colorado | 4 | 1.45 | 1.97 | 2.39 | 2.78 |
| Fife, Scotland (cell 2) | 3 | 1.14 | 1.26 | 1.29 | 1.41 |
| Fife, Scotland (cell 3) | 3 | 1.25 | 1.31 | 1.40 | 1.45 |
| Fife, Scotland (cell 4) | 3 | 1.20 | 1.27 | 1.41 | 1.49 |
| Fife, Scotland (cell 1) | 3 | 1.12 | 1.22 | 1.39 | 1.50 |
| Grand Lake, Minnesota | 2 | 1.52 | 2.40 | 2.78 | 3.42 |
| Nun Monkton, U.K. | 9 | 1.38 | 1.48 | 1.64 | 1.80 |
| Waipoua, New Zealand | 4 | 1.29 | 1.36 | 1.44 | 1.52 |
| NERCC1, Minnesota | 2 | 1.38 | 1.84 | 1.96 | 2.00 |
| NERCC2, Minnesota | 2 | 1.54 | 1.75 | 1.94 | 2.01 |
| **Median** | **3.5** | **1.38** | **1.76** | **1.95** | **2.00** |
| **Mean** | **5.4** | **1.38** | **1.76** | **2.12** | **2.63** |

*Note:* Numbers are the fractional multipliers $(1 + \Psi)$ on the trend as indicated by Equation 6.61; site names for U.K. systems are approximate.

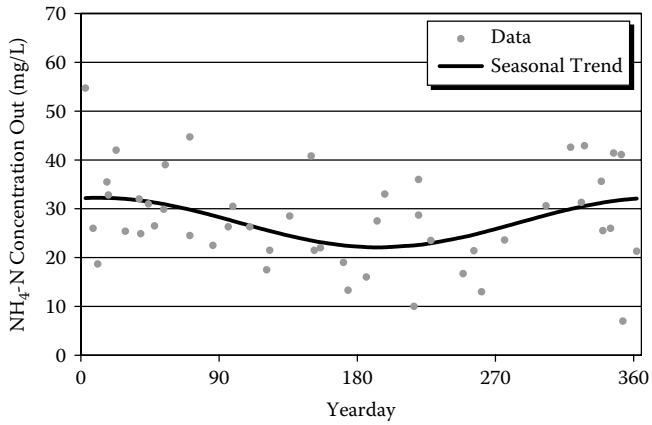



**FIGURE 9.50** Seasonal changes from a VF wetland, Cornwall, England. For this example system, a weak cosine trend is observed, with $R^2 = 0.16$. (Other systems will exhibit different seasonal changes, based on temperature, climate, latitude, and influent loadings). (Data from CWA database (2006) *Constructed Wetlands Interactive Database*, *Version 9.02*. Compiled by Job and Cooper. United Kingdom Constructed Wetland Association (CWA): Gloucestershire, United Kingdom.)

## Variability in VF Wetlands

While vertical flow (VF) wetlands provide an environment conducive to the oxidation of organic and ammonia nitrogen, some variability in the performance of these systems should be expected. This variance can reasonably be expected to be a combination of seasonal changes and stochastic variability. Seasonal changes in the effluent ammonia concentration for an example VF wetland system in Cornwall, England, are illustrated in Figure 9.50.

In addition to seasonal changes, VF wetlands will experience stochastic variability. Data presented over the period-of-record for the 11 pulse-loaded VF systems in Table 9.36 indicate that 10% of the time, the effluent ammonia concentration will be at least 2.27 times the mean effluent concentration.

**TABLE 9.36**
**Trend Multipliers for Effluent Ammonia Concentrations in VF Wetlands Located in the United Kingdom**

| Approximate System Location | Years of Data | Excursion Frequency 80% | 90% | 95% |
|---|---|---|---|---|
| Londonderry, Northern Ireland (gravel bed) | 2 | 1.30 | 1.62 | 1.86 |
| Londonderry, Northern Ireland (peat bed) | 6 | 2.51 | 3.17 | 3.58 |
| Staffordshire 1, England (1st stage) | 3 | 1.34 | 1.47 | 1.75 |
| Staffordshire 1, England (2nd stage) | 3 | 1.47 | 2.59 | 2.64 |
| Buckinghamshire, England (1st stage) | 4 | 3.96 | 5.85 | 7.14 |
| Buckinghamshire, England (2nd stage) | 4 | 2.84 | 4.55 | 11.06 |
| Cornwall, England (1st stage) | 3 | 1.36 | 1.58 | 1.63 |
| Cornwall, England (2nd stage) | 3 | 1.52 | 1.64 | 1.97 |
| Staffordshire 2, England (1st stage) | 2 | 1.60 | 2.07 | 2.27 |
| Staffordshire 2, England (2nd stage) | 2 | 1.90 | 2.27 | 2.48 |
| Somerset, England | 2 | 3.41 | 6.53 | 7.89 |
| **Median** | | **1.60** | **2.27** | **2.48** |
| **Mean** | | **2.11** | **3.03** | **4.02** |

*Note:* Data are approximately monthly. For instance, one month out of ten, we can expect an ammonia concentration 2.27 times the long-term mean value based on the median of the 11 wetlands. Trend multiplier is $(1 + \Psi)$; see Equation 6.61. Site names for U.K. systems are approximate.

## 9.10 PERFORMANCE FOR OXIDIZED NITROGEN

The combination of nitrite and nitrate is oxidized nitrogen, often referred to simply as nitrate, because nitrite is usually a small fraction of the total. Nitrate can serve as a source of nitrogen for plant growth, but may not be the preferred form in the presence of ammonia nitrogen because plants must reduce nitrate prior to further use. Nitrate and nitrite are also important in water quality control because, when present in drinking water, they may result in a potentially fatal condition known as methylglobanemia, or "blue baby" syndrome because in the blood supply, the nitroso group is more readily bound to hemoglobin than oxygen. The current regulatory criteria for nitrate in groundwater and drinking water supplies in the United States is 10 mg/L. This limit has on occasion been exceeded in water supplies in the midwestern United States due to agricultural impacts, leading to deliveries of bottled water for babies and pregnant women. Eutrophication of marine environments is also a nitrate concern. The nitrogen content of the streams and rivers of the midwestern United States is of particular importance at this point in history, because of hypoxia in the Gulf of Mexico, together with the associated ecological and economic consequences (Diaz and Solow, 1999). Nitrogen pollution is of similar concern for the Baltic Sea.

Curiously, nitrate removal has been discussed in the more recent treatment wetland literature, but often with widely diverging viewpoints. U.S. EPA (1999) states that "essentially no relationship exists between nitrate loading and effluent quality …" U.S. EPA (2000a) states that the idea "Constructed wetlands can remove significant amounts of nitrogen" is a "misconception." U.S. EPA (2000a) focuses on the removal of ammonia, but indicts nitrate removal by association. In contrast, Crites and Tchobanoglous (1998) conclude that "When nitrogen is present in the nitrate form, nitrogen removal is generally rapid and complete." Crites et al. (2006) state that "nitrate will be denitrified within a few days of detention." It is the purpose of this section to present information on nitrate removal, and to set forth potential models for calculating reductions.

### Loading Considerations

Nitrate is potentially tied quite closely to the process of nitrification in wetlands that receive both ammonia and oxidized nitrogen, because incoming nitrate loads may be supplemented by produced nitrate. However, there may easily be confusion with processes such as ammonia volatilization (which reduces implied nitrification) and other pathways, such as the reaction of nitrite with ammonia (anammox), in which case denitrification is not necessary.

### Removal of Oxidized Nitrogen in FWS Wetlands

Operational practice with FWS wetlands includes both systems that receive influent nitrogen mainly in the form of nitrate, and also systems where the influent nitrogen is present primarily as TKN and ammonia (often with significant influent BOD concentrations as well). Accordingly, a distinction is drawn here between systems receiving primarily nitrate and those receiving influential loadings of TKN, which may ultimately be dissipated by nitrification–denitrification or other alternate nitrogen chemistries.

**TABLE 9.37**
**Annual Reduction of Nitrate Nitrogen in FWS Wetlands**

**Stipulations**

1. Data restricted to wetlands receiving high proportions nitrate.
2. Period of record averages are used in calculations.
3. For *k*-value calculations, the following *P*-*k*-*C** parameters are selected:
   a. $C^* = 0$ mg/L
   b. *P* = measured *N*-value or 3 TIS
4. Ranges of variables:

| | HLR (cm/d) | $NO_x$-N In (mg/L) | $NO_x$-N Out (mg/L) |
|---|---|---|---|
| **Mean** | 11.4 | 12.8 | 5.9 |
| **Median** | 7.1 | 4.0 | 1.4 |
| **Max** | 110.0 | 121.0 | 66.0 |
| **Min** | 0.6 | 0.05 | 0.01 |

**Results (N = 72 wetlands)**

| Percentile | Load Removed ($g/m^2 \cdot yr$) | Rate Coefficient (m/yr) |
|---|---|---|
| 0.0 | 1 | 2.1 |
| 0.1 | 3 | 9.6 |
| 0.2 | 10 | 14.4 |
| 0.3 | 22 | 18.5 |
| 0.4 | 37 | 22.0 |
| 0.5 | 51 | 26.5 |
| 0.6 | 74 | 29.0 |
| 0.7 | 105 | 33.6 |
| 0.8 | 156 | 38.9 |
| 0.9 | 336 | 54.4 |
| 1.0 | 1,207 | 133.1 |

#### *Nitrate-Rich Influents*

When waters dominated by nitrate pass through a FWS treatment wetland, two processes may dissipate the oxidized nitrogen: denitrification and plant uptake. In the latter case, the nitrate used for growth is in major part returned to the ecosystem as organic or ammonia nitrogen, thus adding to the potential requirement for nitrification–denitrification.

An illustrative set of 72 nitrate-dominated FWS wetlands was analyzed for performance (Table 9.37). As for other nitrogen compounds, there is an increasing outlet concentration in response to increasing nitrate loadings (Figure 9.51). The inlet loadings span a range that includes the agronomic nitrogen loading of 120 $g/m^2 \cdot yr$, and extends well beyond. The actual nitrogen requirement of the vegetation is

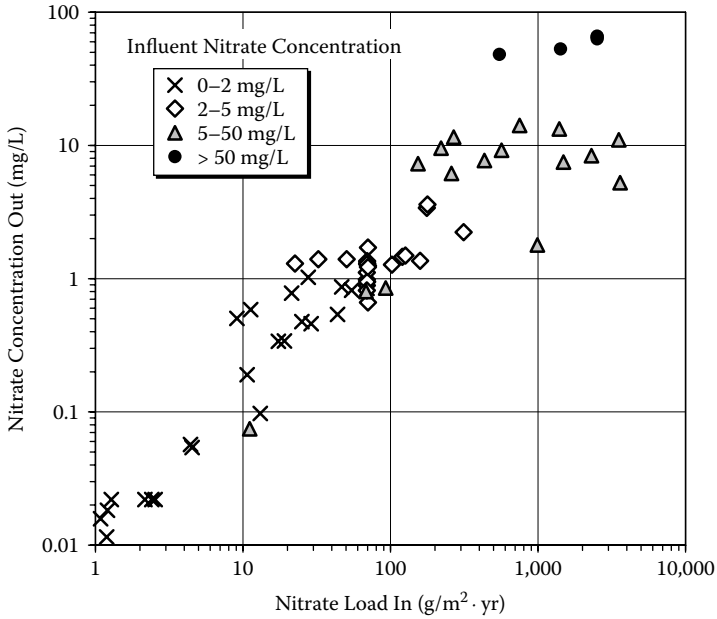



**FIGURE 9.51** Nitrate outlet concentrations from FWS treatment marshes as a function of inlet loading. Each point represents the entire data set for one wetland, typically spanning one to four years. Data are from project operating records and published papers. N = 66.

less than the criterion for vegetative influence on removal, and it is doubtful that the wetland plant and algal community would derive its entire nitrogen requirement from nitrate. In general, the wetland vegetation is capable of supplying the carbon needed for traditional denitrification in these systems, although sulfur may contribute to nitrate reduction in some cases.

### *TKN/BOD-Rich Influents*

The situation is quite different for wetlands which receive and reduce large amounts of organic and ammonia nitrogen. If sequential nitrification–denitrification is supposed, as indicated in Figures 9.14 and 9.16, then the inferred denitrification is much different from the net loss or gain of nitrate from inflow to outflow. In fact, the traditionally supposed route of full nitrification followed by denitrification sometimes becomes highly implausible based on the required oxygen and carbon supplies. To illustrate, consider the performance data for 29 FWS wetlands at four sites, used to treat animal wastewaters, which are very rich in TKN and BOD. Table 9.38 shows the annual mass balances for the various nitrogen species and their theoretical interconversions via traditional nitrification–denitrification. For these wetlands, full nitrification and BOD reduction would require a median of 32 gO/m$^2$·d oxygen supply, which is far above the observed reaeration potential of FWS wetlands. Further, the carbon supply for denitrification, in excess of that provided by the incoming BOD, would be a median of 7,700 g/m$^2$·yr of biomass decomposition, also not a realizable number (see Chapter 3). Thus, the hypothesis of traditional full nitrification followed by denitrification for these TKN-rich FWS wetlands is unlikely to be realistic, and alternate nitrogen chemistries probably play a significant role in these systems.

Alternative removal processes include anaerobic ammonia oxidation, as discussed previously, and close-coupled (simultaneous) nitrification–denitrification, in which the oxidizing power of the nitrate is recovered in ammonia oxidation. These alternatives relieve some of the need for carbon and oxygen supplies, while also eliminating the need for mass transport of nitrate to remotely located wetland anoxic zones. Whatever the mechanism(s) may prove to be, it is clear that there is not necessarily an observed large buildup of nitrate in heavily loaded wetlands.

## Removal of Oxidized Nitrogen in HSSF Wetlands

Typically, HSSF wetlands have low influent nitrate concentrations, and as a result, comparison of influent and effluent nitrate values presents a very incomplete picture of nitrogen processing in these systems. Organic nitrogen may be mineralized to ammonia (via ammonification); ammonia may be oxidized to nitrite or nitrate (by conventional or alternate nitrogen pathways); and oxidized nitrogen may be reduced to $N_2$ or $N_2O$ gas and expelled from the system. To gain a more complete understanding of nitrogen processing, mass balances must be used to determine the full amount of denitrification.

For 22 HSSF wetlands receiving more than 9 mg/L of oxidized nitrogen (Table 9.39), the amount of denitrification is about double the apparent nitrate removal rate. For the entire available HSSF data set of 123 wetlands, denitrification is about five times higher than the apparent nitrate removal rate (inlet–outlet). This is because many HSSF

**TABLE 9.38**
**Examples of Theoretical Mass Balances for Traditional Nitrification–Denitrification in FWS Wetlands**

| System Name | Cell | $NO_3$–N In-Out ($gN/m^2 \cdot yr$) | Theoretical Ammonification ($gN/m^2 \cdot yr$) | Theoretical Nitrification ($gN/m^2 \cdot yr$) | Theoretical Denitrification ($gN/m^2 \cdot yr$) | BOD Removed ($g/m^2 \cdot d$) | Oxygen Needed ($g/m^2 \cdot d$) | Biomass Needed @15% Carbon ($g/m^2 \cdot yr$) |
|---|---|---|---|---|---|---|---|---|
| Oregon State University | 1 | — | 354 | 1,103 | 1,102 | 17.11 | 40 | 1,500 |
| Oregon State University | 2 | — | 350 | 1,138 | 1,138 | 18.43 | 42 | 1,260 |
| Oregon State University | 3 | — | 380 | 1,111 | 1,111 | 17.55 | 40 | 1,394 |
| Oregon State University | 4 | — | 326 | 1,028 | 1,028 | 16.33 | 37 | 1,259 |
| Oregon State University | 5 | — | 294 | 977 | 977 | 15.89 | 36 | 1,061 |
| Oregon State University | 6 | — | 357 | 1,192 | 1,192 | 18.14 | 42 | 1,757 |
| Purdue University, Indiana | A1 | −28 | 77 | 1,186 | 1,157 | 1.40 | 17 | 7,732 |
| Purdue University, Indiana | A2 | −17 | 74 | 1,253 | 1,236 | 1.42 | 18 | 8,288 |
| Purdue University, Indiana | A3 | −3 | 39 | 1,678 | 1,675 | 1.45 | 23 | 11,410 |
| Purdue University, Indiana | A4 | −8 | 79 | 1,191 | 1,183 | 1.47 | 17 | 7,888 |
| Purdue University, Indiana | B1 | −18 | 124 | 2,063 | 2,044 | 2.50 | 30 | 13,651 |
| Purdue University, Indiana | B2 | −18 | 110 | 2,199 | 2,180 | 2.53 | 32 | 14,613 |
| Purdue University, Indiana | B3 | −11 | 48 | 3,041 | 3,030 | 2.44 | 42 | 20,710 |
| Purdue University, Indiana | B4 | −25 | 127 | 1,889 | 1,864 | 2.51 | 28 | 12,364 |
| Purdue University, Indiana | C1 | −14 | 156 | 2,484 | 2,470 | 2.91 | 36 | 16,537 |
| Purdue University, Indiana | C2 | −39 | 88 | 3,254 | 3,215 | 2.57 | 45 | 21,976 |
| Purdue University, Indiana | C3 | −33 | 121 | 2,520 | 2,487 | 2.75 | 36 | 16,716 |
| Purdue University, Indiana | C4 | 3 | 119 | 2,495 | 2,498 | 2.84 | 36 | 16,764 |
| Purdue University, Indiana | D1 | −50 | 210 | 4,680 | 4,630 | 5.38 | 67 | 31,025 |
| Purdue University, Indiana | D3 | 5 | 250 | 4,297 | 4,303 | 4.64 | 61 | 28,965 |
| Purdue University, Indiana | D4 | −32 | 182 | 4,061 | 4,029 | 4.83 | 58 | 26,940 |
| Pontotoc, Mississippi | 1-1 | 0 | 75 | 391 | 390 | 0.32 | 5 | 2,663 |
| Pontotoc, Mississippi | 1-2 | −2 | 132 | 504 | 502 | 0.31 | 7 | 3,464 |
| Auburn Poultry, Alabama | 1-1 | 7 | 284 | 616 | 624 | 3.56 | 13 | 3,126 |
| Auburn Poultry, Alabama | 1-2 | −3 | 63 | 265 | 261 | 1.62 | 6 | 1,261 |
| Auburn Poultry, Alabama | 2-1 | −5 | 293 | 910 | 904 | 3.13 | 16 | 5,281 |
| Auburn Poultry, Alabama | 2-2 | −2 | 64 | 254 | 252 | 1.13 | 5 | 1,376 |
| Auburn Poultry, Alabama | 3-1 | 7 | 245 | 693 | 699 | 2.45 | 12 | 4,075 |
| Auburn Poultry, Alabama | 3-2 | −16 | 46 | 239 | 223 | 2.01 | 6 | 843 |
| **Median** | | **−11** | **127** | **1,191** | **1,183** | **3** | **32** | **7,732** |

*Note:* The oxygen requirement is taken as 1.5 × BOD removed + 4.6 × nitrification. The biomass carbon requirement is taken as 1.07 × denitrification.

wetlands receive and discharge little or no nitrate, but all the ammonia and organic nitrogen loss is through oxidation and reduction (regardless of the stoichiometric pathway).

### *Carbon Supply*

It is well known that HSSF wetlands are limited in their capacity to denitrify unless an adequate carbon source is present. One source is the incoming BOD in the wastewater, and a second is the internal supply of carbon compounds due to plant biomass decomposition. For modest amounts of incoming nitrate, these two sources may be sufficient. However, if the incoming water is from pretreatment that removes BOD effectively, that source is lost. If the vegetation remains standing, biomass decomposition may occur without sending the carbon into the subsurface regions. The studies at Santee, California, showed that denitrification could be restored if carbon supplements were added to the water (methanol, molasses) or to the wetland surface (mulch) (Gersberg *et al*., 1983, 1984, 1986).

Gersberg *et al*. (1984) also showed that a top application of mulch (straw, grass clippings, marsh plant material) enhances nitrate reduction. In the absence of mulch, nitrate removal was from 9–19%. With mulch, the removal increased to 70–95%. This is presumably due to decomposition of the mulch material, which leaches organic carbon into the HSSF wetland bed. If the organic carbon mass transfer from the mulch significantly exceeds the stoichiometric demand for denitrification, BOD removal will be adversely affected (Wallace and Knight, 2006).

## Removal of Oxidized Nitrogen in VF Wetlands

In general terms, vertical flow, pulse-loaded wetlands have an environment that is conducive to oxidation of organic and ammonia-nitrogen. Water is rapidly introduced into these systems to trap air within the wetland bed. Water percolates down through the wetland bed through unsaturated flow mechanisms (see Chapter 2). As a result of this flow regime, the potential for movement of atmospheric gases (notably oxygen) is much higher in VF wetlands than other types of treatment wetlands. The relative availability of oxygen in these wetlands results in a large fraction of the organic

**TABLE 9.39**
**Annual Denitrification in HSSF Wetlands**

**Stipulations**

1. The decomposition of 2,000 g/m²·yr of biomass causes production of 36 gN/m²·yr of organic nitrogen.
2. Inlet oxidized nitrogen above 9 mg/L.
3. Annual averages are used in calculations.
4. For *k*-value calculations, the following *P-k-C** parameters are selected:
   a. $C^* = 0.0$ mg/L
   b. $P = 8$ TIS
5. Ranges of variables:

| | HLR (cm/d) | $NO_x$-N In (mg/L) | $NO_x$-N Out (mg/L) |
|---|---|---|---|
| **Mean** | 12.9 | 18.7 | 10.0 |
| **Median** | 10.9 | 19.4 | 11.3 |
| **Max** | 41.2 | 36.3 | 25.9 |
| **Min** | 1.5 | 3.4 | 0.5 |

**Results (N = 22; N·*t* = 40 wetland-years)**

| Percentile | Denitrification (g/m²·yr) | Rate Coefficient (m/yr) |
|---|---|---|
| 0.05 | 3.3 | 3.3 |
| 0.10 | 7.4 | 7.4 |
| 0.20 | 27.8 | 27.7 |
| 0.30 | 32.7 | 32.0 |
| 0.40 | 40.7 | 40.1 |
| **0.50** | **42.3** | **41.8** |
| 0.60 | 46.9 | 46.4 |
| 0.70 | 75.4 | 73.0 |
| 0.80 | 104.9 | 103.2 |
| 0.90 | 161.5 | 151.8 |
| 0.95 | 188.9 | 173.2 |

and ammonia nitrogen being oxidized to nitrate and nitrite (Figure 9.52).

### Background Concentrations of Nitrate

Nitrate is entirely consumable in treatment wetlands. The presumed value for $C^*$ is zero for all types of treatment wetlands, because no investigation has shown a lower limit to the reduction of nitrate. Minimum reported concentrations appear to be bounded only by the analytical detection limits.

### Rates and Rate Constants

In conventional activated sludge treatment system design, reduction of nitrate is often modeled with a Monod formulation, with a half-saturation constant of 0.1–0.2 mg/L (U.S. EPA, 1993b):

$$J_{AN} = \frac{k' C_{AN}}{K + C_{AN}} \tag{9.52}$$

where

$C_{NN}$ = nitrate nitrogen concentration, mg/L (= g/m³)
$J_{NN}$ = nitrate nitrogen removal flux, g/m²·yr
$k'$ = Monod rate constant, m/yr
$K$ = half-saturation constant, mg/L

Experimental studies on activated sludge indicate that the nitrate half-saturation constant ($K$) is in the range of 0.1 to 0.2 mg/L (U.S. EPA, 1993b). This very low half-saturation constant results in zero-order reaction kinetics (no effect of nitrate concentration) for denitrification at nitrate concentrations above about 1 to 2 mg/L. At lower concentrations, nitrate removal approaches first order. Some wetland literature presumes that nitrate reduction is zero order (Horne, 1995; White and Reddy, 2003; Mitsch *et al*., 2005), which offers the simplistic concept of an implied constant removal rate. However, treatment wetlands are most often best described by first-order denitrification kinetics (with respect to nitrate), even at very high nitrate concentrations (Gale *et al*., 1993; Phipps and Crumpton, 1994; Spieles and Mitsch, 2000; Hume *et al*., 2002a; Kadlec, 2005a). Here, the first-order model is adopted, because it does the best job of describing performance in a wide variety of conditions. Batch cattail FWS mesocosm data confirms that model (Crumpton *et al*., 1993). Full-scale wetlands also show first-order, exponential declines in nitrate, such as that observed in the Lakeland, Florida, system (Figure 9.53). The Lakeland wetland influent is nitrified, with 90% of the dissolved inorganic nitrogen (DIN) occurring as oxidized nitrogen, and therefore this system qualifies as a nitrate-dominated FWS wetland.

Therefore, the area-based first-order expression is the preferred model for the disappearance of oxidized nitrogen (denitrification and other processes combined) in wetland treatment systems:

$$J_{NN} = k_{NN} C_{NN} \tag{9.53}$$

where

$k_{NN}$ = removal rate constant for oxidized N, m/yr

### Oxidized Nitrogen Rate Constants for FWS Wetlands

Results across systems for the value $P = 3$ are given here. The value $C^* = 0.0$ mg/L is used, and the remaining model parameter is the *k*-value, selected to fit the model:

$$\frac{C_{NN,out}}{C_{NN,in}} = \left(1 + \frac{k_{NN}}{3q}\right)^{-3} \tag{9.54}$$

Seventy-two nitrate-dominated wetlands were calibrated for $k_{NN}$. The median annual rate constant was $k_{NN} = 26.5$ m/yr, while the average was $k_{NN} = 30.0$ m/yr (Table 9.37). The 10th–90th percentile range is 9.6–54.4 m/yr. There is a

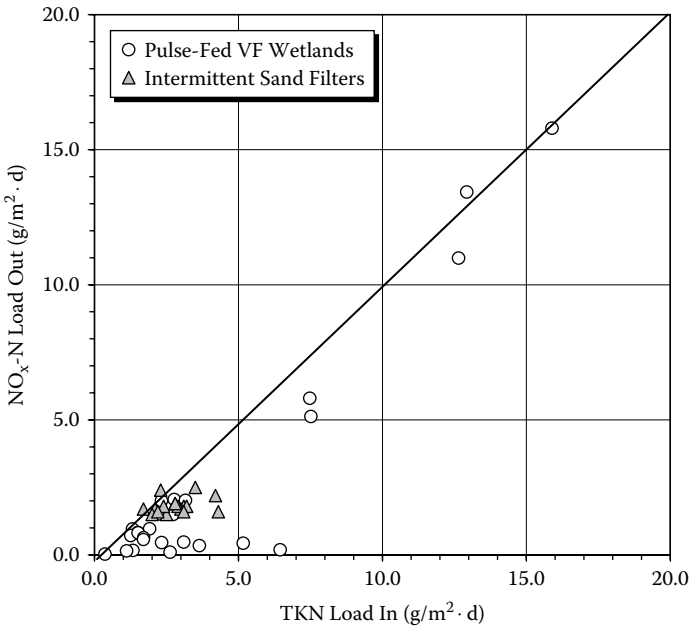



**FIGURE 9.52** Export of oxidized nitrogen ($NO_3^- + NO_2^-$) in pulse-loaded vertical flow wetlands and intermittent sand filters. Data includes period-of-record performance for 20 vertical flow wetlands, annual average reductions for another 6 vertical flow wetlands (17 system-years of data), and annual average reductions for three intermittent sand filters (17 system-years of data) that were operated under similar loading regimes.

significant temperature dependence of nitrate *k*-values; thus even on an average annual basis, temperature or season may be an important determinant of the rate constant, and these factors are thus responsible for the some of the intersystem variability in annual *k*-values. Accordingly, it is necessary to examine intra-annual effects.

Results of calibration of $k_{NN}$-values for entire periods of record for representative wetlands are summarized in Table 9.40. Monthly averages were used to avoid synoptic error (transit time offset). Calibrations were performed for best estimates of the internal hydraulics for each wetland. Therefore, *P*-values range from 2 (New Hanover, measured $P = N = 2$) to near plug flow conditions, based upon system geometry. The median $k_{20}$-value for oxidized nitrogen is 31 m/yr, and the average is 44 m/yr.

There are strong seasonal effects on the nitrate rate constant, which is generally in synchrony with the water temperature. A temperature coefficient (θ) is capable of accounting for these effects (see Chapter 6). Temperature coefficients had a median value of 1.102, indicating a strong thermal effect on the suite of microbial processes that contribute to nitrate reduction, corresponding to a sevenfold reduction in the rate coefficient over a 20°C temperature drop.

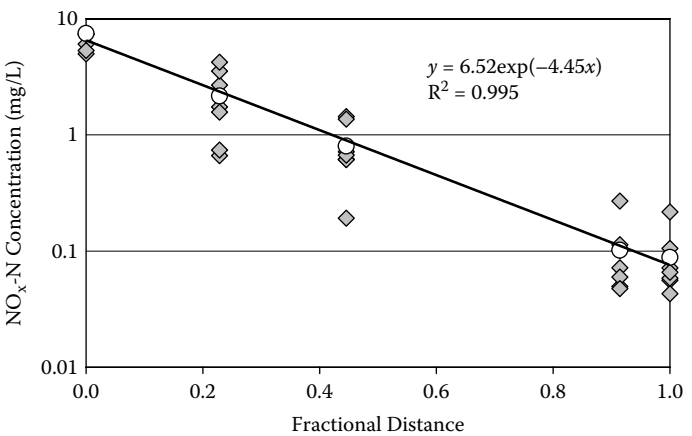



**FIGURE 9.53** Oxidized nitrogen reduction in the Lakeland, Florida, FWS wetland showing first-order, exponential declines. Each point represents the annual mean performance for each of nine years of operation. (Data courtesy of the city of Lakeland.)

**TABLE 9.40**
**Dependence of Oxidized Nitrogen Rate Constants on Temperature for FWS Systems**

| System | Location | Cell | HLR (cm/d) | In (mg/L) | Out (mg/L) | Load In (g/m²·yr) | *P* (TIS) | $k_{20}$ (m/yr) | Theta | Annual T (°C) |
|---|---|---|---|---|---|---|---|---|---|---|
| Lakeland | Florida | Cell 1 | 4.46 | 7.58 | 2.44 | 124 | 3 | 16 | 1.079 | 22.9 |
| Titusville | Florida | All | 0.78 | 1.95 | 0.05 | 5 | 3 | 29 | 1.049 | 21.8 |
| New Hanover | North Carolina | Treated | 1.49 | 101 | 48 | 546 | 2 | 5 | 1.043 | 18.5 |
| Eskilstuna | Sweden | Ekeby | 16.43 | 14.25 | 10.62 | 791 | 6 | 28 | 1.070 | 8.8 |
| Hassleholm | Sweden | Magle | 16.74 | 12.77 | 10.06 | 212 | 3 | 24 | 1.091 | 6.8 |
| Linköping | Sweden | All | 4.38 | 8.59 | 2.29 | 138 | 3 | 114 | 1.054 | 7.0 |
| Tarrant | Texas | Train 1 | 3.02 | 3.91 | 0.42 | 51 | 6 | 64 | 1.170 | 19.2 |
| Imperial | California | All | 29.3 | 2.27 | 1.69 | 243 | ∞ | 30 | 1.136 | 20.7 |
| Tres Rios | Arizona | 0 DZ | 7.14 | 3.38 | 2.13 | 88 | 3 | 26 | 1.174 | 17.7 |
| Tres Rios | Arizona | 1 DZ | 7.16 | 3.38 | 1.58 | 88 | 3 | 32 | 1.167 | 17.7 |
| Tres Rios | Arizona | 2 DZ | 7.15 | 3.38 | 1.58 | 88 | 3 | 31 | 1.210 | 18.1 |
| Tres Rios | Arizona | 3 DZ | 7.15 | 3.38 | 1.66 | 88 | 3 | 29 | 1.186 | 18.2 |
| Tres Rios | Arizona | H1 Premodification | 13.1 | 2.75 | 1.34 | 131 | 6 | 74 | 1.126 | 18.9 |
| Tres Rios | Arizona | H2 Premodification | 12.9 | 2.75 | 1.19 | 129 | 6 | 77 | 1.093 | 18.9 |
| Tres Rios | Arizona | C1 Premodification | 26.0 | 2.91 | 1.56 | 276 | 6 | 79 | 1.123 | 18.4 |
| Tres Rios | Arizona | C2 Premodification | 13.5 | 2.91 | 1.05 | 143 | 6 | 87 | 0.997 | 18.7 |
| Tres Rios | Arizona | H1 Postmodification | 11.4 | 3.77 | 1.14 | 157 | 6 | 26 | 1.177 | 19.9 |
| Tres Rios | Arizona | H2 Postmodification | 13.0 | 3.77 | 1.79 | 179 | 6 | 20 | 1.095 | 20.7 |
| Tres Rios | Arizona | C1 Postmodification | 65.1 | 4.00 | 3.57 | 950 | 6 | 55 | 1.059 | 23.9 |
| Tres Rios | Arizona | C2 Postmodification | 27.7 | 4.00 | 2.45 | 404 | 6 | 39 | 1.108 | 21.7 |
| **Mean** | | | | | | | | **44.3** | **1.110** | |
| **Median** | | | | | | | | **30.6** | **1.102** | |

### *Low BOD Wetlands*

Wetlands with modest inlet BOD do not have enough carbon in the source water to drive full denitrification of the incoming nitrate load. The requirement is not precisely defined, but a range of 5–9 g BOD per gram of nitrate–nitrogen has been suggested (Crites *et al*., 2006, based upon the U.S. EPA 1993 Nitrogen Control Manual). Both nitrified municipal effluents and agricultural runoff are typically very low in BOD, and the required carbon is not present. Accordingly, traditional denitrification requires the carbon generated in wetland sediments and litter, and those are also the locations of the highest numbers of denitrifying bacteria. Nitrate must diffuse to those locations for reaction to occur.

Nitrate nitrogen is a potential nutrient source to support plant growth in the wetland. However, most waters to be treated have some ammonia, which is likely to be the preferred form for plant growth. Further, nitrate would have to diffuse through the top layers of wetland sediment to reach the root zone for uptake, which is the denitrification zone. It is therefore not surprising that there is no evidence of a seasonal nitrate uptake during the growing season. Here it is presumed that microbial processes are responsible for nitrate loss, but the pathways may include more than traditional denitrification.

In those cases where there is a significant contribution of internal wetland nitrification to the nitrate loading, a sequential correction has been made to the kinetic scheme. The method for doing so is described in more detail in Section 9.11. However, there are many situations in which there is little nitrification in the wetland, because incoming and outgoing ammonia nitrogen are low compared to nitrate. In the latter cases, the concentrations of nitrate alone suffice to estimate rate constants.

The first-order model does a creditable job of representing the behavior of this low BOD class of FWS wetlands. It provides a means of accounting for variable hydraulic loads as well as changing temperatures. Nonetheless, random variability remains, as evidenced by the model fit for the Imperial, California, wetland (Figure 9.54).

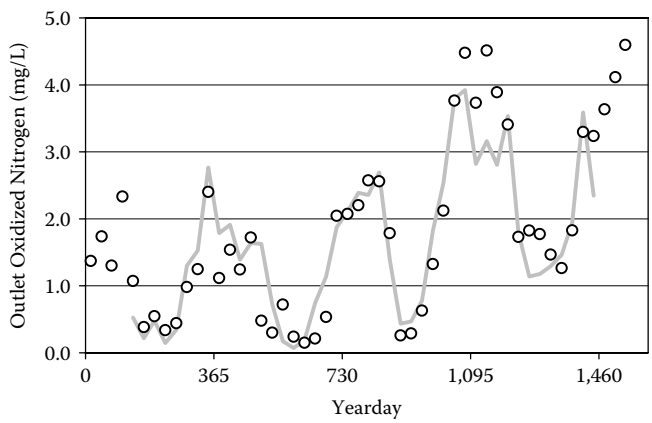



**FIGURE 9.54** Model fit (line) to the monthly oxidized nitrogen data (circles) from the Imperial, California, wetland. Both temperature and flow variations are accounted.

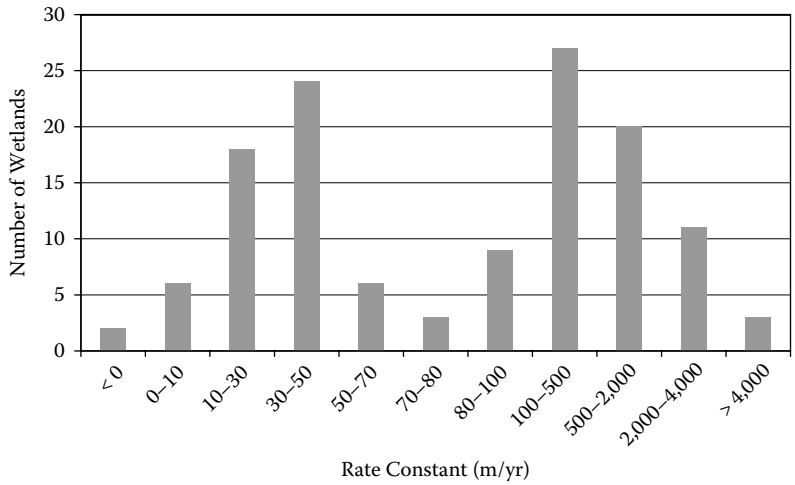



**FIGURE 9.55** Numbers of FWS wetlands with 3 TIS and varying denitrification constants. Of the 129 wetlands, 43% have $k_{NN} < 70$ m/yr, and 53% have $k_{NN} > 100$ m/yr. The latter group typically have heavy BOD loads.

### *High BOD Wetlands*

Oxidized nitrogen is processed quite differently in wetlands with high BOD, with little or no oxidized nitrogen being present in wetland waters. Wetlands in this category include those processing animal, food, and pulp and paper wastes, as well as municipal and domestic effluents high in BOD and low in nitrate. Precise criteria cannot be set from the diverse data available, but these systems possess calibrate rate constants that are 10–100 times greater than those for systems with higher nitrate and lesser BOD (Figure 9.55). For instance, pilot project studies with pretreated (partially nitrified) potato processing water showed $k_{NN} = 34$ m/yr for high nitrate loading, possibly under carbon limitation (Kadlec *et al*., 1997). Subsequent full-scale operation utilized a feed-forward stream of untreated water blended with the pretreated water, and yielded $k_{NN} = 370$ m/yr (Burgoon, 2001).

The high $k_{NN}$-values in Figure 9.55 are not suitable for precise design calculations, for several reasons. Firstly, most are based upon the presumption that TKN losses result in nitrate, to the exclusion of other losses, such as volatilization or anammox-type processes. Those inferred internal nitrate loads are then presumed to be dissipated by denitrification, utilizing the first-order model assumption. The result is that large assumed denitrification must occur at low nitrate concentrations, yielding high rate constants. However, it is true that nitrate accumulations do not typically occur in these systems, and therefore small nitrate rate constants do not fit the data.

### *The Role of Vegetation*

The temperature pattern for $k_{NN}$ values may be confused to some degree by the annual patterns of the vegetation growth and nitrogen utilization in temperate climates. There is maximum growth in spring, and in fall a period of translocation to rhizomes. The annual uptake of nitrogen is often small but nontrivial compared to the removal rates. However, the extent to which plants use nitrate is determined in part by the availability of ammonia, which may be a preferred source of nitrogen (Martin and Reddy, 1997). Direct wetland measurements have shown uptake into shoots was relatively small—6–8% compared to that of denitrification, 61–63% (Matheson *et al*., 2002). Weisner *et al*. (1994) measured only 6–17% of added $^{15}N–NO_3$ in plants after eight days.

One of the strongest indications that the observed nitrate reductions are microbially mediated is the rapid response of wetland ecosystems to varying inlet conditions. Time-varying nitrate concentrations follow a dynamic response that is too fast to involve the wetland vegetative uptake (Spieles and Mitsch, 2000; Werner and Kadlec, 2000b).

Unvegetated open water does not promote effective denitrification. Denitrifying microbes are overwhelmingly located on underwater solid surfaces, including the sediment–water interface and submerged plant parts and litter. Those locations with low dissolved oxygen, or low redox potential, are more effective. The data of Arheimer and Wittgren (1994) yield low calibrated *k*-values for streams (12 m/yr) and for deep water, natural "pond type wetlands" (1.5-m depth, 17 m/yr). Smith *et al*. (2000) have shown nitrate removal proportional to number of shoots in a *Schoenoplectus* spp. wetland. Therefore, vegetated systems are to be preferred.

Deeper water is of little benefit after the anoxic sites in sediments and litter have been immersed. Therefore, added detention time created by deep water is of little or no value in nitrate removal. Data from the Jackson Bottoms, Oregon, side-by-side study (SRI, 1990) indicate that volumetric rate constants decrease with increasing depth. This is evidenced by Figure 9.35 in which the TN is dominated by oxidized nitrogen (60%).

Submergent as well as emergent vegetation has the advantage of producing available carbon within the water column of the wetland. Epiphytic biofilms on living and dead plant material add to the denitrification potential of the system (Eriksson and Weisner, 1997). However, the oxygen production of living underwater plants inhibits denitrification, and

therefore the balance of effects must be evaluated from field data. Weisner *et al.* (1994) found *Potamageton* to be more effective than *Glyceria*, and *Phragmites* stands to be better than open water. Eriksson and Weisner (1997) measured very high rates of denitrification in a reservoir with dense *Potamageton pectinatus*. Conversely, Gumbricht (1993a) found low rates for *Elodea canadensis*. Data from 8 of the 61 systems are in the range of $26 < k < 55$ m/yr, with an average of 37 m/yr. These included values that are characteristic of the central portion of the distribution for nonwoody emergent marshes (see Figure 9.55) with low BOD.

These considerations lead to the conclusion that fully vegetated marshes with either emergent or submergent communities are the preferred option for nitrate reduction. Weisner *et al.* (1994) also reached this conclusion, and suggested that an alternating banded pattern perpendicular to flow would additionally provide hydraulic benefits.

### *Carbon-Starved Wetlands*

At the opposite extreme, denitrification is sometimes observed to occur under conditions of inadequate carbon supply. Examples include the unvegetated (post) Tres Rios wetlands H1 and H2 in Phoenix, Arizona (Kadlec, 2006b), the Imperial, California, wetland (TTI and WMS, 2006), and the Irvine Ranch Water District wetlands in San Joaquin, California (Fleming-Singer and Horne, 2006), which were essentially unvegetated ponds receiving nitrate but little or no carbon. For the first two of these, $20 < k_{NN20} < 30$ m/yr (Table 9.38); for Irvine Ranch, $22 < k_{NN} < 34$ m/yr. A fourth example is the Musselwhite, Ontario, wetland, which receives ponded gold mine wastewater containing essentially no carbon (Bishay and Kadlec, 2005), but had $k_{NN20} = 170$ m/yr. In none of these cases was the vegetation adequate to support the observed nitrate losses.

There is a possibility of algal activity contributing to nitrate removal. Two processes might contribute: uptake and export of phytoplankton, and carbon generation via decomposition of algal debris, capable of supporting denitrification. The first process was examined by Fleming-Singer and Horne (2006), and estimated to be only about 20% of the loss of oxidized nitrogen. Algal productivities at San Joaquin were inadequate to provide the carbon necessary for denitrification.

At least two other possibilities exist whereby the traditional concept of carbon-fueled denitrification may be a secondary route of nitrate loss. As discussed above, sulfur-driven autotrophic denitrification is possible if there is enough sulfur in the waters to provide sulfide in anoxic sediments. That is the case for Tres Rios (approximately 200 mg/L sulfate sulfur in the water), and for Imperial (approximately 600 mg/L $SO_4$-S), which led Wass (2003) to suggest this route for nitrate loss. The second alternative route is anammox, or equivalent close-coupled nitrification–denitrification mechanisms. This may have been a contributing process at Musselwhite, because of the relatively large abundance of nitrite in the incoming water (Bishay and Kadlec, 2005).

## Oxidized Nitrogen Rate Constants for HSSF Wetlands

Results across systems for the value $P = 8$ are given here. The value $C^* = 0.0$ mg/L is used, and the remaining model parameter is the $k$-value, selected to fit the model:

$$\frac{C_{NN,out}}{C_{NN,in}} = \left(1 + \frac{k_{NN}}{8q}\right)^{-8} \tag{9.55}$$

Seventy-two nitrate-dominated wetlands were calibrated for $k_{NN}$. The median annual rate constant was $k_{NN} = 26.5$ m/yr, while the average was $k_{NN} = 41.8$ m/yr (Table 9.39). The 10th–90th percentile range is 7.4–151.8 m/yr.

At present, there is not a sufficient data set to determine the temperature dependency of oxidized nitrogen removal independently for HSSF wetlands, mainly because there are few applications of HSSF technology to remove oxidized nitrogen. Most HSSF wetlands have little or no oxidized nitrogen in the influent, and few are monitored for temperature. However, since reduction of oxidized nitrogen is a microbially mediated process, it is likely that a temperature dependency does exist. The limited data that is available suggests that the temperature dependency is similar to FWS wetlands (Liehr *et al.*, 2000). The use of θ-factors for FWS wetlands (Table 9.40) is recommended as a substitute until more information is available.

## Intrasystem Variability

In common with other forms of nitrogen, oxidized nitrogen is susceptible to random variation around annual trends. Those trends are reasonably well described by *P-k-C** models, but monthly averages display scatter (see, e.g., Figure 9.54). The 95th percentile of that monthly scatter is contained below a line of about double the deterministic annual trend (median relative error = 1.025 × trend) (Table 9.41). The reader is

**TABLE 9.41**
**Trend Multipliers for Effluent Oxidized Nitrogen Concentrations in FWS Wetlands**

| System | Years of Data | Excursion Frequency 50% | 80% | 90% | 95% |
|---|---|---|---|---|---|
| Lakeland 1 | 4 | 0.79 | 1.25 | 1.47 | 1.64 |
| Linköping | 3 | 0.60 | 1.46 | 1.83 | 2.30 |
| New Hanover | 2 | 0.97 | 1.34 | 1.59 | 1.70 |
| Magle | 2 | 1.00 | 1.14 | 1.18 | 1.20 |
| Imperial | 4 | 1.03 | 1.38 | 1.71 | 2.26 |
| Brawley | 4 | 1.11 | 1.67 | 2.08 | 2.32 |
| Ekeby | 2 | 0.90 | 1.02 | 1.17 | 1.42 |
| Musselwhite | 6 | 1.01 | 1.30 | 1.45 | 1.66 |
| Boggy Gut | 5 | 0.78 | 1.73 | 2.25 | 2.50 |
| Tres Rios H1 | 5 | 1.01 | 1.58 | 2.67 | 3.19 |
| **Median** | | **1.00** | **1.38** | **1.71** | **0.95** |
| **Mean** | | **0.93** | **1.39** | **1.74** | **2.02** |

*Note:* Trend multiplier is $(1 + \Psi)$; see Equation 6.61.

again reminded that wetland design must account for both the seasonal variability in the nitrate trends, and the stochastic variation that is superimposed upon it.

## 9.11 MULTI-SPECIES NITROGEN MODELING

The nitrogen processing network exemplified by Figures 9.13–9.16 is assumed to consist of interconversions of nitrogen in the water accompanied by exchanges with the sediments and biomass and the atmosphere. Over the last decade, this network has become recognized as the fundamental basis for simulating nitrogen flows and conversions in treatment wetlands. There are, however, significant differences in literature reports, based upon different hydraulic assumptions, and upon differences in the functional form of transfer rates. As a basis for understanding, a simplified version is presented here.

### Sequential Nitrogen Models: An Illustration

A version of the nitrogen network is shown in Figure 9.56. This highly simplified illustrative nitrogen reaction network restricts nitrogen biomass uptake to ammonium. This is accounted by an uptake flux ($J_{AU}$), where this and other fluxes are in gN/m²·d. Biomass decomposition can also release organic nitrogen into the water via decomposition ($J_R$). The residual of necromass nitrogen is accreted or buried, at rate $J_B$. Incoming organic nitrogen loss is ascribed to ammonification ($J_A$). Ammonia is nitrified at rate $J_N$, thus adding to the amount of nitrate in the system. Nitrate nitrogen (oxidized nitrogen) from all sources is presumed to undergo denitrification, at rate $J_D$.

The hydraulic assumption for this illustration is taken to be steady, nonaugmented flow, with a flow pattern of TIS. Under more complicated assumptions, the water budgets for the TIS are required. This model has been described for a single removal reaction (see Chapter 6), but here it is necessary to extend the ideas to the network of conversions. This reaction network will be presumed to consist of zero and first-order reactions, which represent the simplest possible rate equations that may be chosen. The resulting model for the progress of concentrations as water moves through the wetland is tractable, but necessarily more complicated than for a single species. The mass balances are:

$$Q(C_{ON,in} - C_{ON,out}) = (-J_R + J_A)A_j \tag{9.56}$$

$$Q(C_{AN,in} - C_{AN,out}) = (-J_A + J_{AU} + J_N)A_j \tag{9.57}$$

$$Q(C_{NN,in} - C_{NN,out}) = (-J_N + J_D)A_j \tag{9.58}$$

$$\Delta N = J_{AU} - (J_R + J_B) \tag{9.59}$$

where

$A_j$ = area of the $j$th tank, m²
$J$ = transfer flux, gN/m²·d
$Q$ = flow rate, m³/d
$\Delta N$ = increase in phytomass nitrogen, gN/m²·d

and the rate subscript notation is shown in Figure 9.56.

The rates for waterborne species are specified as first order, as described in the preceding sections of this chapter, according to Equations 9.39, 9.50, and 9.53. The rate constants are assumed to be temperature-dependent, thus requiring the monthly time series of water temperatures (Figure 9.57). Further for this illustration, the uptake, decomposition, and burial rates are specified as seasonally dependent. The monthly time series of phytomass nitrogen content is specified (estimated) (Figure 9.57). The ratio of burial-to-return from vegetation is set, and the burial assumed to be

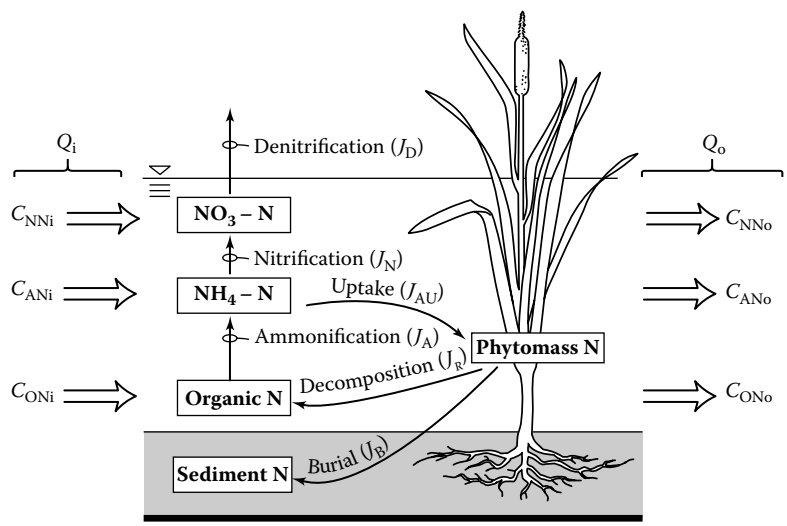



**FIGURE 9.56** Conceptual model for nitrogen routing in a FWS wetland. This simplified version omits factors such as atmospheric gains and losses, infiltration, volatilization, and sorption.

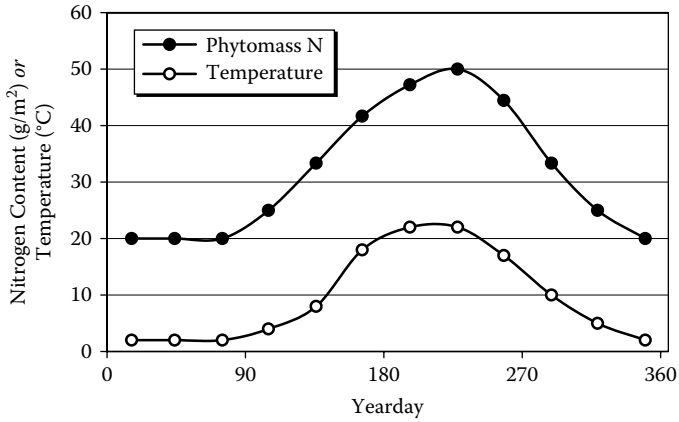



**FIGURE 9.57** Time series of phytomass nitrogen and temperature for a nitrogen model illustration.

constant throughout the year. The assumed rate constants and parameters are given in Table 9.42. Results are shown in Figure 9.58.

Despite the oversimplicity of this illustration, several important features are manifest. This example is set in the range of microbial control, with an inlet ammonia loading of 548 gN/m²·yr. First, the ammonia outlet concentrations display a midsummer minimum, and a winter maximum. However, the component nitrogen transfers are not synchronized. Plant uptake is an important contributor to the annual pattern, with a peak in the spring at maximum growth. Nitrification and denitrification peak much later, and are of the same order of magnitude as plant uptake (and release). Second, the choice of a high denitrification rate constant forces the nitrate levels to remain low throughout the year. Third, there is a considerable return flux of organic nitrogen from decomposition (peaking at 120 gN/m²·yr during October), but the average organic nitrogen content of the water remains low (average = 1.54 mg/L).

There is no calibration data set for this illustration, but several studies have explored variants of this compartment model, and some have performed such calibrations.

**TABLE 9.42**
**Assumed Rate Constants and Parameters for the Nitrogen Species Monthly Model Calculations**

| Parameter | Value | Units |
|---|---|---|
| End-of-season phytomass | 2,500 | g/m² |
| | 50 | gN/m²@2% |
| Winter phytomass | 1,000 | g/m² |
| | 20 | gN/m²@2% |
| Phytomass burial | 10 | gN/m²·yr |
| Phytomass decomposition | 40 | gN/m²·yr |
| Ammonification $k_{20}$ | 30 | m/yr |
| Ammonification θ | 1.02 | — |
| Nitrification $k_{20}$ | 20 | m/yr |
| Nitrification θ | 1.06 | — |
| Denitrification $k_{20}$ | 400 | m/yr |
| Denitrification θ | 1.10 | — |
| Number of TIS | 3 | — |
| System HLR | 10 | cm/d |
| Inlet organic N | 1 | mg/L |
| Inlet ammonia N | 15 | mg/L |
| Inlet nitrate N | 0 | mg/L |

*Note:* The temperature and phytomass time series are shown in Figure 9.57.

### Sequential Nitrogen Models in the Literature

Nitrogen modeling may be divided into efforts targeting FWS wetlands and those which aim to simulate SSF systems. Both applications contain most of the same pieces, with SSF models typically designating a solids compartment related to the media, whereas FWS models contain one or more sediment/soil compartments. The microbial interconversions are the same in character, as is the contribution of plant growth.

### Free Water Surface Systems

#### *Dorge (1994), Jorgensen (1994)*

This work considers two vertical layers: a surface layer and an active layer. Otherwise, there is no consideration of spatial variability, and it is therefore a one-tank model. The model is dynamic, utilizing a daily time-step. The waterborne species are nitrate and ammonia; water-phase organic nitrogen is not considered. Hydrology may be variable, in response to rain and evapotranspiration. Plant uptake is assumed first order, while nitrification and denitrification are described by Monod kinetics. All rate constants are considered to be temperature-dependent. The principal calibration parameters were for mineralization of detrital nitrogen, and uptake rates for nitrate and ammonia. Calibration was conducted for three event-driven Danish wetlands, with good success; but these were not constructed treatment wetlands.

#### *Kadlec and Knight (1996)*

The previous edition of this book contains a description of a steady-state model for accounting the various nitrogen

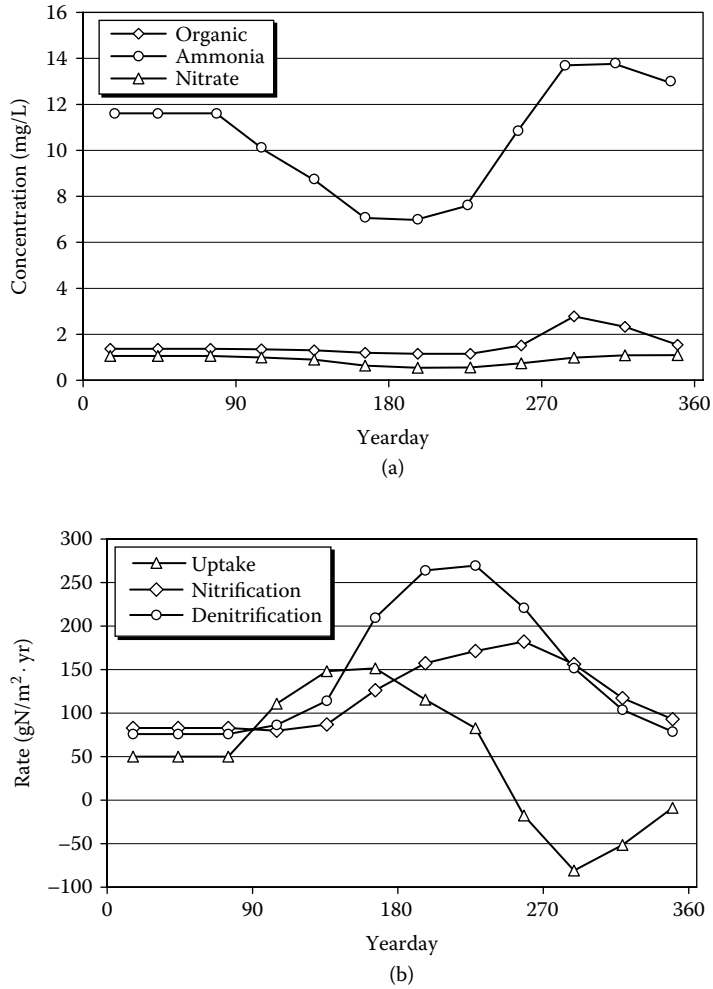



**FIGURE 9.58** Illustration of the monthly time series for a 3-TIS multispecies nitrogen model. (a) Concentrations; (b) monthly fluxes. The incoming water contains 1, 15, and 0 mg/L of organic, ammonia, and nitrate respectively.

species in treatment wetlands. Equations are set forth for the interconversions of waterborne organic, ammonia, and nitrate nitrogen under plug flow circumstances. Plant uptake and return are estimated and superimposed upon the water-phase model. Both ammonia and nitrate were considered available for plant uptake, according to their relative availability. First-order rates were presumed for the water phase, and zero-order for return fluxes. Arrhenius temperature dependence was assumed for rate constants. Calibrations were conducted for several treatment wetlands, primarily FWS and for organic and ammonia nitrogen, with good success.

### *Martin and Reddy (1997)*

These authors set forth a spatially explicit, 2-D steady-state model based upon unvarying flow (no rain or ET effects). The simulated nitrogen species were particulate organic, ammonia, nitrate, and vegetative. Seven vertical zones were considered, consisting of surface water, an aerobic zone under it, and five anaerobic zones below the aerobic zone. Rates were taken to be either zero or first order, with no temperature or seasonal dependence. Ammonia and nitrate were considered available for plant uptake, according to their relative availability. Soil sorption and volatilization of ammonia were included. Movement into the soil zones was attributed to diffusion, but diffusion coefficients had to be included in the calibration set, with very large alterations. Longitudinal compartmentalization was accomplished by dividing the wetland into segments of one-day detention time. Therefore, the model had variable numbers of TIS, corresponding to the number of days of detention. No calibration was attempted, but literature was used to set model parameters and explore sensitivity. Consequently, many assumptions were untested in the treatment wetland environment.

### *Gerke et al. (2001)*

The FWS wetland at Kingman, Arizona, was the platform for a sequential nitrogen model that included ammonification, nitrification, and denitrification. Plug flow was assumed, because the system comprised three cells in series, each with an aspect ratio of 14. Despite the fact that the wetland was observed to have maximum ammonia removal in April, during maximum plant growth, this model had no plant uptake

or release component. Rate constants for organic and ammonia did not display temperature control, but the denitrification constant was very much greater in summer than in winter. First-order areal removal kinetics were presumed, but use of Monod kinetics as tested and found to offer no improvement. Calibration and verification for data from the wetland produced reasonable results, but the authors concluded that calibration was site-specific.

*Howell et al. (2005)*

The Hemet/San Jacinto constructed wetland provided data for calibration of a TIS model including sequential nitrogen conversions. The water-phase conversions are modeled as either first order or Monod, but Monod half-saturation constants were determined to be unimportant. Plant uptake was considered minor, based upon 10% annual turnover in aboveground biomass. No temperature effects were included, despite the fact that ammonia removal was greater in summer. Large seasonal swings in ammonia removal were ascribed to algal uptake, with the summer increase attributed to light availability. The layout of the wetland is star-shaped, with flow inward from the points of unequal arms. However, for computational convenience, a single linear flow geometry, consisting of 10 TIS, was assumed. Model parameters were adjusted to provide a best fit to data. Not surprisingly, algal cycling parameters were found to be the most sensitive.

## Subsurface Flow Systems

*McBride and Tanner (2000)*

This model was built upon the IAWQ model for activated sludge processing of nitrogen, and involved 11 state variables, including biomasses of autotrophic and heterotrophic organisms. Plant uptake was considered, but temperature was not. Sorption to the gravel substrate was found to be an important process. The model was calibrated to dynamic mesocosm operations, with high ammonia (approximately 80 mg/L starting). The hydraulic framework was a single well-mixed unit. The model contained 28 parameters, and was capable of good calibration fits.

*Wynn and Liehr (2001)*

This model included the nitrogen and carbon cycles, autotrophic and heterotrophic bacterial growth, and water and oxygen balances. The TIS hydraulic framework was adopted. A plant growth component was included, but substrate sorption was not. Monod bacterial kinetics were presumed, with temperature coefficients. Bacterial parameters were the most sensitive. The model was calibrated to the Mayo, Maryland, treatment wetland, which treats nitrified influent. Model predictions were not impressive.

*Langergraber (2001)*

This modeling effort simulated 12 constituents and 9 processes. Included were DO, chemical oxygen demand (COD), ammonia, nitrite, nitrate, phosphorus, and autotrophic and heterotrophic organisms. Adsorption was considered, but plant uptake was considered negligible. Volatilization was not considered. Temperature effects were modeled. Dynamic saturated and unsaturated flows were considered, in horizontal and vertical flow SSF wetlands. The hydraulic framework was plug flow plus dispersion. Calibration fits to data were generally good.

*Senzia et al. (2002b)*

This model considered organic, ammonia, and nitrate in water, together with nitrogen in plants and sediments. The computations include dynamics, but are restricted to a single well-mixed unit. Temperature-dependent Monod kinetics were used for microbial processes, with plant uptake being drawn from both nitrate and ammonia in proportion to their abundance. Calibration and validation were carried out using data from the HSSF wetland at the University of Dar es Salaam. The system loading of 530 $gN/m^2 \cdot yr$ was reduced by net plant uptake (71 $gN/m^2 \cdot yr$), accretion (102 $gN/m^2 \cdot yr$), and nitrification–denitrification (80 $gN/m^2 \cdot yr$).

*Liu and Dahab (2004)*

These authors considered each of three SSF wetlands at Lincoln, Nebraska to behave as three TIS in a dynamic simulation. Water-phase concentrations of organic, ammonia, and nitrate nitrogen were assumed to follow first-order temperature-dependent kinetics. Plants were assumed to take up both nitrate and ammonia according to zero-order kinetics. The inflow of 850 $gN/m^2 \cdot yr$ was dominated by ammonia (67%). Nitrogen losses were equally attributed to vegetative uptake and to nitrification–denitrification.

## Sequential Modeling Generalizations

All the existing efforts to account for the speciation of nitrogen as water progresses through treatment wetlands have some elements in common. The water-phase species are organic, ammonia, and oxidized nitrogen. Sediments are important repositories for accreted particulate nitrogen, some of which may be recyclable to the water. There is spatial variability in at least the flow direction, and at least slow dynamics are required to account for changing input flows and concentrations. Most studies agree that microbial processes are temperature-mediated, and most agree that volatilization of ammonia is not significant in most cases.

There is no agreement on the importance of plant uptake and cycling, mostly because of different perceptions of the nitrogen loadings that the wetland has seen or will see. Langergraber (2005) considers several different scenarios, all involving very high hydraulic loadings (up to 0.6 m/d) or very high nitrogen concentrations (up to 72 mg/L). It is, therefore, not surprising that in most instances Langergraber (2005) finds plant uptake to be negligible, with the exception of grey water treatment, in which gross annual plant uptake is estimated at 30 out of an applied 65 $gN/m^2 \cdot yr$.

Another factor that leads to confusion about the importance of plant uptake is the length of time over which uptake occurs. In temperate climates, growth occurs over a period of only three months, resulting in very much higher

instantaneous rates during that period. In general, the gross annual productivity is about 1.3 times the end-of-season standing crop, and so the uptake rate during growth is several times the annual average based on standing crop. In tropical or semitropical climates, this seasonal change in standing crop does not necessarily occur, but that does not mean that biomass turnover is slow. For instance, Davis (1994) has estimated on the order of five turnovers per year in the Everglades of South Florida.

Here it is suggested that a distinction be made between wetlands functioning in the microbial regime and those functioning in the agronomic regime. In the latter case, plant cycling cannot be ignored, particularly as it changes the concepts of timing and temperature dependence of nitrogen removal.

It seems not to be critical to describe bacterial numbers, nor to include Monod kinetics for nitrification or denitrification, because sequential models contain enough parameters to provide adequate results without that level of detail.

The rates of individual conversion processes are sufficiently different that it is not sufficient to describe mixtures by the parameter of TN. Furthermore, regulatory considerations focus on nitrate and ammonia, as well as TN, thus requiring speciated design calculations. To that end, seasonality becomes a factor, and monthly or more frequent time periods need to be analyzed. Most wetlands are not single well-mixed units, or plug flow, and consequently, spatial variability should be included. At a minimum, dynamic, spatially variable spreadsheet mass balancing should be conducted.

## SUMMARY

Organic, ammonia, and oxidized nitrogen forms interconvert in the wetland environment, with the net effect of reductions in virtually all cases. In situations of low nitrogen loading, the biogeochemical cycling of nitrogen is a critical part of the processing network for continuous flow wetlands. Plant nitrogen cycling is important for a major fraction of existing operating wetlands, which leads to seasonal behavior rather than temperature-driven behavior common for microbial processing. Therefore, a distinction is drawn between agronomic wetlands and microbial wetlands, although there are obviously systems with dual control. A benchmark instantaneous growing season rate is suggested to be 120 $gN/m^2 \cdot yr$, below which agronomic control should be considered. In more heavily loaded wetlands, there is preponderance of microbial processes, and removals are strongly temperature modulated. Plants utilize nitrate and ammonium for growth, but the majority of the assimilated nitrogen is subsequently released during death and decay. A small amount is permanently stored as new soil and sediment, on the order of 10 $gN/m^2 \cdot yr$.

The descriptions of microbial processes in conventional activated sludge and attached growth wastewater treatment systems partially carry over to processing in wetland systems under microbial control. However, there are wetland phenomena that are outside those patterns. Traditional nitrification–denitrification may not apply, because of alternative microbial routes, such as anammox or autotrophic denitrification. Conditions of oxygen supply and carbon supply are sometimes outside the bounds of conventional WWTP ideas.

Several features of nitrogen processing are known with some degree of certainty:

1. The profiles of TKN and TN are typically monotonic decreasing with detention time, when the input to the wetland is above background values. The background level of organic nitrogen is approximately 1.5 mg/L in FWS wetlands, and hence that is the background for TN and TKN as well.
2. Ammonia calculations must acknowledge the ammonification process, as well as nitrification and plant uptake. The background level of ammonium nitrogen is approximately zero.
3. Nitrate rate constants divide into two groups, one that follows a slow course dictated by sequential nitrification and denitrification, and one that displays almost instantaneous removal of nitrate. The latter may be attributable to availability and spatial proximity of a carbon supply. Influent BOD is one of the most labile forms of organic carbon within the wetland, and treatment wetlands can "burn" BOD to fuel denitrification. The background level of nitrate nitrogen is approximately zero.

First-order, area-based nitrogen loss models provide a suitable method for design of wetland treatment systems in most circumstances. These have the advantage of correctly describing internal phenomena in flow through wetlands, as well as describing batch wetland operation. Studies on side-by-side wetlands confirm the effects of the principal variables of inlet concentrations and hydraulic loading rates (or the equivalent detention times). The parent mass balance equation for water movement may be adjusted to fit extreme environmental conditions of precipitation or evapotranspiration. The rate equations account for return fluxes from the wetland biomass, and thus can fit the entire range of hydraulic loadings. In parameter estimation, the sequential nature of the nitrogen transformations cannot be ignored.

Wetland treatment systems consistently reduce nitrogen concentrations for many types of wastewaters. The magnitude of these reductions depends on many factors including inflow concentrations, chemical form of the nitrogen, water temperature, season, organic carbon, and dissolved oxygen. Although these factors can be incorporated with some success into design of wetland treatment systems, variability is inevitable. Regardless of the operative mechanisms, there now exist large databases that allow for identification of process rates and rate constants. Both annual and seasonal rates have been measured for dozens of wetlands, and found to cluster, but with considerable intersystem variability.

Designers may now choose the level of risk in sizing a new wetland. Because the major nitrogen transformation mechanisms vary seasonally, conservative design must be based on specific permit limits, with different assumptions used for design to meet annual, monthly, or daily limits. Data indicates that if design is based on deterministic, seasonal equations, target outflow concentrations should be divided by a sizeable factor, of about two, as compensation for the usual maximum monthly limit versus model equations.

The information in this chapter forms the basis for design sizing procedures discussed in Part II of this book. It is clear from the considerations in this chapter that nitrogen processing is too complex to admit simple rules-of-thumb for design.

# 10 Phosphorus

A great deal has been learned about wetland phosphorus processing over the last 15 years (Kadlec, 1999c, 2005c; Toet, 2003; Reddy *et al*., 2005). Treatment wetlands are capable of phosphorus (P) removal from wastewaters, on both short-term and long-term bases. Phosphorus is a nutrient required for plant growth, and is frequently a limiting factor for vegetative productivity. A measure of relative ecosystem requirements is the proportion among the nutrient elements in the biomass, which is often represented as a molar proportion of C:N:P = 106:16:1, or 41:7:1 on a mass basis (the Redfield ratio). Wastewaters do not have this ratio except by rare chance, and most often, there is excess phosphorus in municipal or domestic wastewater. The introduction of trace amounts of this element into receiving waters can have profound effects on the structure of the aquatic ecosystem.

When a wetland, either natural or constructed, is given a new supply of water and phosphorus, it responds by readjusting storages, pathways, and structure. If those new supplies are variable and within the stochastic band of historic inputs, a mature ecosystem will not change in character or function. But, treatment of water for phosphorus removal in the wetland implies that additions will significantly exceed the historic stochastic band of the natural wetland, and a newly constructed wetland will require a successional period to adapt to the intended inputs. In both cases, a period of adaptation and change is to be expected. Thereafter, the wetland functions in a long-term sustained mode, which is tuned to the inputs, but displays probabilistic variation as well. In general, treatment wetlands built specifically for phosphorus removal are area-intensive compared to conventional wastewater treatment technologies.

For the purpose of understanding phosphorus cycling, wetlands may be visualized as consisting of several compartments: water, plants, microbiota, litter, and soil (Figure 10.1). Naturally occurring inputs of phosphorus are from surface inflows, and atmospheric deposition that consists of both wet deposition and dryfall. Outputs may be in the form of surface outflows or infiltration to groundwater. Inputs from groundwater and gaseous release to the atmosphere are less common or probable. Animal migration, ranging from insect movement to fish and bird travel, has been identified as a potential contribution to the phosphorus budget; but to date no quantification of this process exists.

A large number of transfer and alteration processes occur, as indicated in Figure 10.1, but only soil-building provides a net long-term storage of phosphorus. Sediment and soil accretion provides phosphorus storage that can alternate between deposition and erosion on a short-term basis. The wetland environment provides appropriate conditions for net long-term buildups, because inundation slows oxidative processes. Historical accumulations in natural wetlands are the genesis of peatlands. Natural accretions are on the order of a few millimeters per year (Mitsch and Gosselink, 1993).

## 10.1 PHOSPHORUS FORMS IN WETLAND WATERS

Wetland science has evolved to focus on categories of phosphorus compounds that are defined by methods of analysis (Table 10.1). In every case, the analytical procedure is reported as the elemental phosphorus content of the category, which is in contrast to the agricultural practice of reporting as $P_2O_5$. The most reactive forms are the dissolved phosphates, which change hydration in response to pH. The most common species are mono- and dibasic phosphates, which dominate at all typical wetland pH values ($4 < pH < 9$) (Morel and Hering, 1993):

$$H_2PO_4^- \rightleftharpoons HPO_4^{2-} + H^+ \qquad pK = 7.2 \tag{10.1}$$

The generic term used for these inorganic phosphate ions is *orthophosphate* ($PO_4$–P). The molybdate analytical test nominally finds this form of phosphorus, but has been shown to also detect exchangeable phosphorus and colloidally bound phosphorus in eutrophic wetlands (Baldwin, 1988; Hens and Merckx, 2002).

However, phosphorus readily combines with, and may be part of, dissolved organic materials, and in that form has the designation of dissolved organic phosphorus (DOP). In fact, DOP has been characterized in great detail for treatment wetland situations, and found to consist of several kinds of organics (Turner and Newman, 2005). Some of them are readily hydrolyzed by soil enzymes, and together with $PO_4$–P are called soluble reactive phosphorus (SRP). The organic components of SRP can move readily in soils and sediments (Anderson and Magdoff, 2005). Phosphorus also may be associated with suspended particles, and is called PP.

Wetlands provide an environment for the interconversion of all these forms of phosphorus, with the eventual sink being one or more of the wetland solid compartments. A variety of cations can precipitate phosphate under certain conditions. The important potential mineral precipitates in the wetland environment include apatite ($Ca_5(Cl,F)(PO_4)_3$) and hydroxylapatite ($Ca_5(OH)(PO_4)_3$) (Reddy and D'Angelo, 1994). In addition to direct chemical reaction, phosphorus can co-precipitate with other minerals, such as ferric oxyhydroxide and

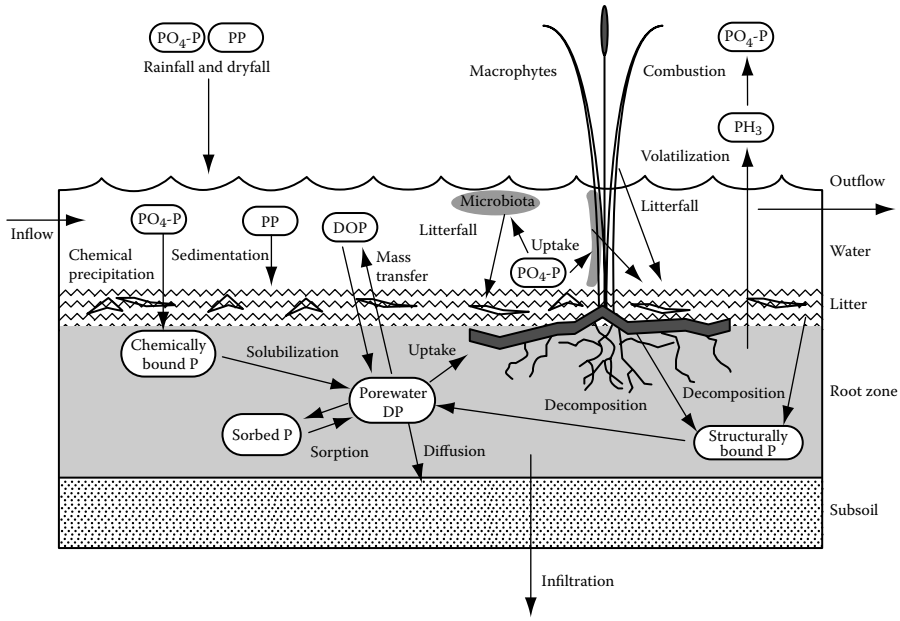



**FIGURE 10.1** Phosphorus storages and transfers in the wetland environment. $PO_4$–P = orthophosphate; PP = particulate phosphorus; DP = dissolved phosphorus; $PH_3$ = phosphine. PP may consist of all the forms shown in the root zone. (From Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

## TABLE 10.1
## Forms of Phosphorus in the Wetland Environment

**Dissolved forms (filtered (0.45 μm) samples):**

- Orthophosphate ($PO_4$–P).
- Condensed phosphates. These consist primarily of pyro-phosphate, meta-phosphate, and poly-phosphates.
- Soluble reactive phosphorus (SRP). $PO_4$–P, together with some condensed phosphates.
- Total dissolved phosphorus (TDP). Phosphorus that is convertible to $PO_4$–P upon oxidative digestion.
- Dissolved organic phosphorus (DOP). Phosphorus, in forms other than SRP, that is convertible to $PO_4$–P upon oxidative digestion (= TDP–SRP).

**Dissolved plus associated with suspended solids.** The procedures above performed on unfiltered samples yield, by analogy:

- Total reactive phosphorus (TRP)
- Total acid hyrolyzable phosphorus (TAHP)
- Total phosphorus (TP)
- Total organic phosphorus (TOP) (= TP–TAHP)
- Particulate phosphorus (PP) (= TP–TDP)

**Sorbed to the surface of soil particles:**

- Sorbed phosphorus is removed using extractants such as water, or solutions of KCl or bicarbonate.

**Contained in the structure of biomass:**

- Total phosphorus may be found by analyzing for $PO_4$–P in digests of biomass samples. Digestion may involve dry or wet ashing, followed by re-dissolution.

**Contained in the structure of soil particles:**

- Structural, internal forms of phosphorus in the solid are removed (solubilized) using harsh extracts of wet soil samples. Typical extractants include:
  - Sodium hydroxide (0.1 M). The SRP in the extract is representative of iron and aluminum bound phosphorus. The balance of the TP in the extract (TP–SRP) is representative of organic phosphorus associated with humic and fulvic acids.
  - Hydrochloric acid (0.5 M). The SRP in the extract is representative of calcium bound phosphorus.
- Total soil phosphorus may be found by analyzing for $PO_4$–P in digests of soil samples. Digestion may involve dry or wet ashing, followed by re-dissolution.

the carbonate minerals, such as calcite (calcium carbonate), $CaCO_3$ (Reddy and D'Angelo, 1994). The overall P-mineral chemistry is very complex; consequently, quantitative calculations of solubilities are generally not possible and computer models are used as estimation tools. Trends for phosphorus movement in wetland soils are as follows (Reddy and D'Angelo, 1994):

- In acid soils, phosphorus may be fixed by aluminum and iron, if available.
- In alkaline soils, phosphorus may be fixed by calcium and magnesium, if available.
- Reducing conditions lead to solubilization of iron minerals and release of phosphorus co-precipitates.

If free sulfide is present due to sulfate-reducing conditions, iron sulfide can form and preclude iron mineralization of phosphorus.

## 10.2 WETLAND PHOSPHORUS STORAGES

Phosphorus compounds are a significant fraction of the dry weight of wetland plants, detritus, microbes, wildlife, and soils, although they are about ten times less than nitrogen compounds. The mass of these phosphorus storages varies in different wetland types, and with the season of the year. A general idea of the relative size of these various storage compartments is necessary to understand the phosphorus fluxes discussed herein (Figure 10.2). The dominant fraction of phosphorus is contained in the wetland soils and sediments. Plants and litter comprise most of the remainder, with very little mass contained in microbes, algae, and water.

SRP is taken up by plants and converted to tissue phosphorus or may become sorbed to wetland soils and sediments. Organic structural phosphorus may be released as soluble phosphorus if the organic matrix is oxidized. Insoluble precipitates form under some circumstances, but may redissolve under altered conditions.

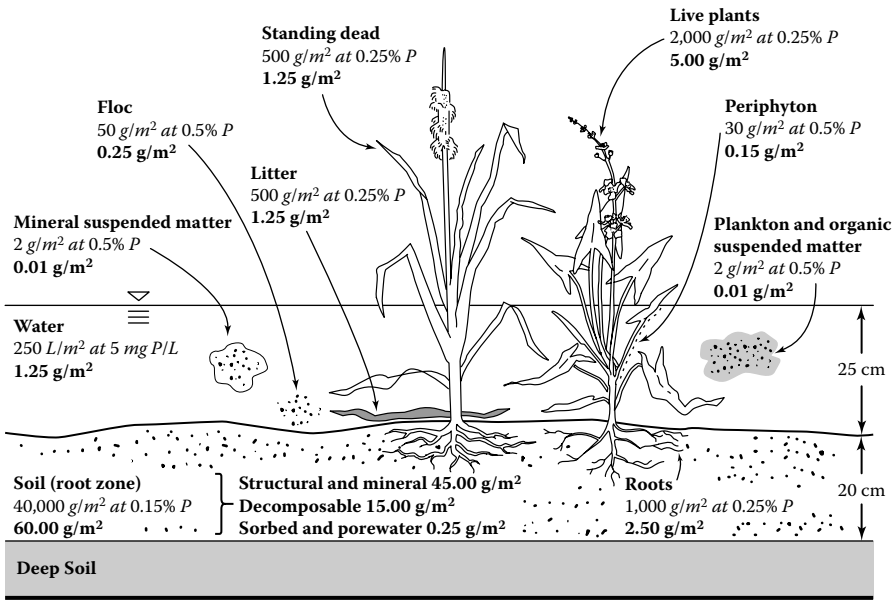
A large amount of the phosphorus in the soil and sediments is present in the organic fraction. Organic phosphorus forms can be generally grouped into:

1. Easily decomposable organic phosphorus (nucleic acids, phospholipids, and sugar phosphates)
2. Slowly decomposable organic phosphorus (inositol phosphates or phytin)

Organic phosphorus can be classed in decreasing order of bioavailability: microbial biomass phosphorus, labile organic phosphorus, fulvic acid-bound phosphorus, humic acid-bound phosphorus, and residual organic phosphorus. In general, large quantities of organic phosphorus can be immobilized in wetland soils, and only a small portion of the total organic phosphorus content is bioavailable. A major portion of organic phosphorus is stabilized in relatively recalcitrant organic phosphorus compounds (Dunne and Reddy, 2005).

### Plant Biomass

The phosphorus content of living biomass in marsh wetlands varies among species, among plant parts, and among wetland sites. There is little variation from location to location within a homogeneous stand (Boyd, 1978). Example ranges of dry weight phosphorus percentages in natural wetlands are: 0.14–0.30% for emergent plants; 0.14–0.40% for floating




Note: Dry mass is in *italics* and standing stock is in **bold**.

**FIGURE 10.2** Example of phosphorus storages in a treatment wetland.

**TABLE 10.2**
**Examples of Phosphorus Content of Various Wetland Plants (mg/kg Dry Weight)**

| Wetland Name | Concentration | Plant Community | Above Live | Above Dead | Below | Reference |
|---|---|---|---|---|---|---|
| **Low P Wetlands** | | | | | | |
| Boney Marsh, Florida | 25–50 µg/L | Mixed emergents | 741 | 327 | — | Unpublished data |
| Houghton Lake, Michigan | 40 µg/L | *Typha latifolia* | 895 | — | — | Unpublished data |
| ENRP, Florida | 50–150 µg/L | *Typha* spp. | 1,343 | 709 | 1,945 | Unpublished data |
| Aiken, South Carolina | "Infertile" | *Typha latifolia* | 1,460 | — | — | Boyd (1971) |
| Lauwersoog, The Netherlands | <1 mg/L | *Phragmites australis* | 1,870 | 520 | — | Mueleman *et al.* (2002) |
| Alderfen, United Kingdom | ~150 µg/L | *Typha angustifolia* | 2,000 | — | — | Mason and Bryant (1975) |
| Alderfen, United Kingdom | ~150 µg/L | *Phragmites australis* | 2,800 | 500 | 1,000 | Mason and Bryant (1975) |
| **Median** | | | **1,460** | **510** | **1,473** | |
| **Treatment Wetlands** | | | | | | |
| Houghton Lake, Michigan | Lagoon | *Typha angustifolia* | 1,842 | — | — | Unpublished data |
| Brisbane, Australia | Secondary | *Baumea articulata* | 1,900 | — | 2,300 | Browning and Greenway (2002) |
| Sacramento, California | Secondary | *Schoenoplectus acutus* | 2,000 | 2,000 | 4,000 | Nolte and Associates (1998b) |
| TVA Mussel Shoals, Alabama | 15 mg/L P | *Schoenoplectus acutus* | 2,000 | 1,000 | 2,000 | Behrends *et al.* (1996a) |
| TVA Mussel Shoals, Alabama | 15 mg/L P | *Schoenoplectus atovirens* | 2,000 | 1,000 | 3,000 | Behrends *et al.* (1996a) |
| TVA Mussel Shoals, Alabama | 15 mg/L P | *Schoenoplectus cyperinus* | 2,000 | 1,000 | 2,000 | Behrends *et al.* (1996a) |
| TVA Mussel Shoals, Alabama | 15 mg/L P | *Phalaris arundinacea* | 2,000 | 1,000 | 2,000 | Behrends *et al.* (1996a) |
| TVA Mussel Shoals, Alabama | 15 mg/L P | *Phragmites australis* | 2,000 | 1,000 | 2,000 | Behrends *et al.* (1996a) |
| TVA Mussel Shoals, Alabama | 15 mg/L P | *Typha* spp. | 2,000 | 1,000 | 3,000 | Behrends *et al.* (1996a) |
| Malham, United Kingdom | Secondary | *Phalaris arundinacea* | 2,000 | 1,000 | 6,000 | Hurry and Bellinger (1990) |
| Brisbane, Australia | Secondary | *Phragmites australis* | 2,400 | — | 3,300 | Greenway (2002) |
| Brisbane, Australia | Secondary | *Philydrum lanuginosum* | 2,400 | — | 3,100 | Browning and Greenway (2002) |
| Listowel, Ontario | Lagoon | *Typha latifolia* | 2,440 | — | — | Herskowitz (1986) |
| Sacramento, California | Secondary | *Typha latifolia* | 2,500 | 2,500 | 3,800 | Nolte and Associates (1998b) |
| Brisbane, Australia | Secondary | *Carex fascicularis* | 2,700 | — | 1,900 | Browning and Greenway (2002) |
| Sacramento, California | Secondary | *Schoenoplectus acutus* | 2,800 | 2,800 | 4,000 | Nolte and Associates (1998b) |
| Houghton Lake, Michigan | Lagoon | *Typha latifolia* | 3,259 | 2,570 | — | Unpublished data |
| Kirinya Jinja, Uganda | Primary | *Cyperus papyrus* | 3,540 | — | 5,890 | Okurut (2001) |
| Brisbane, Australia | Secondary | *Schoenoplectus mucronatus* | 3,900 | — | 3,500 | Browning and Greenway (2002) |
| Kirinya Jinja, Uganda | Primary | *Phragmites mauritianus* | 3,910 | — | 2,600 | Okurut (2001) |
| Sacramento, California | Secondary | *Typha latifolia* | 4,200 | 4,200 | 5,300 | Nolte and Associates (1998b) |
| **Median** | | | **2,400** | **1,000** | **3,050** | |

leaved plants; and 0.12–0.27% for submersed plants (Boyd, 1978). Bedford *et al*. (1999) report a range of 0.1–0.64% for live plant tissue in 41 marshes. The biomass of either live or dead microbiota is virtually impossible to measure, but it is small compared to the biomass of live or dead macrophytes, or macrodetritus (litter).

Treatment wetlands are often nutrient-enriched, and display higher values of tissue nutrient concentrations than natural wetlands. For instance, live cattail leaves in the discharge area of the Houghton Lake wetland averaged 0.18% phosphorus; those in nutrient-poor control areas averaged 0.09% phosphorus (Table 10.2). In general, the median aboveground tissue phosphorus concentration of example wetlands increases from 0.15–0.24% as the water phosphorus increases from 0.1–0.5 mg/L to 3–15 mg/L. If the nutrient status of a wetland is increased from low (*oligotrophic*) to high (*eutrophic*), there is a pronounced increase in tissue phosphorus concentration. The standing dead leaves have lesser phosphorus concentrations than their live counterparts. Litter may have slightly greater or slightly lesser concentrations.

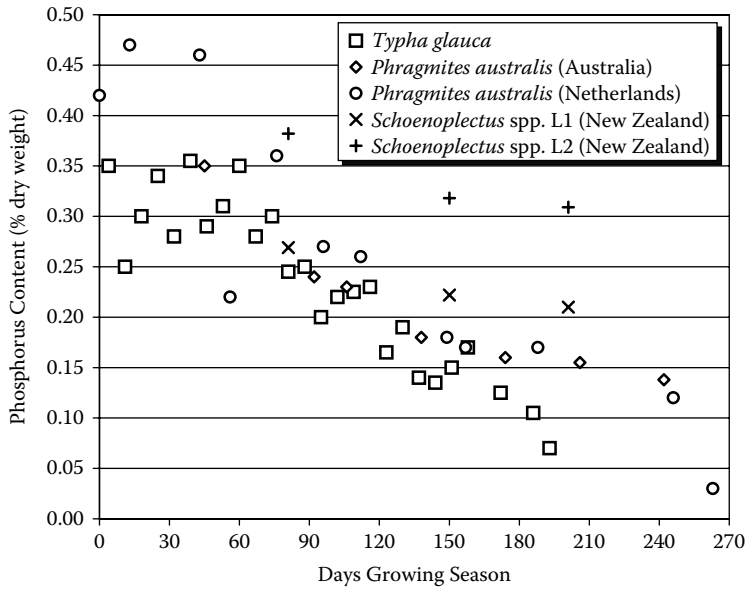



**FIGURE 10.3** The decline of phosphorus content in aboveground tissues of wetland plants. The *Typha* marsh was unenriched, but the *Phragmites* stand was nutrient-enriched. The *Schoenoplectus* (*Scirpus*) wetlands were loaded at 0.2 g P/m$^2$·d and 0.9 g P/m$^2$·d for L1 and L2. (*Typha* data from Bayly and O'Neill (1972) *Ecology* 53(4): 714–719. *Phragmites* Australia data from Hocking (1989b) *Australian Journal of Marine and Freshwater Research* 40: 445–464. *Phragmites* Netherlands data from Mueleman *et al.* (2002) *Wetlands* 22(4): 712–721. *Schoenoplectus* data from Tanner (2001a) *Wetlands Ecology and Management* 9: 49–73.)

The phosphorus content of periphyton and plankton is higher than that for macrophytes, especially in phosphorus rich waters. Vymazal (1995) reports values typically ranging from 0.2–2.0% dry weight (2,000–20,000 mg P/kg) for a dozen different periphyton and plankton species.

The nutrient content of many plant species, including *Typha* spp., display increasing tissue phosphorus with increasing phosphorus availability ranging from 0.05–0.5% dry weight. There is also an increase in standing crop with an increase in nutrient (phosphorus) availability, ranging from about 1,000 g/m$^2$ of biomass at low nutrient to 6,000 g/m$^2$ of biomass at high nutrient conditions.

Aboveground biomass collected at the end of the growing season displays much lower phosphorus content than in spring (Figure 10.3). Klopatek (1978) has shown trends of the same magnitude for cattail shoots, growing under lower nutrient availability. It is apparent that the timing of vegetation sampling can greatly affect subsequent calculations of phosphorus storage and biomass translocation to belowground rhizomes.

Different plant parts may show large differences in phosphorus content, and the seasonal variability may be very large. The extent of this variability is shown in Table 10.3 for *Phragmites australis*, for a reed stand near Griffith, New South Wales, Australia, with a warm dry continental climate and a water phosphorus concentration was 12 mg/L. There is about a factor of two difference among samples due to plant part and location within the plant part.

Plant growth changes the proportions of stored phosphorus in various plant parts as the seasons progress. The growth patterns vary with climate, as discussed in Chapter 3 (see Figures 3.10 and 3.11). The time-varying phosphorus concentrations may be combined with the time-varying biomass for each compartment, which in total represent the phosphorus storage on a seasonal basis. Typically that storage follows the pattern of a growing-season increase to a maximum, followed by a senescence-season decrease to a minimum, with the cycle repeated each year (Figure 10.4).

After the plants are fully mature, there is, on average, no net year-to-year increase in plant storage. However, on a seasonal basis, plant uptake can be a very large part of phosphorus removal or release. For instance, the Lauwersoog, Netherlands, system receives an annual phosphorus loading of 30–40 g P/m$^2$, or an instantaneous loading of 0.08–0.11 g P/m$^2$·d. In Figure 10.4, it is seen that the plants use about 10 g P/m$^2$ over a growing period of 100 days, or an instantaneous removal of 0.10 g P/ m$^2$·d. Thus it seems that 100%

**TABLE 10.3**
**The Variation of Phosphorus Content with Plant Part for *Phragmites australis* at the Peak of the Standing Crop**

| | Stem | Leaves | Whole Shoot |
|---|---|---|---|
| Top | 0.135 | 0.173 | 0.163 |
| Middle | 0.095 | 0.148 | 0.125 |
| Bottom | 0.090 | 0.106 | 0.092 |

*Source:* Data from Hocking (1989b) *Australian Journal of Marine and Freshwater Research,* 40:445–464.

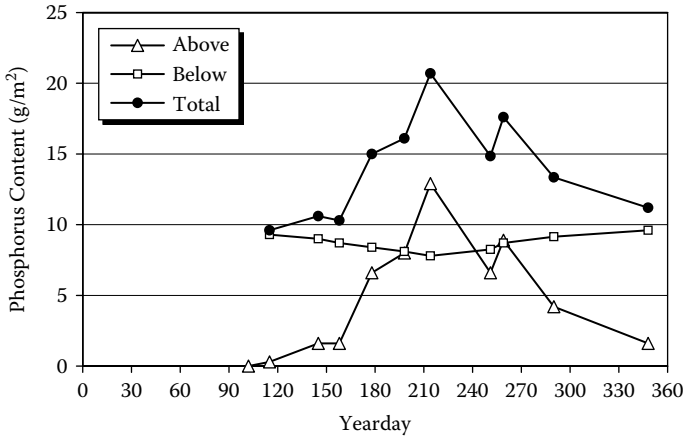



**FIGURE 10.4** Variation of phosphorus storage in the Lauwersoog, Netherlands, VF treatment wetland over the growing season. The amount in belowground organs is relatively constant, but storage in aboveground parts increases, and then decreases. This wetland received water at 10.6 mg/L. (Data from Mueleman *et al.* (2000) *Wetlands*, 22(4): 712–721.)

of the applied phosphorus is utilized by the plants during the growing season. This phosphorus is returned during autumn senescence, less some fraction retained as permanent accretion of nondegradable organic and mineral matter.

### Soils and Sediments

Most of the phosphorus in the soil column is structural phosphorus, both organic and inorganic. Very small fractions are found in pore water or as sorbed phosphorus. Most wetland treatment systems will build an organic sediment and soil layer with time. The first step in soil-building is often the formation of an unconsolidated material of low density, called *floc*. This material partially decomposes, and consolidates with time, forming new soil layers in a FWS wetland. The initial soils in a constructed treatment wetland are the choice of the wetland designer. One choice is to use on-site mineral soils; another is to use imported organic materials, such as peats or compost. Typically, about 30 cm of soil is placed, and becomes involved as the root zone of the wetland plants. Selected soils may be deficient in phosphorus if infertile mineral soils are used, or may contain a surplus of phosphorus if fertile, agricultural soils are employed. Here, we examine the phosphorus content of soils that have adapted to treatment conditions over a period of several years.

### Floc

It has only been in recent years that the presence of a loose and movable layer of material in FWS treatment wetlands has been recognized in the literature. Terminology varies: Kadlec and Bevis (1990) referred to it as "a grey gelatinous material (ooze)"; Nolte and Associates (1998b) as "*Layer A*"; DeBusk *et al.* (2001) as "muck or unconsolidated peat"; and SFWMD (2006) "floc." Floc is easily disturbed, and may then move with water until it resettles (see Figure 7.9). Floc is harvested from tube corers by pouring or vacuuming it out. The phosphorus content is typically 0.1–0.4% dry weight (Table 10.4), but the bulk density is low, about 0.02–0.04 g dry weight/cm$^3$. Nevertheless, thicknesses up to 20–30 cm can contain 4–40 g P/m$^2$. Movement rates have not been measured, but are speculatively about 1,000 m/yr in slow-moving wetland waters. Because most of the phosphorus is structural rather than sorbed in the floc, these solids provide for the slow transport of phosphorus even in waters of extremely low dissolved phosphorus (Gaiser *et al.*, 2005). However, decomposition can later release the phosphorus transported with floc.

### Base Soils

The phosphorus content of organic soils that have experienced only low water-phase phosphorus (about 50 µg/L) are generally in the range 300–500 mg/kg. However, if the ecosystem has long been exposed to higher phosphorus concentrations (e.g., greater than 3 mg/L), then organic soil TP is 1,000–2,000 mg/kg (Table 10.4). These values pertain to the upper-most layer, which may still be experiencing decomposition.

Because of the competition among plant uptake, detrital decomposition, and the transpiration flux in the root zone, there may exist a vertically decreasing profile of soil TP and pore water phosphorus in the soil column. An example is given in Figure 10.5, for the lightly fertilized zone of WCA2A in south Florida. The top soil layer (0–10 cm) typically contains the most roots (see Chapter 3), which remove phosphorus from this zone of high concentration. Those roots also undergo a cycle of growth, death, and decomposition, and hence the return flux of available phosphorus is greatest in this top zone. Lower layers (10–20 cm and 20–30 cm) become progressively depleted by plant uptake,

**TABLE 10.4**
**Example Phosphorus Contents of Wetland Flocs and Sediments**

| System | Water | Vegetation | Horizon | Bulk Density (g/cm³) | TP (mg/kg) | Storage (gP/m²) | Reference |
|---|---|---|---|---|---|---|---|
| **Low P Wetlands** | | | | | | | |
| WCA2A, Florida | 10 µg/L | *Cladium jamaicense* | 0–5 cm | 0.05 | 666 | 1.8 | Reddy *et al.* (1991) |
| WCA2A, Florida | 10 µg/L | *Cladium jamaicense* | 5–10 cm | 0.10 | 380 | 1.9 | Reddy *et al.* (1991) |
| Boney Marsh, Florida | 25–50 µg/L | Mixed Emergents | 0–5 cm | 0.18 | 652 | 5.9 | Unpublished data |
| Boney Marsh, Florida | 25–50 µg/L | Mixed Emergents | 5–10 cm | 0.39 | 487 | 9.5 | Unpublished data |
| STA6, Florida | 19 µg/L | Mixed Emergents | 0–10 cm | 0.60 | 492 | 29.7 | Unpublished data |
| STA5 C1A, Florida | 80 µg/L | Mixed Emergents | 0–10 cm | 0.65 | 446 | 28.8 | Unpublished data |
| STA2 C3, Florida | 26 µg/L | Mixed Emergents | 0–10 cm | 0.24 | 526 | 12.7 | Unpublished data |
| **Moderate P Wetlands** | | | | | | | |
| Sacramento, California | Secondary | *Schoenoplectus acutus* | 0–5 cm | *0.6* | 147 | 4.4 | Nolte and Associates (1998b) |
| Houghton Lake, Michigan | 3 mg/L | *Typha latifolia* | 0–5 cm | *0.2* | 1,268 | 12.7 | Unpublished data |
| Houghton Lake, Michigan | 3 mg/L | *Typha latifolia* | 5–10 cm | *0.2* | 1,180 | 11.8 | Unpublished data |
| Tres Rios H1, Arizona | 3 mg/L | *Scirpus* spp. | 0–5 cm | *0.6* | 1,242 | 37.3 | NADB database (1998) |
| Tres Rios H2, Arizona | 3 mg/L | *Scirpus* spp. | 0–5 cm | *0.6* | 1,365 | 41.0 | NADB database (1998) |
| Tres Rios C1, Arizona | 3 mg/L | *Scirpus* spp. | 0–5 cm | *0.6* | 958 | 28.7 | NADB database (1998) |
| Tres Rios C2, Arizona | 3 mg/L | *Scirpus* spp. | 0–5 cm | *0.6* | 1,294 | 38.8 | NADB database (1998) |
| | | | | **Areal Density (g/m²)** | **TP (mg/kg)** | **Storage (gP/m²)** | |
| **Floc** | | | | | | | |
| Houghton Lake, Michigan | 3 mg/L | *Typha latifolia* | — | 634 | 4,088 | 2.6 | Unpublished data |
| Houghton Lake, Michigan | 0.1 mg/L | *Typha latifolia* + *Carex* spp. | — | 266 | 1,767 | 0.5 | Unpublished data |
| Sacramento, California | Secondary | *Schoenoplectus acutus* | — | 11,630 | 4,000 | 46.5 | Nolte and Associates (1998b) |
| Sacramento, California | Secondary | *Schoenoplectus acutus* | — | 6,320 | 4,000 | 25.3 | Nolte and Associates (1998b) |
| STA6, Florida | 19 µg/L | Mixed Emergents | — | 4,398 | 991 | 4.4 | Unpublished data |
| STA5 C1A, Florida | 80 µg/L | Mixed Emergents | — | 3,509 | 1,075 | 3.8 | Unpublished data |
| STA2 C3, Florida | 26 µg/L | Mixed Emergents | — | 5,700 | 699 | 4.0 | Unpublished data |

*Note:* Bulk densities in *italics* are estimated.

and because of lesser root biomass. The top layer is also the site of newest depositions of detrital and particulate phosphorus (PP).

It is not common for the new sediments and soils in a treatment wetland to be inorganic in character. However, systems treating runoff may receive considerable quantities of inorganic solids from soil erosion in the watershed, which then combine with organic materials generated within the wetland. An example is Chiricahueto marsh in Mexico (Soto-Jiménez *et al*., 2003). Agricultural runoff brought water at about 10.2 mg/L of total phosphorus (TP) to the marsh for over 50 years. The soil column is now mostly inorganic, with less than 5% carbon. Both carbon and phosphorus decreased together as depth increased, indicating

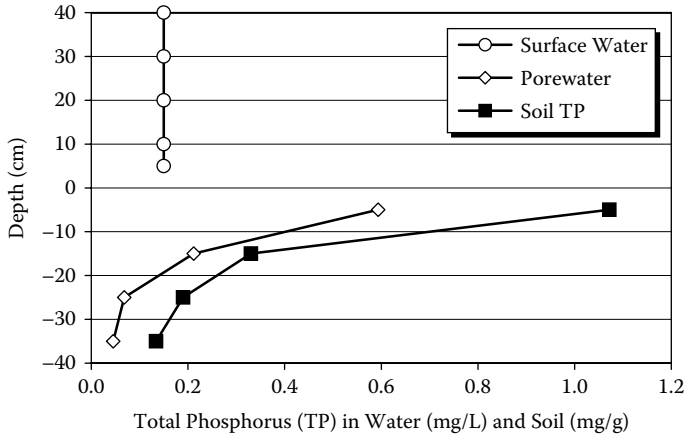



**FIGURE 10.5** Vertical profiles of phosphorus concentrations in water and soil at a station in Water Conservation Area 2A of the Everglades, Florida. The overlying water was at 100–200 µg/L, without much vertical variation. Pore water under the surface was at about four times higher concentration, but rapidly became less concentrated with increasing depth. (Data from Reddy *et al*. (1991) *Physico-chemical properties of soils in the Water Conservation Area 2 of the Everglades*. Report to the South Florida Water Management District, West Palm Beach, Florida.)

that most of the soil phosphorus was associated with the organic content. The phosphorus content of the upper 10 cm at Chiricahueto was 1,200 µg P/g, decreasing to 300 µg P/g at 60–70-cm depth.

## Soil Phosphorus Speciation

Some of the accreted phosphorus occurs as minerals that are not particularly susceptible to remobilization, but new organic substances are subject to degradation if conditions change within the wetland. For instance, dry-out can cause oxidation of some of the stored organics, which may be completely stable under submerged conditions. The associated organic-bound phosphorus may then be mineralized, and redissolve on rewetting (Olila *et al*., 1997; Pant and Reddy, 2001a).

Phosphorus that is bound to ferric iron compounds under oxic conditions may be released if reducing conditions subsequently occur. Under reducing conditions, ferric iron can be converted to soluble, ferrous forms. Soto–Jiménez (2003) speculate that under reduced conditions, decomposition processes of organic matter use manganese oxides and iron oxyhydroxides as electron acceptors, releasing $PO_4^{-3}$ ions to pore water solution. These $PO_4^{-3}$ ions are then transported from the 0–8-cm sediment layer to the water column, where they are transformed into suspended particulate oxides and trapped in the surface layer. This upward migration and reincorporation into the sediment favored the enrichment of phosphorus in the upper sediments. Sorbed orthophosphate may partially desorb if the water concentrations become lower. However, under continuous inundation and stable water chemistry and redox conditions, stored phosphorus is not likely to be released.

The forms of soil phosphorus vary depending upon the chemistry of the wetland that formed them. Figure 10.6 shows two distinct speciations, one is dominated by iron and aluminum compounds, and the other is dominated by calcium and magnesium compounds. The ferruginous system is likely to respond to lowering redox conditions by releasing sorbed phosphorus, but the calcitic system will not.

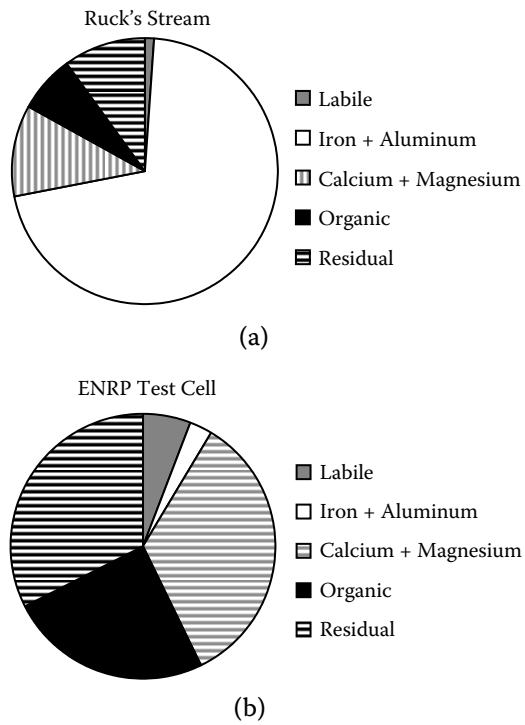



**FIGURE 10.6** Speciation of phosphorus in soils. The upper panel shows a stream bed that is dominated by iron- and aluminum-bound phosphorus. The lower panel shows a cattail wetland dominated by calcium- and magnesium-bound phosphorus, plus a large fraction of refractory organic phosphorus.

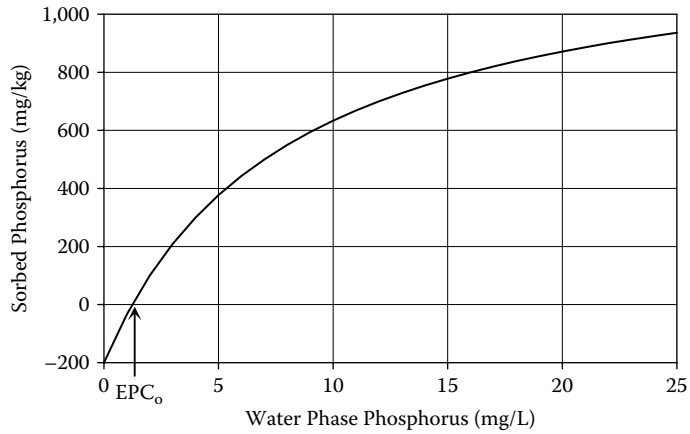



**FIGURE 10.7** Hypothetical example of Langmuir adsorption of phosphorus on a wetland sediment. The initial amount sorbed is $S_o$ = 200 mg/kg, and $EPC_o$ = 1.23 mg/L (arrow). The Langmuir parameters are $S_{max}$ = 1,500 mg/kg and $1/k$ = 8 mg/L.

## 10.3 PHOSPHORUS PROCESSING IN FWS WETLANDS

There are three principal categories of phosphorus removal processes in wetlands: sorption, utilization to build a bigger biomass compartment, and storage as newly created, refractory residuals (burial). The first two mechanisms (sorption and biomass storage) have finite phosphorus retention capacities, whereas the third, accretion, is a sustainable process. Sorption and biomass-building are sometimes termed *saturable mechanisms*, because they reach that limited capacity, and then can remove no more phosphorus. However, *sustainable mechanisms* continue, with no capacity limit. In addition to these, secondary processes such as particulate settling and movement among storage compartments also exist. Particulate settling can rapidly remove large amounts of phosphorus from runoff waters that carry high amounts of suspended sediment. There may also be rearrangements of the phosphorus storages within the wetland that affect its availability and mobility.

### Sorption

There is a misconception that wetlands provide phosphorus removal only through sorption processes on existing soils. It is true that most soils do have sorptive capacity for phosphorus, but this storage is soon saturated under any long-term increase in phosphorus loading. A number of different sorption isotherms have been proposed, with the Langmuir formulation among them:

$$\frac{S}{S_{max}} = \frac{C_{eq}}{C_{eq} + \frac{1}{k}} \tag{10.2}$$

where

$S$ = sorbed phosphorus concentration, mg P/kg soil
$S_{max}$ = maximum sorbed phosphorus concentration, mg P/kg soil
$C_{eq}$ = equilibrium phosphorus concentration in water, mg/L
$1/k$ = half saturation concentration, mg/L

It is sometimes the case that a soil sample contains sorbed phosphorus when it is retrieved from the field. The amount of pre-sorbed initial phosphorus ($S_i$) corresponds to a water concentration termed the *initial equilibrium phosphorus concentration* (at $S_i$, $C_{eq} = EPC_o$), and these are related by Equation 10.2. If water at concentration $EPC_o$ is added to the soil, there is neither adsorption nor desorption. Water at lower concentrations causes desorption, while water at higher concentration causes adsorption. A hypothetical example of sorption behavior is shown in Figure 10.7.

Typical values of the Langmuir sorption parameters are shown in Table 10.5. The maximum sorbed concentration ($S_{max}$) of soils for binding phosphorus have been found to depend strongly upon the amount of iron and aluminum in the soil (Lijklema, 1977; Reddy *et al*., 1991, 1998):

$$S_{max} = a\,(C_{Fe} + C_{Al}) \tag{10.3}$$

where

$a$ = proportionality constant, mg P/mmol
$C_{Fe}$ = oxalate-extractable iron concentration in soil, mmol/kg
$C_{Al}$ = oxalate-extractable aluminum concentration in soil, mmol/kg

The constant $a$ was reported as $0.24 \times 31 = 7.44$ (Reddy *et al*., 1991), and for a different set of soils $0.17 \times 31 = 5.27$ (Reddy *et al*., 1998).

An alternative sorption formulation is the Freundlich isotherm:

$$S = KC_{eq}^{n} \tag{10.4}$$

Sorption is generally described as a two-step process (Kadlec and Rathbun, 1984):

1. Phosphorus rapidly exchanges between the soil pore water and soil particles or mineral surfaces (adsorption).
2. Phosphorus slowly penetrates into solid phases (absorption).

**TABLE 10.5**
**Summary of Langmuir Isotherm Parameters for Natural and FWS Wetlands**

| System Type | Material Type | Material | Location | Half-Saturation Concentration, 1/*k* (mg P/L) | $S_{max}$ (mg P/kg) | Sorbed P @ 1 mg/L (mg P/kg) | Estimated Bed Life[a] (years) | Reference |
|---|---|---|---|---|---|---|---|---|
| FWS | Soil | Dunhill | Ireland | 0.3 | 1,463 | 1,125 | 166 | Dunne *et al.* (2005b) |
| FWS | Soil | Johnstown Castle | Ireland | 0.8 | 618 | 343 | 51 | Dunne *et al.* (2005b) |
| FWS | Soil | Byron Bay Wetland (original) | NSW, Australia | 3.1 | 4,484 | 1,094 | 161 | Sakadevan and Bavor (1998) |
| FWS | Soil | Byron Bay Wetland 2 | NSW, Australia | 5.6 | 4,237 | 642 | 95 | Sakadevan and Bavor (1998) |
| FWS | Soil | Byron Bay Wetland 3 | NSW, Australia | 4.6 | 5,208 | 930 | 137 | Sakadevan and Bavor (1998) |
| FWS | Soil | Carcoar soil | NSW, Australia | 5.3 | 934 | 148 | 22 | Sakadevan and Bavor (1998) |
| FWS | Soil | Richmond subsoil | NSW, Australia | 15.2 | 1,727 | 107 | 15.9 | Sakadevan and Bavor (1998) |
| FWS | Soil | Richmond topsoil | NSW, Australia | 42.5 | 1,153 | 27 | 4.1 | Sakadevan and Bavor (1998) |
| FWS | Soil | BEEC wetland | Nova Scotia, Canada | — | 1,270 | — | — | Stratton *et al.* (2005) |
| FWS | Soil | Eureka wetland | Nova Scotia, Canada | — | 1,862 | — | — | Stratton *et al.* (2005) |
| FWS | Soil | Banweol high marsh | South Korea | 1.3 | 227 | 99 | 15 | Yoo *et al.* (2006) |
| FWS | Soil | Banweol low marsh | South Korea | 0.7 | 260 | 153 | 23 | Yoo *et al.* (2006) |
| FWS | Soil | Donghwa high marsh | South Korea | 1 | 227 | 114 | 17 | Yoo *et al.* (2006) |
| FWS | Soil | Donghwa low marsh | South Korea | 2.8 | 165 | 43 | 6.6 | Yoo *et al.* (2006) |
| FWS | Soil | Samhwa marsh | South Korea | 2 | 155 | 52 | 7.8 | Yoo *et al.* (2006) |
| Natural | Soil | Biscayne wetland soil | Florida | 74.6 | 3,156 | 42 | 6.3 | Zhou and Li (2001) |
| Natural | Soil | Fisheating Creek Basin | Florida | 4.3 | 98 | 18 | 1.8 | Reddy *et al.* (1995) |
| Natural | Soil | Immokalee soil | Florida | 4.7 | 355 | 62 | 9.4 | Graetz and Nair (1995) |
| Natural | Soil | Indian River (aerobic) | Florida | 0.7 | 133 | 78 | 4.0 | Pant and Reddy (2001b) |
| Natural | Soil | Indian River (anaerobic) | Florida | 2.3 | 32 | 10 | 0.7 | Pant and Reddy (2001b) |
| Natural | Soil | Istokpoga Basin | Florida | 2.1 | 28 | 9 | 0.4 | Reddy *et al.* (1995) |
| Natural | Soil | Lower Kissimmee River S-154 Basin | Florida | 7 | 297 | 37 | 1.9 | Reddy *et al.* (1995) |
| Natural | Soil | Lower Kissimmee River S-65D Basin | Florida | 9 | 418 | 42 | 2.3 | Reddy *et al.* (1995) |
| Natural | Soil | Myakka soil | Florida | 1.7 | 505 | 187 | 28 | Graetz and Nair (1995) |
| Natural | Soil | Pomello soil | Florida | 6.8 | 168 | 22 | 3.4 | Graetz and Nair (1995) |
| Natural | Soil | Chao Lake (mesotrophic) | Jiangsu, China | 1.8 | 211 | 75 | 11.3 | Wang *et al.* (2006b) |
| Natural | Soil | East Taihu Lake (mesotrophic) | Jiangsu, China | 0.7 | 208 | 122 | 18.2 | Jin *et al.* (2005) |
| Natural | Soil | Hongze Lake (mesotrophic) | Jiangsu, China | 3.2 | 714 | 170 | 25.2 | Wang *et al.* (2006b) |
| Natural | Soil | Poyang Lake (mesotrophic) | Jiangsu, China | 1.7 | 432 | 160 | 23.7 | Wang *et al.* (2006b) |

**TABLE 10.5 (CONTINUED)**
**Summary of Langmuir Isotherm Parameters for Natural and FWS Wetlands**

| System Type | Material Type | Material | Location | Half-Saturation Concentration, 1/*k* (mg P/L) | $S_{max}$ (mg P/kg) | Sorbed P @ 1 mg/L (mg P/kg) | Estimated Bed Life[a] (years) | Reference |
|---|---|---|---|---|---|---|---|---|
| Natural | Soil | Taihu Lake (eutrophic) | Jiangsu, China | 0.5 | 495 | 330 | 49 | Zhou *et al.* (2005a) |
| Natural | Soil | Taihu Lake (eutrophic) | Jiangsu, China | 2.5 | 531 | 152 | 22.5 | Wang *et al.* (2006b) |
| Natural | Soil | Taihu Lake (hypereutrophic) | Jiangsu, China | 0.3 | 1,490 | 1,146 | 169 | Zhou *et al.* (2005a) |
| Natural | Soil | Taihu Lake (mesotrophic) | Jiangsu, China | 0.6 | 307 | 192 | 28 | Zhou *et al.* (2005a) |
| Natural | Soil | Wuli Lake (hypereutrophic) | Jiangsu, China | 9.2 | 798 | 78 | 11.7 | Jin *et al.* (2005) |
| Natural | Soil | Xuanwu Lake (hypereutrophic) | Jiangsu, China | 3.9 | 667 | 136 | 20.2 | Wang *et al.* (2006b) |
| Natural | Soil | Yue Lake (hypereutrophic) | Jiangsu, China | 8.6 | 833 | 87 | 13.0 | Wang *et al.* (2006b) |
| Natural | Soil | Pokegama River wetland | Minnesota | — | 2,589 | — | — | Bridgham *et al.* (2001) |
| Natural | Soil | St. Louis River wetland | Minnesota | — | 2,564 | — | — | Bridgham *et al.* (2001) |

[a]Bed life is estimated assuming an influent phosphorus concentration of 3 mg/L, an HLR of 3.4 m/yr, an active bed depth of 0.3 m, and an effluent target concentration of 1 mg/L.

Similarly, desorption of phosphorus can also occur in a two-step process (Dunne and Reddy, 2005).

When water of a given concentration is added to a substrate that has sorption capacity, sorption occurs until the entire soil of the wetland is loaded to the solid phase concentration corresponding to that water concentration. The time period to saturation can be short for solids with low sorption capacity, but can be long for soils with high sorption capacity. Furthermore, the initial amount of sorbed phosphorus on the soil affects the time to sorption saturation. Times are longer for soils with less of their capacity used up prior to initiation of exposure. Heavily loaded soils may actually release phosphorus if abruptly inundated with water that contains a phosphorus concentration that is lower than historical phosphorus concentrations (this is sometimes observed during start-up of FWS wetlands).

Table 10.6 lists the saturation times for initially barren, 30-cm-thick substrates exposed to 3 mg/L of phosphorus at a hydraulic loading of 5 cm/d (3.4 m/yr). The median time to saturation listed in Table 10.6 is about one year. It should be noted that the wetland is not "worn out" at this point; rather, one of the phosphorus removal mechanisms has simply been exhausted.

### Biomass Storage and Cycling

Wetland biota undergo a cycle of growth, death, and partial decomposition. This pertains to microbial and algal components of the ecosystem as well as the macrophytes. Of course, the microbial components are present in much lesser quantities, and cycle much more rapidly, than the macrophytes.

### Microbial Phosphorus

Microbial communities of wetland soils are important both for the decomposition of organic material, and remobilization and cycling of nutrients, and thus play a central role in the metabolism of the entire ecosystem (Prenger and Reddy, 2004). Reddy *et al.* (2002) found that approximately 15–25% of the organic phosphorus in phosphorus treatment wetland soils and flocs was microbial. Some estimates place the proportion of phosphorus uptake by microflora and microfauna at about 50% (Richardson, 1985). As the life cycle of these small organisms is short, turnover is quick and it is likely that most of the uptake is returned as DOP and PP, leaving only a small fraction as permanently buried.

Pietro *et al.* (2006), based upon Pietro (1998), found extremely high rates of uptake of orthophosphate in submerged aquatic vegetation (SAV), and associated algae. Half-lives of spike phosphorus doses were found to be in the range of 22–61 minutes. This high rate of uptake obviously cannot be sustained over a long period of time, but does indicate that the SAV community, likely via its microbes, was poised to quickly claim and convert available phosphorus. Similarly, Havens *et al.* (1999) found that various forms of algae (epiphyton, metaphyton, epipelon, surface mat) readily took up phosphorus doses in mesocosms. A total dose of 1,100 μg/L was reduced to 50 μg/L

**TABLE 10.6**
**Summary of Freundlich Isotherm Parameters for Various Substrates in Treatment Wetlands**

| Material | $K_d$ (mg/kg)*(mg/L)$^n$ | $n$ | Sorbed P@1 mg/L (mg P/kg) | Estimated Bed Life[a] (years) | Reference |
|---|---|---|---|---|---|
| Sand | 4.79 | 1 | 4.79 | 0.9 | CH2M Hill (2003a) |
| Peat | 47.1 | 1 | 47.1 | 1.9 | CH2M Hill (2003a) |
| Shellrock | 812 | 1 | 812 | 112 | CH2M Hill (2003a) |
| Quartz gravel | 6.94 | 1 | 6.94 | 1.2 | Breen (1990) |
| Peat | 22 | 1 | 22 | 0.4 | Richardson and Vaithiyanathan (1995) |
| Peat | 340 | 0.48 | 340 | 3.5 | Hammer and Kadlec (1980) |
| Mineral soil | 58 | 0.83 | 58 | 8.2 | Wong and Geiger (1997) |
| 0–10 cm unimpacted peat | 27 | 0.78 | 27 | 0.3 | Richardson and Vaithiyanathan (1995) |
| 10–20 cm unimpacted peat | 31 | 0.84 | 31 | 0.4 | Richardson and Vaithiyanathan (1995) |
| 0–10 cm impacted peat | 12.3 | 0.99 | 12.3 | 0.2 | Richardson and Vaithiyanathan (1995) |
| 10–20 cm impacted peat | 46.5 | 0.8 | 46.5 | 0.5 | Richardson and Vaithiyanathan (1995) |
| Half-burned dolomite sand | 1,190 | 0.353 | 1,190 | 175.2 | Roques *et al.* (1991) |

[a] Bed life is estimated assuming an influent phosphorus concentration of 3 mg/L, an HLR of 3.4 m/yr, an active bed depth of 0.3 m, and an effluent target of 1 mg/L.

in 28 days, with most of this phosphorus storage in the benthos (metaphyton and epipelon communities).

Similarly, $^{32}P$ studies have shown that orthophosphate is very rapidly converted to particulate forms, within five minutes, probably due to microbial processes (Noe *et al*., 2003). Silvan *et al*. (2003) measured 0.75 g P/m$^2$·yr microbial immobilization in a Finnish peatland dosed with phosphorus during the growing season, and concluded that microbial phosphorus immobilization was an important factor in phosphorus removal in the wetland.

Studies such as these indicate very strongly that the first line of interaction with added phosphorus in a wetland is the microorganisms and not the macrophytes. Although the biomass of these organisms is small, they nonetheless can store significant amounts of phosphorus, and, importantly, place the intercepted phosphorus in the sediment–soil systems, where other processes can operate to convert it to recalcitrant forms.

## Vegetation

Plants utilize phosphorus, and decomposition processes release phosphorus back to the water column (Figure 10.8). There are two direct effects of vegetation on phosphorus processing and removal in treatment wetlands:

- *Storage and release*. The plant growth cycle seasonally stores and releases phosphorus, thus providing a "flywheel" effect for a phosphorus removal time series.
- *Sediment accretion*. The creation of new, stable residuals, which accrete in the wetland. These residuals contain phosphorus as part of their structure, and hence accretion represents a burial process for phosphorus.

At any given moment, live aboveground plant material is produced at a rate designated as the gross primary production (GPP) rate (g/m$^2$·yr). At the same point in time, some of the live aboveground plant material is dying. The net primary production (NPP) rate is the gross uptake rate less the death rate. Standing crop is defined as the total amount of live or dead plant material to be found at a given moment. In cold-climate wetlands, nonwoody aboveground growth occurs from a zero starting point in early spring. Most aboveground leaves and stems persist through the growing season, and are measurable as the end-of-season standing crop. Under these circumstances, the turnover of plant material is the ratio of the end-of-season standing crop to the average growing-season GPP rate. Macrophyte turnover in cold-climate environments is usually in the range of 1.0–2.0 reciprocal years, which means the live aboveground biomass is replaced between one and two times per year (Mitsch and Gosselink, 2000a). However, aboveground growth occupies only about one third of the year. Belowground growth, however, continues into late autumn (Prentki *et al*., 1978).

Turnover may be much higher in warm climates, because growth continues over a longer growing season. Macrophyte turnover in warm climates is rapid, with as many as five turnovers of the standing aboveground crop per year (Davis, 1994). Thus, the speed of the cycle depicted in Figure 10.9 is roughly the same during the growing season, regardless of geographic location. But the cycle is much reduced during winter in high latitudes.

Leaf litter decomposes to a stable residual material in a time span of 12–24 warm-weather months; litter from plankton and periphyton decomposes much faster. As the residual is a small fraction of the parent biomass, there is only a small annual buildup of these residual solids. However,

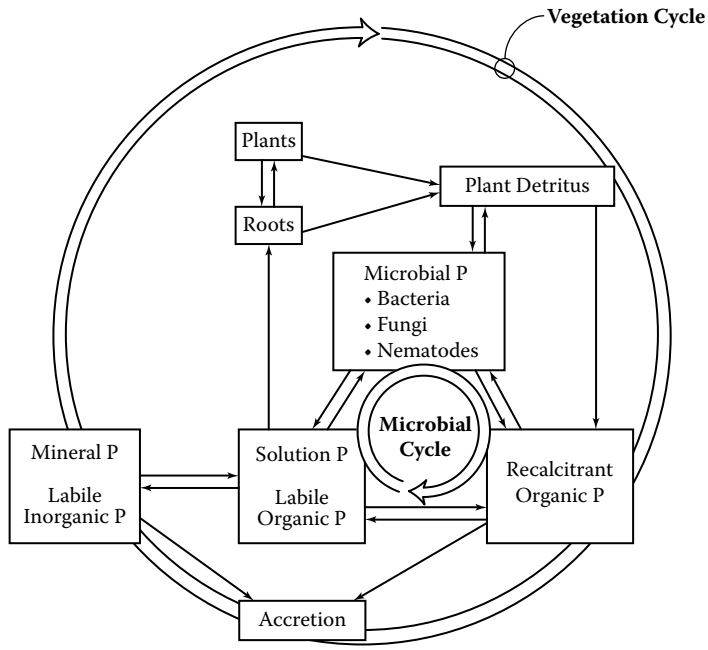



**FIGURE 10.8** The microbial cycle accompanying the vegetation cycle. (Adapted from Reddy *et al*. (1995) *Ecological Engineering* 5:183–207.)

the accretion of biomass residuals and minerals is the only sustainable removal mechanism for phosphorus in FWS wetlands.

## The Effects of Vegetation Growth and Cycling

The removal of phosphorus from water by wetland plants has been the subject of many studies (e.g., Reddy and DeBusk, 1985; Busnardo *et al*., 1992; Tanner, 1996). Many such studies have been characterized by measurements of gross phosphorus uptake, with no deduction for subsequent losses due to plant death and decomposition, with the attendant leaching and resolubilization of phosphorus. The processes of growth, death, litterfall, and decomposition operate year-round, and with different speed and seasonality depending on climatic conditions and genotypical habit. Even in cold climates the total annual growth is slightly larger than the end-of-season standing crop, by about 20% (Whigham *et al*., 1978). In warm climates, measurements show 3.5–10 turnovers of the live aboveground standing crop in the course of a year (Davis, 1994). Decay and translocation processes release most of the phosphorus uptake, with the residual accreting as new sediments and soils.

From the standpoint of phosphorus removal from wetland water, it is the net effect of the macroflora on water-phase

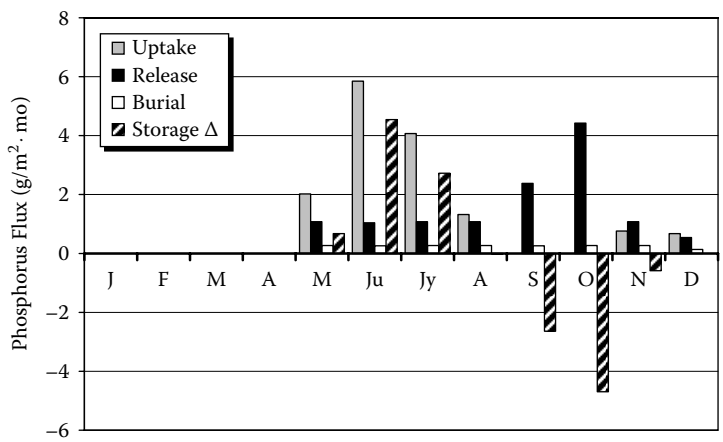



**FIGURE 10.9** Hypothetical seasonal transfers of phosphorus corresponding to the measured growth pattern of Figure 10.4. The phosphorus loading to the wetland was 30–40 g/m$^2$·yr. (Data from Mueleman *et al*. (2002) *Wetlands*, 22(4): 712–721.)

concentrations that is of interest. Here the terminology of Mueleman *et al*. (2002) will be used (see Chapter 3):

- *Phytomass* refers to all vegetative material, living plus dead.
- *Biomass* refers to all living vegetative material.
- *Necromass* refers to all dead vegetative material.

The seasonal patterns of vegetation growth and phosphorus storage embody complex patterns of biomass allocation among plant parts, as well as the phosphorus content of those various portions of living and dead material. However, from the point of view of the annual ecosystem removal of phosphorus, uptake and return from the combination of biomass and necromass are the principal features of concern. On an annual average basis, the only concern is net removal to permanent storage. However, during the course of the year, uptake and return may occur at different times, thus influencing removals differently in different seasons. For these reasons, it is necessary to examine the transfers to and from the collective parts of the macrophytes, which is here defined as phytomass. During the course of the year, especially in temperate climates, phytomass, and its phosphorus content, increases during the growing season, and shrinks during senescence.

### *Mass Balance Framework*

The purpose here is to make order-of-magnitude assessments of the role of vegetation in the overall set of phosphorus processes. This choice has the effect of establishing a "green and brown box" that interacts with the balance of the wetland ecosystem, which has been described in Figure 3.7. The phosphorus mass balance for that storage compartment is (instantaneously):

$$(J_u - J_r - J_b) = \frac{d\mathrm{P}}{dt} \tag{10.5}$$

or for a fixed time period, $\Delta t$:

$$(J_u - J_r - J_b)\Delta t = \Delta \mathrm{P} \tag{10.6}$$

where

$J_u$ = uptake of phosphorus by phytomass (= $U_e$), g/m$^2$·d
$J_r$ = release of phosphorus from phytomass (= $L + D_a + D_b$), g/m$^2$·d
$J_b$ = burial of phosphorus from phytomass (= $A_a + A_b$), g/m$^2$·d
$d$P/$dt$ = storage change rate of phosphorus in phytomass, g/m$^2$·d
$\Delta$P = increase in phosphorus storage in phytomass, g/m$^2$
$t$ = time interval, d
$U_e$ = external uptake rate, g/m$^2$·d
$L$ = leaching rate, g/m$^2$·d
$D_a$ = aboveground decomposition rate, g/m$^2$·d
$D_b$ = belowground decomposition rate, g/m$^2$·d
$A_a$ = aboveground accretion rate, g/m$^2$·d
$A_b$ = belowground accretion rate, g/m$^2$·d

The uptake of phosphorus is via the root system, which is usually below ground. Phosphorus must therefore be transported into the rhizosphere, by processes of diffusion (minor) and vertical movement driven by transpiration flux (major) (Reddy *et al*., 2005). Some of the new plant growth nutrient requirement is supplied by translocation from stores in the rhizomes, and some from uptake from pore water. It is possible that the presence of phosphorus-rich pore waters causes less withdrawal from rhizomes, and results in lesser storage in belowground tissues (Tanner, 2001a).

Phosphorus is returned to surface waters and pore waters via the mechanisms of leaching and decomposition. It is likely that the majority of phosphorus in the necromass is returned, with lesser amounts transferred to permanent burial in the form of new soils and sediment. Over the course of a full calendar year, for a repetitively stable ecosystem, there is no change in the total phytomass, and the net change in phosphorus storage is zero ($\Delta$P = 0). For that annual period, plant uptake is either returned (more) or buried (less). But, as can be seen from Figure 10.4, the total phytomass phosphorus grows in spring and early summer, and recedes in autumn. This annual cycle is more pronounced in cold climates, in response to the more pronounced seasonal conditions.

At this point in the development of knowledge about wetland plant phosphorus cycling, there is some good idea of the change in storage ($\Delta$P) for a given time interval, but less about the three individual fluxes that lead to the storage ($J_u$, $J_r$, and $J_b$).

### *A Speculative Numerical Assessment*

The purpose here is to assess the approximate magnitude of phosphorus withdrawals and returns. Some useful insights may be gained by speculatively assigning uptake and burial. Estimates are:

- A fixed proportion of the necromass phosphorus is returned to the water.
- A constant rate of burial ($J_b$) is apportioned to the unfrozen season.
- Phosphorus release is driven by the amount of necromass during the unfrozen season.

As an order-of-magnitude illustration, an annual phytomass phosphorus cycle is presumed to follow a smoothed version of Figure 10.4. An annual accretion of 2 g P/m$^2$·yr is proposed. This is apportioned over a growing season (unfrozen) of eight months, at a constant rate of 0.25 g P /m$^2$·mo. Four times that amount, 8 g P/m$^2$·yr, is presumed to be returned to water. Growth begins at the end of April, and ends in December, causing phosphorus uptake from April through August, totaling 14.7 g P/m$^2$. During September through December, 4.7 g P/m$^2$ is returned from senescing and decaying necromass from the current year. TP return is 8 + 4.7 = 12.7 g P/m$^2$ for the year, or 87% of the uptake. Only 13% of the phosphorus uptake finds its way into recalcitrant residual forms. However, during the spring growth period, the entire external phosphorus loading is consumed to create the standing crop. The monthly amounts of phosphorus that move into and out of the phytomass are

shown in Figure 10.9. The loading to the wetland was 30–40 g P/m$^2$·yr, or 2.5–3.33 g P/m$^2$·mo. In June and July, growth requirements exceed the external phosphorus loading, and phosphorus must be withdrawn from internal wetland storages other than vegetation. Thus it is seen that vegetative transfers sometimes take up major fractions of the external load.

Treatment wetland data show annual vegetative gross uptakes of 2–10 g P/m$^2$, which occurs during a four to six-month period in temperate climates. This results in instantaneous growing season uptake rates of 4–20 g/m$^2$·yr. Examination of a large number of operational data sets for FWS wetlands leads to the conclusion that emergent and submergent plants are important contributors to the processing of phosphorus in free water surface wetlands. *However, the net uptake that is eventually buried as recalcitrant residual phosphorus is only a very small fraction of the gross uptake. That burial is manifested in accretion of new wetland soils and sediments.*

It must also be remembered that uptake and release from microflora and microfauna is as important as short-term storage. That storage may take place at times other than the growing season of the macrophytes, such as under ice in frozen months. Consequently, the quantitative illustration of macrophyte cycling shown here is not the complete story, and detailed modeling must account for the entire suite of processes. This more complete apportionment of removed phosphorus has been reported in detail by Headley *et al.* (2003).

### Plant Harvest to Remove Phosphorus

The literature contains conflicting "recommendations" concerning the potential for phosphorus removal by harvesting. Crites and Tchobanoglous (1998) conclude: "Harvesting for nutrient removal is not practical and is not recommended." Wang and Mitsch (2000) conclude the opposite: "Harvesting macrophytes has the potential to enhance phosphorus removal in wetlands.… On average, harvesting would remove the equivalence of three quarters of the phosphorus inflow.…" Vymazal (2004) strikes the middle ground, and concludes: "The amount of phosphorus removed via harvesting is usually low. However, for tertiary systems with inflow <20 g P/m$^2$·yr this route could be significant."

Removal of biomass has been investigated as a means of enhancing the removal of phosphorus. However, removals are often less than 10% of the annual load even in lightly loaded wetlands (Herskowitz, 1986; Toet, 2003). It is difficult to harvest rooted emergent macrophytes in wetlands, and when successful, sometimes only small amounts of phosphorus have been reclaimed in the harvested biomass. Herskowitz (1986) reported an average of 2.5% of the TP removal in FWS wetlands was achieved by harvest. Hosoi *et al.* (1998) studied different harvesting strategies for *Phragmites* and concluded that two harvests per year were optimal, each containing approximately 2–3 g P/m$^2$. A survey of data concerning phosphorus removal by harvesting is provided in Vymazal (2004).

Floating aquatic plants are somewhat easier to harvest; in a study by Fisher and Reddy (1987), over 20% of the TP removal was achieved by water hyacinths (*Eichhornia crassipes*). Harvesting is labor intensive and costly, which is antithetical to the passive character of wetland technology. The problem of biomass utilization exacerbates the difficulties. CH2M Hill (2003b) found that *Eichhornia* was effective at reducing phosphorus in the 0.1–0.6 mg/L range, with an average removal of 62% ascribable to wetland processes. However, only 8.6% of the inlet load was contained in harvest on a monthly schedule, which amounted to 15% of the removed load. Harvest was approximately 4 g P/m$^2$·yr.

The various results and conclusions are in fact all part of a fairly simple pattern. The range of standing crops and their phosphorus content is not widely variable, being mostly constrained by space under conditions of ample nutrient supply. The aboveground, harvestable amount of phosphorus typically ranges 1–5 g P/m$^2$. Multiple harvests might claim a bit more than one crop per year, but there is a danger of stand damage. However, the inlet phosphorus loadings to treatment wetlands are extremely variable, spanning the range of 1–10,000 g P/m$^2$·yr. The fraction of the incoming load that may be harvested therefore theoretically decreases logarithmically as the inlet phosphorus load increases, and data from wetland harvesting studies bear this out (Figure 10.10). As a result, whether or not harvesting could be important depends almost entirely on that inlet loading.

Harvest may involve complete removal in the case of floating plants (*Lemna minor*, *Eichhornia crassipes*), or cutting of aboveground parts of rooted plants (*Typha, Schoenoplectus, Phragmites*). Harvesting typically requires expensive mechanical equipment, and is labor-intensive. For instance, a one-time harvest of floating mats of *Typha* in a Florida treatment wetland cost about $16 per cubic meter of wet material, which translates to about $40 per kilogram of phosphorus removed, and that cost did not include biomass management or disposal.

As noted in Chapter 9, the problem of biomass disposal is often not easily resolved. Biomass may either be composted, or digested to form a biogas product. Composting requires transportation costs, and dedicated land area. Biogas generation has a sludge disposal problem. The capital cost of harvesting and gas generation is often prohibitively expensive (Chynoweth, 1987). As a consequence of these difficulties, plant harvesting is not favored for phosphorus removal, regardless of efficacy, and has seldom been used except for floating plants.

### Herbivory and Phosphorus Movement

Animals can serve as vectors of nutrient transport in wetlands, both by moving nutrients between ecosystems, and by recycling within ecosystems (Frederick and Powell, 1994). Even insect emergence has been considered (Neame, 1976) (12 mg P/m$^2$·yr). However, the greatest potential for influence comes from animals (muskrats, nutria) and birds (ducks, geese, swans). The removal of standing crop by muskrats (*Ondatra zibethicus*) during an unfrozen season in southern Manitoba, Canada, was estimated to be about 82 g dry biomass per kilogram of body mass per day, or per animal, because an adult muskrat weighs approximately 1 kg (Clark, 2000). Consequently, muskrats contributed to observed

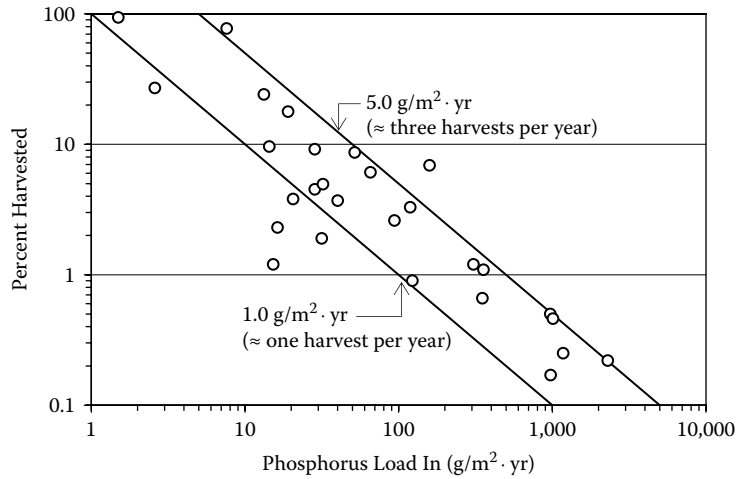



**FIGURE 10.10** The fraction of applied phosphorus loading that can be removed by plant harvesting. Data are from field-scale projects, and span a variety of constructed wetland types. The lines represent rates of harvest, which reflect one to three cuttings per year.

vegetation declines at nontreatment wetlands at Delta, Manitoba, but were not the major factor (Clark, 2000). However, it then becomes clear that densities of twenty or more muskrats per hectare may utilize the majority of the macrophyte standing crop in a given year. At such an exacerbated scale, muskrat herbivory may be termed *eatout*, and will be evidenced by the removal of essentially all emergent plant parts. The attendant movement of phosphorus at 0.2% dry weight would be 164 mg per animal per day, or 60 g phosphorus per animal per year. At a population density of 20/ha, phosphorus movement would be 0.12 g/m$^2$·yr.

Birds also have the ability to move phosphorus, especially when large colonies or migratory flocks are involved. Phosphorus loading rates (PLR) in historical colonies of wading birds in the Florida Everglades have been estimated at over 100 g/m$^2$·yr (Frederick and Powell, 1994). More importantly, migratory waterfowl have been observed to have profound effects on *Phragmites* stands in the Netherlands (van den Wyngaert *et al*., 2003). Goose (*Anser anser*) densities of 6–10/ha lowered the standing crop of leaves and rhizomes by about 50% across a 3,600-ha wetland, but did not destroy the stands. The amount of phosphorus moved was about 0.8 g/m$^2$ during the summer. Submerged macrophytes, such as *Potamogeton pectinatus*, are attractive sources of food to migratory waterfowl, such as swans—*Cygnus* spp. (Jonzen *et al*., 2002). However, resident waterfowl populations on open water or SAV systems have a year-round effect in warm temperate climates. Scherer *et al.* (1995) found loadings of about 0.155 g P/m$^2$·yr due primarily to coots (*Fulica americana*) and mallard ducks (*Anas platyrhynchos*) in a lake in the State of Washington.

The potential impacts of a water bird population may be roughly estimated from their numbers and the production of fecal matter. Ducks produce 17–27 g dry weight per day, coots 13, and geese 82 (Scherer *et al*., 1995). The phosphorus content is about 2% dry weight. It may easily be shown that large flocks of birds may influence the ability of treatment wetlands to achieve ultra-low phosphorus concentrations, but will not affect treatment at the secondary or primary level.

## Sustainable Removal: Accretion

The least studied aspect of phosphorus transfer in wetlands is in the creation of new soils and sediments, with their attendant phosphorus content. Not all of the dead plant material undergoes decomposition. Some small portions of both aboveground and belowground necromass resist decay, and form stable new accretions. Such new stores of phosphorus are presumed to be resistant to decomposition. The origins of new sediments maybe from remnant macrophyte stem and leaf debris, remnants of dead roots and rhizomes, and from undecomposable fractions of dead microflora and microfauna (algae, fungi, invertebrates, bacteria).

The burial of plant detrital residuals is of considerable importance. The majority of the assimilated phosphorus is subsequently released during death and decay, but 10–20% is permanently stored as the residual from the decomposition process. The amount of such accretion has been quantified in only a few instances for FWS (Reddy *et al*., 1993; Craft and Richardson, 1993a; Rybczyk *et al*., 2002), although anecdotal reports also exist (Kadlec, 1997a). Quantitative studies have relied upon either atmospheric deposition markers (radioactive cesium or radioactive lead), or introduced horizon markers, such as feldspar or plaster. Either technique requires several years of continued deposition for accuracy. Representative accretion rates are given in Table 10.7.

Mass balances may also be used to estimate the burial flux. For unfertilized peat wetlands, phosphorus storage is very low, about 0.005–0.024 g/m$^2$·yr (Richardson, 1985). Phosphorus accumulation rates in organic soil natural wetlands in the United States have been reported in the in the range of 0.06–0.90 g/m$^2$·yr (Craft and Richardson, 1993b, 1998). Murkin *et al*. (2000) found 0.73–0.95 g P/m$^2$·yr annual accretion rates for low nutrient, mixed marshes in Manitoba.

**TABLE 10.7**
**Phosphorus Accretion Rates in FWS Wetlands**

| Location | Reference | Method | Water Nutrients | Accretion (cm/yr) | Phosphorus Burial (gP/m²·yr) |
|---|---|---|---|---|---|
| Louisiana | Rybczyk *et al.* (2002) | Feldspar | L | 0.14 | 0.36 |
| Michigan | Kadlec and Robbins (1984) | Lead 210 | L | 0.20 | 0.24 |
| Everglades WCA2A | Reddy *et al.* (1993) | Cesium 137 | L | 0.39 | 0.18 |
| Everglades WCA2A | Craft and Richardson (1993b) | Cesium 137 | L | 0.16 | 0.06 |
| Everglades WCA3 | Craft and Richardson (1993b) | Cesium 137 | L | 0.20 | 0.08 |
| Chiricahueto, Mexico | Soto Jiménez *et al.* (2003) | Lead 210 | M | 1.00 | 0.40 |
| Everglades WCA2A | Craft and Richardson (1993b) | Cesium 137 | M | 0.40 | 0.46 |
| Everglades WCA2A | Reddy *et al.* (1993) | Cesium 137 | M | 1.13 | 1.14 |
| Sacramento, California | Nolte and Associates (1998a) | Visual | H | 1.50 | 0.51 |
| Louisiana | Rybczyk *et al.* (2002) | Feldspar | H | 1.14 | 3.03 |
| Michigan | Kadlec (1997a) | Resurvey | H | 1.80 | 13.7 |

*Note:* L = TP < 0.1 mg/L; M = 0.1 < TP < 1.0; H = TP > 1.0 mg/L.

Hocking (1989a) estimated a 1.6 g P/m²·yr annual accretion rate for *Phragmites australis* in a nutrient-rich Australian setting. Klopatek (1978) estimated a 1.95 g P/m²·yr annual accretion rate for a *Schoenoplectus* (*Scirpus*) *fluviatilis* stand. Annual phosphorus storage in an alluvial swamp in Illinois reached nearly 3.6 g/m²·yr of phosphorus storage, mainly due to high inputs of sediments during flooded conditions (Mitsch *et al*., 1979).

The manner of accretion has sometimes been presumed to be sequential vertical layering (Kadlec and Walker, 1999; Rybczyk *et al*., 2002), but that view is likely oversimplified. At least two factors argue against simple layering: vertical mixing of the top soils and sediments (Robbins *et al*., 1999), and the injection of accreted root and rhizome residuals at several vertical positions in the root zone. Nonetheless, new residuals are deposited on the wetland soil surface, from various sources. The most easily visualized is the litterfall of macrophyte leaves, which results in top deposits of accreted material after decomposition. However, algal and bacterial processing which occurs on submersed leaves and stems results in litterfall and accretion of micro-detrital residuals.

### Sustainable Removal: Particulate Settling

Chemical precipitation and the settling of incoming particulate matter may be significant processes, especially in connection with crop runoff control. For example, in studies on five constructed wetlands treating agricultural runoff in Norway, phosphorus retention was measured to be 26–71 g/m²·yr (Braskerud, 2002b). Mineral PP formed the large majority of these removals. However, phosphorus removal for organic particulate matter has been observed to be considerably less effective (South Florida Water Management District, unpublished data for the Boney Marsh treatment system). The character of the particulate matter is a strong function of the hydrology and soils of the watershed that produces the runoff (Stuck *et al*., 2001). For peat-based farmlands in south Florida, the majority of runoff PP has been found to originate in the drainage canals as a result of biological growth. Conversely, the mineral soils of the Imperial Valley of California, and those of Norway, produce dense mineral particulates during rain storms.

For the example systems in Table 10.8, there is essentially no median difference between the removal of TP and PP. However, closer examination suggests that mineral solids (Brawley, California; Imperial, California; and Norway), and wastewater solids (Richmond, Australia; Byron Bay, Australia; and Benton, Kentucky), have preferential removals of PP.

### Vertical Phosphorus Movement

#### Macrophyte Mining

Rooted macrophytes extract nutrients from the soil (pore water), primarily the top 20–30 cm, but adventitious roots sometimes utilize nutrients taken from the bottom of the water column. The aboveground plant parts seasonally die and create litter, which ultimately decomposes to release some of the structural phosphorus back to the top layer of sediments and to the water column (Figure 10.11). The overall effect is the removal of phosphorus from underground storages, and redeposition of phosphorus on the top layer of sediments. This process has been termed *phosphorus mining*.

However, this upward transfer is counteracted by at least two very important processes: the transpiration phosphorus flux, and translocation of phosphorus from senescing leaves to the rhizomes.

#### The Transpiration Flux

Examination of Figure 10.5 reveals two very interesting features of vertical phosphorus transport potential: the phosphorus gradient is typically *decreasing* downward in the soil pore water, and the gradient of phosphorus concentrations is

**TABLE 10.8**
**Comparison of the Removals of Total Phosphorus (TP) and Particulate Phosphorus (PP) in FWS Wetlands**

| System | Cells | Years | Sedimentation Basin | TP In (mg/L) | TP Out (mg/L) | % Reduction | PP In (mg/L) | PP Out (mg/L) | % Reduction | References |
|---|---|---|---|---|---|---|---|---|---|---|
| Test Cell 8, Florida | All | 4 | N | 0.027 | 0.015 | 44 | 0.011 | 0.006 | 45 | Unpublished data |
| Boney Marsh, Florida | All | 6 | N | 0.054 | 0.021 | 62 | 0.017 | 0.011 | 36 | Unpublished data |
| Norway 5 CW | — | 3 | Y | 0.300 | 0.190 | 37 | 0.234 | 0.129 | 45 | Braskerud (2002b) |
| Tarrant, Texas | 1–2–3 | 4 | Y | 0.870 | 0.448 | 49 | 0.234 | 0.137 | 41 | Unpublished data |
| Tarrant, Texas | 4–5–6 | 4 | Y | 0.870 | 0.293 | 66 | 0.234 | 0.198 | 15 | Unpublished data |
| Tarrant, Texas | 7–8–9 | 4 | Y | 0.870 | 0.314 | 64 | 0.234 | 0.144 | 38 | Unpublished data |
| Byron Bay, Australia | All | 3 | N | 0.920 | 0.420 | 54 | 0.220 | 0.070 | 68 | Unpublished data |
| Imperial, California | All | 5 | Y | 1.297 | 0.812 | 37 | 0.440 | 0.120 | 73 | Unpublished data |
| Brawley, California | All | 5 | Y | 1.404 | 0.706 | 50 | 0.466 | 0.170 | 64 | Unpublished data |
| Benton, Kentucky | 1CT | 1 | N | 4.821 | 4.443 | 8 | 1.776 | 1.050 | 41 | NADB database (1998) |
| Benton, Kentucky | 2WG | 1 | N | 4.821 | 4.057 | 16 | 1.405 | 0.454 | 68 | NADB database (1998) |
| Richmond, Australia | Open Water | 2 | N | 9.650 | 9.380 | 3 | 1.550 | 1.410 | 9 | Bavor *et al.* (1988) |
| Richmond, Australia | Myriophyllum | 2 | N | 9.650 | 9.190 | 5 | 1.550 | 0.710 | 54 | Bavor *et al.* (1988) |
| **Median** | | | | | | **44** | | | **45** | |

typically *increasing* downward from the water column to the surface of the sediment-soil matrix. Thus, we would expect phosphorus exports from the soil to the overlying water if diffusion were the only operative factor, which is contrary to the overwhelmingly prevalent case of phosphorus removal from the water column. One obvious mechanism that can overcome this wrong-directional diffusional driving force is vertical downflow of water into the root zone.

### *Transpiration Flows*

Vertical flows of water in the upper soil horizon are driven by gravity and by plant uptake to support transpiration. In an aquatic

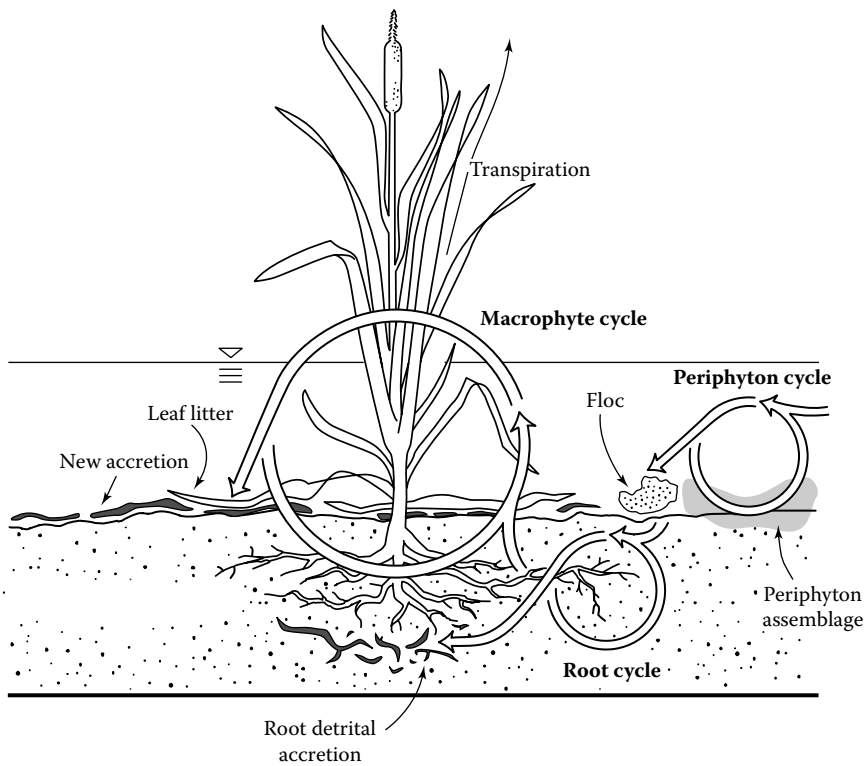



**FIGURE 10.11** The total biogeochemical cycle may be broken into loops representing, roots, shoots, and periphyton. The two macrophyte cycles interact via translocation and transpiration flows. The periphyton cycle draws phosphorus directly from the water column. (Adapted from Kadlec and Walker (1999). In *Phosphorus Biogeochemistry in Subtropical Ecosystems*. Lewis Publishers. Boca Raton, Florida, pp. 621–642.)

system, without emergent transpiring plant parts, vertical downflow will be driven solely by gravity, if there is infiltration out of the wetland system. Water infiltration flow is then computed from the water pressure (hydraulic head) gradient between the saturated soil surface and the receiving aquifer, multiplied by the hydraulic conductivity of the soil. This flow moves the phosphorus in the overlying water downward into the root zone.

In aquatic and wetland systems with fully saturated soils or free surface water, the meteorological energy budget requires the vaporization of an amount of water sufficient to balance solar radiation and convective losses (see Chapter 4). Some of this vaporization is from the water surface (evaporation); some is from the emergent plants (transpiration). Emergent plants "pump" water from the root zone to the leaves, from which water evaporates through stomata, which constitutes the transpiration loss. In a densely vegetated wetland, transpiration dominates the combined process, termed evapotranspiration (*ET*) (Kadlec *et al*., 1987). Water for transpiration must move through the soil to the roots. That movement is vertically downward from overlying waters in most wetland situations. In temperate climates, *ET* ranges from 60 to 200 cm/yr, but is concentrated in the part of the year with greatest solar radiation.

Thus, transpiration has the potential to move approximately 1 m/yr of water vertically downward to the root zone. That water carries with it the phosphorus concentration associated with the bottom layer of overlying water. In the case of Figure 10.5, that concentration is about 0.6 mg/L, and the vertical flux of phosphorus due to 1 m per year of transpiration is therefore 1.0 m/yr × 0.6 g/m$^3$ = 0.6 g P/m$^2$·yr. It is, however, not the case that the amount of phosphorus taken up by the plants is simply the water uptake times the prevailing phosphorus concentration in the root zone. Most researchers identify a transpiration stream concentration factor (TSCF), which is a multiplier on pore water concentration, and may be greater or less than unity (see, e.g., Kim *et al*., 2004).

### *Diffusion in the Soil Matrix*

The presence of the soil matrix prevents convection currents, and therefore the diffusive process is restricted to molecular diffusion in the porespace. The model for this process is the diffusion equation:

$$J_D = -\phi D \frac{dC}{dz} \tag{10.7}$$

where

$C$ = phosphorus concentration, g/m$^3$
$D$ = diffusion coefficient, m$^2$/d
$J_D$ = vertical diffusion flux, g/m$^2$·d
z = vertical distance, positive downward, m
$\phi$ = matrix blockage coefficient, dimensionless

The value of the diffusion coefficient in pure water is approximately 7.3 × 10$^{-5}$ m$^2$/d at 25°C, for $H_2PO_4^-$ which is the dominant inorganic form of phosphate at low to circumneutral pH (Lide, 1992). Values in the soil pore water are likely to be lower because of a tortuosity effect, which has been estimated to be about four times lower (Fisher and Reddy, 2001); and because of the matrix blockage coefficient, which is approximately 0.8. In general, concentration gradients are variable in both magnitude and direction.

Some idea of the expected magnitude of phosphorus diffusion may be gained by examining the situation of mildly eutrophic surface waters overlying a fully saturated peatland. In Figure 10.5, the pore water phosphorus gradient is −4.0 g/m$^3$·m in the top 20 cm. Under these circumstances, the diffusion flux predicted by Equation 10.7 becomes:

$$J_D = -\frac{(0.8)(7.3\times10^{-5})}{4}(-4)(365) = 0.02 \text{ g/m}^2\cdot\text{yr}$$

This estimated flux is insufficient to maintain plant growth, and is much less than the transpiration flux. It is concluded that the vertical movement of phosphorus into the root zone must be driven by vertical flow from the water column into the root zone, in order to overcome negative (upward from soil to overlying water) driving forces, and that diffusional transport in the soil is small in comparison.

### *Translocation*

During the late summer to early autumn period, phosphorus is in part translocated to belowground plant rhizomes. For instance, Tanner (2001a) found 36% of the end of season standing crop loss of phosphorus in *Schoenoplectus* (*Scirpus*) *tabernaemontani* translocated to rhizomes at light loading (0.20 g P/m$^2$·d), but only 13% at higher loading (0.90 g P/m$^2$·d). This is in contradiction to Mueleman *et al*. (2002), who found 58% translocated in a *Phragmites* wastewater wetland, compared to 25% in a natural *Phragmites* wetland. Hocking (1989a) found 45% translocation in a nutrient enriched *Phragmites* wetland. Smith *et al*. (1988) found 39% translocation in natural *Typha latifolia*. Mixed stands (*Phragmites*, *Schoenoplectus, Typha,* and *Scolochloa*) exhibited 29–36% translocation during a five-year study of ten low nutrient status marshes in Manitoba (Murkin *et al*., 2000). The central tendency of these results is the translocation of about one third of the phosphorus standing aboveground crop to belowground storage each autumn in temperate climates.

During the spring growth spurt, phosphorus may be withdrawn from storage in the belowground biomass. The amount has been estimated to be up to 100% of the early growth needs of the plant (Bernard, 1999). However, it seems probable that the total return cannot exceed the excess stored in autumn, else the plant system would be unstable. Models such as that of Wang and Mitsch (2000), which withdraw phosphorus from soils but have no mechanism to replace it, cannot accurately describe long-term sustainable phosphorus processing in soils.

## Soil Phosphorus Release

Many papers in the literature address what is called *Phosphorus Release Potential* for wetland soils (Fisher and Reddy, 2001; Pant and Reddy, 2001a; Malecki *et al*., 2004).

The general procedure is to expose soil cores to a layer of phosphorus-free water, and then to measure either or both the soil pore water phosphorus gradient and the time rate change of the phosphorus content of the overlying water. These are typically abiotic laboratory experiments, in which plants are absent. Therefore, there is no vertical water flow and the experimental durations are relatively short (on the order of weeks).

Such experiments do place a limit on how much labile phosphorus is in a particular soil, and therefore how much might be "rinsed out" by exposure to low phosphorus floodwaters. However, it is dangerous to extrapolate the results to full-scale field situations. One need look no further than what may be termed the "removal paradox." For instance, the results of Fisher and Reddy (2001) indicate phosphorus release along virtually the entire gradient of a *de facto* treatment wetland, WCA2A of the Florida Everglades, ranging from 0.47 to 1.83 g P/m$^2$·yr during ten days' exposure to 25 µg/L floodwater. However, as water flows through this wetland, phosphorus is stripped from the water. Entering concentrations are about 80 µg/L, and these decrease to 15 µg/L at downstream locations. Removals range from about 1.2 g P/m$^2$·yr near the inlet to about 0.2 g P/m$^2$·yr at downstream locations. So the apparent paradox is simply that the ecosystem is taking up phosphorus (and has for over 30 years), but the core experiments show release back to overlying water.

The correct interpretation of the results is that the soil core rinsing experiments help to understand what might happen if a treatment wetland began receiving clean (low phosphorus content) water. It is logical to expect that the principal liability for desorbable phosphorus is the loosely bound fraction, which is characterized by bicarbonate-extractable phosphorus. Unfortunately, most experiments have not run long enough to define the end of the leaching process, but there are clear indications that only a limited amount of the phosphorus storage is capable of release and back diffusion. Nguyen *et al.* (1997) determined that only 15% of the bicarbonate-extractable phosphorus was leached from sediments of a treatment wetland in short-term (hours) extractions with clean water. Speculatively, typical treatment wetland soils might give back something like one year's phosphorus storage if exposed to low phosphorus content water. This has in fact been observed (Kadlec and Bevis, 1990). Novak *et al.* (2004) studied soil profiles and the phosphorus mass balance for an in-stream wetland in North Carolina, and found both uptake and release at different times and places, which they attributed to hydroperiod and time-varying input phosphorus.

## Dry-Out and Rewetting

Continuous-flow FWS treatment wetlands maintain standing water over the entire year, which allows the maximal rates of accretion (see Table 10.7). This distinguishes them from natural wetlands and stormwater wetlands, which often have hydroperiods of less than 100%. A period of water loss can be a drawdown, in which overlying water is drained, but moist soil conditions continue to prevail; or it may be a dry-out, in which the soils become desiccated. Oxygen reaches the soil in either drawdown or dry-out, thus providing opportunities for rearrangement of the forms of phosphorus in the soil. Dry-out causes sediment oxidation and subsequent mobilization and export of sediment-bound pollutants. In some climatological regions, extended dry periods, coupled with seepage, can drop water levels in the permanent pool, and jeopardize the integrity of the wetland ecosystem. Oxidation may also cause a decrease in the sorption potential for phosphorus (Baldwin, 1996). There may be decreased phosphorus affinity due to the aging of the oxyhydroxides of iron (Watts, 2000).

Olila *et al.* (1997) performed laboratory core experiments on sediments from the Lake Apopka, Florida, treatment wetland, and showed that newly accreted floc materials underwent rapid mineralization during drawdown. The resulting inorganic phosphorus was released to floodwaters upon rewetting of the cores. Similar experiments in WCA2A of the Everglades showed that reflooding caused phosphorus releases intermediate between anoxic and aerobic conditions (Fisher and Reddy, 2001). The pulse of phosphorus is more rapid if the reflood occurs via underground flows (Corstnje and Reddy, 2004). When detrital material from STA1W C1, Florida, was subjected to dry-out and reflooding, only a few percent of the stored phosphorus was mineralized and mobilized (Pant and Reddy, 2001a).

The presence of an algal community in the desiccated material fosters a much more rapid release and recovery. Thomas *et al.* (2006) found that rewetting dry periphyton mats led to an initial release, for a period of about one day. However, regrowth of the biological communities led to reabsorption of that phosphorus in the ensuing few days.

There is evidence of the release after dry-out in the performance records of large treatment wetlands. Figure 10.12 shows the fifth year of operation for STA6, one of the phosphorus-removal treatment wetlands in south Florida. The very first waters to enter the system after the dry season produce a surge of phosphorus in the water, which declines as the wetland refills, prior to any outflow. This batch-filling mode produces treatment, and the mobilized phosphorus is essentially removed before outflow resumes about a month later. Similar results have been observed for the Orlando Easterly treatment wetland (T.A. DeBusk, personal communication).

Of course, the ultimate in oxidation of phosphorus storage in wetlands is the dry-out and burning of the soils and sediments. Such burns may be surficial, affecting only the top few centimeters of material; or deep burns, also called peat fires. Both types of fire have the effect of converting organic forms of phosphorus to mineral forms, in large proportions (Smith *et al.*, 2001). The mineral forms are conducive to resolubilization on rewetting, and potential export if there is flow through the wetland.

Accretion in some natural wetlands leads to the elevation of the wetland soil surface, which may in some circumstances reduce the hydroperiod of the wetland. If unchecked,

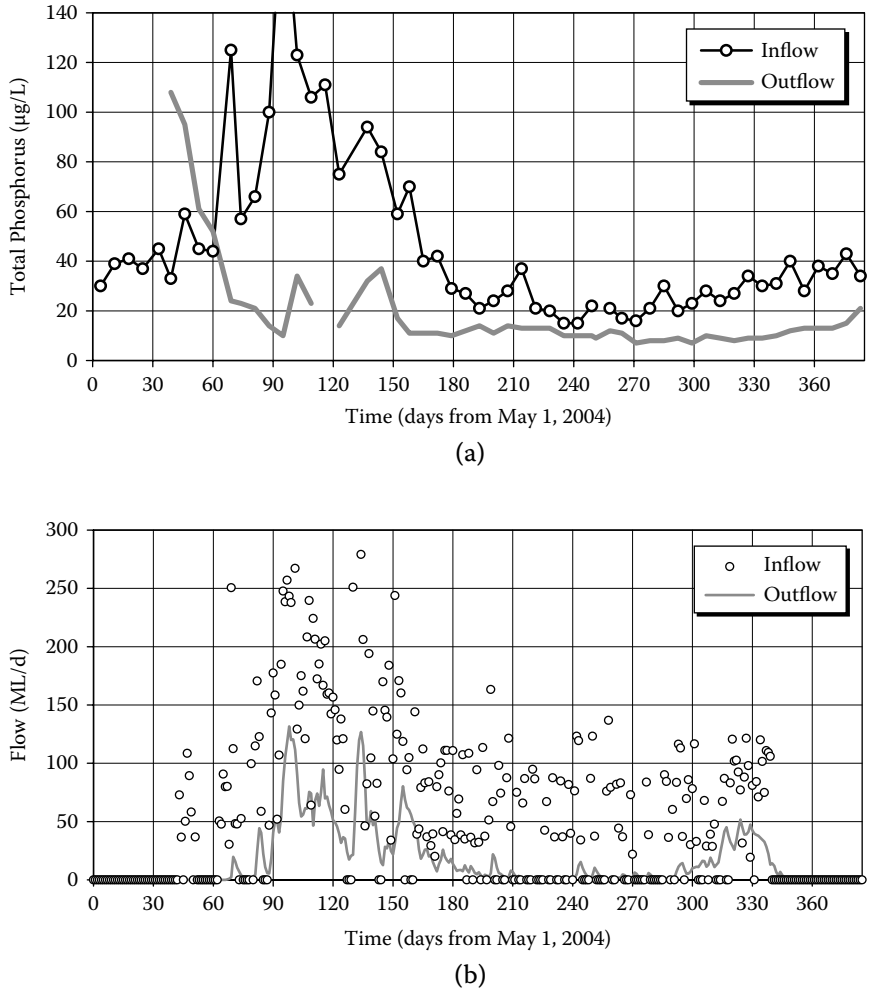



**FIGURE 10.12** The behavior of STA6, Florida, over a one-year period. There is little inflow and no outflow in the spring dry season, but inflows commence in June and rehydration is complete and outflows begin in July. The flow-weighted mean outflow concentration was 16.5 µg/L during the year, and the flow-weighted mean inflow concentration was 53.4 µg/L. Note that the rewetting water peaked at 110 µg/L, but rapidly subsided over a six-week period to below 20 µg/L. (Unpublished data from South Florida Water Management District.)

such accretion eventually can reach a stage in which the oxidation processes during the dry periods remove the accretions formed when the system is wet. This situation is one of *hydrologic burnout*, in which no further net phosphorus accretion occurs, and the system stores no more phosphorus. The wetland may then alternate between storage and release during the wet and dry seasons of the year. This phenomenon has created the impression that wetlands are sometimes phosphorus sinks and sometimes phosphorus sources. However, continuously wet treatment wetlands should not be included in such a generalization.

### Atmospheric Phosphorus Processes

There are atmospheric processes that involve phosphorus transfers in wetlands, but they are not normally of sufficient magnitude to warrant consideration in design. That may not always be the case, and such processes are briefly discussed here to alert practitioners to potential areas of concern in exceptional situations.

### Atmospheric Deposition

Atmospheric deposition of phosphorus contributes measurable quantities to receiving land areas. All forms are involved: particulate and dissolved, inorganic and organic, and wetfall and dryfall (Table 10.9). The phosphorus concentration of rainfall is variable depending on atmospheric conditions, air pollution, and geographical location. A typical range of concentrations associated with rainfall is 10–50 µg/L. These concentrations can be used with local rainfall amounts to estimate rainfall inputs in phosphorus mass balances (Table 10.9). Annual total atmospheric phosphorus loadings are 2–80 mg/$m^2$·yr. Consequently, atmospheric sources are almost always a negligible contribution to the wetland phosphorus budget for all but ombrotrophic, nontreatment wetlands.

**TABLE 10.9**
**Atmospheric Deposition of Phosphorus at Example Locations**

| Site | Rain (μg/L) | Total (μg/L) | Total (mg/m²·yr) | Dry (mg/m²·yr) | Reference |
|---|---|---|---|---|---|
| Wales | 10 | — | 17 | — | Neal *et al.* (2003b) |
| Crete | 4 | 13 | 2 | 3.9 | Markaki *et al.* (2003) |
| Erdemli | 8 | 24 | 5 | — | Markaki *et al.* (2003) |
| Madison, Wisconsin | 32 | — | — | 61.7 | Soranno *et al.* (1997) |
| Lake Erie | — | — | 14 | — | Baker and Richards (2002) |
| Slapy, Czechoslovakia | — | 31 | 15 | — | Kopacek *et al.* (1997) |
| Rimov, Czechoslovakia | — | 51 | 24 | — | Kopacek *et al.* (1997) |
| Eastern Washington | — | — | 7 | — | Newman (1995) |
| Northern Ontario | — | — | 33 | — | Newman (1995) |
| North Carolina | — | — | 49 | — | Newman (1995 |
| England | — | — | 80 | — | Newman (1995) |
| Southern France | — | — | 75 | — | Newman (1995) |
| Massachusetts | — | — | 4 | 0.1 | Hu *et al.* (1998) |
| South Florida | 9 | 37 | 41 | — | Ahn and James (2001) |
| **Median** | | **31** | **17** | | |

## Phosphorus Volatilization

Gaseous forms of phosphorus, phosphine ($PH_3$), and diphosphine ($P_2H_4$) are known to exist. These are in some sense analogous to hydrogen sulfide, in that they form in low redox conditions, are volatile, and are extremely toxic. Phosphine is soluble in water, but has a high vapor pressure. It may be emitted from regions of extremely low redox potential, together with methane. It has been reported that the methane–phosphine mixture emitted from marshes and bogs can auto-ignite, forming the flickering lights known as "Will-o'-the-wisp." In the early 20th century, there were reports of failure to find these compounds, which are now thought to be the result of flawed analytical procedures. Burford and Bremner (1972) found that formation of phosphine could not be excluded on thermodynamic grounds. However, their attempts to measure $PH_3$ emission failed, which they attributed to strong sorption of the $PH_3$ on the soil.

With better analytical techniques, Devai *et al.* (1988) were able to detect emissions of phosphine from Imhoff tanks and from reed/bulrush wastewater treatment wetlands. The wetland in question was found to emit 1.7 g P/m²·yr. The concept of measurable biological $PH_3$ emissions was boosted by the report of Gassmann and Glindemann (1993), who found half-lives of 5–28 hours in air. Because of the strong sorption of phosphine on aquatic and wetland sediments, Gassmann and Schorn (1993) looked for phosphine in the "black muds" of the Hamburg, Germany, harbor. They found 26–57 ng $PH_3$/kg wet sediment, and concluded that biogenic phosphine was prevalent in nature under low redox conditions.

Spurred by the continuing controversy over the existence and importance of phosphine in the biogeochemical cycle of phosphorus, Devai and Delaune (1995) measured 16 pg $PH_3$/kg·d wet sediment for Everglades soils at 73 mg P/kg wet soil. However, when the Everglades, Florida, soil was amended to 2,200 mg P/kg wet soil, the $PH_3$ emission rate increased to 380 pg $PH_3$/kg·d wet sediment. For the top 30 cm of the Everglades peat, this is a loss of 38 mg P/m²·yr, which is about double the median rate of atmospheric deposition. Devai *et al.* (1999) measured rates of emission from sewage sludge up to 268 pg $PH_3$/g·d wet sediment, or 700 times the rate for Everglades soils. Most of the phosphine in sediments is matrix-bound. Han *et al.* (2000) also found that most $PH_3$ in rice paddy fields was matrix bound, and that emission rates averaged 16 μg P/m²·yr.

Phosphine has been implicated in the phosphorus cycle of contaminated lakes (Niu *et al.*, 2004). Concentrations in lake Taihu, China, were measured in the range of 0.37–0.40 pg/L for dissolved $PH_3$, but were 5–9 times higher for unfiltered samples. The source of phosphine was inferred to be the sediments, in which concentrations up to 0.92 μg/kg were measured.

Landfills are a potential low-level source of phosphine emission. Roels and Verstraete (2004) measured gaseous concentrations, and estimated a loss rate of about 0.5 g P/m²·yr from a 35-ha landfill in Belgium. The source of the phosphine was identified as biogenic corrosion of iron. Further work (Roels *et al.*, 2004) found no evidence of microbial conversion of phosphate to phosphine.

To date, all North American wetland phosphorus mass balance studies have ignored the possibility of a significant phosphine production or loss. The importance of $PH_3$ remains

to be defined, but there are now several research teams working to improve understanding.

## 10.4 SPATIAL AND TEMPORAL PHOSPHORUS EFFECTS IN FWS WETLANDS

This section is devoted to the examination of internal gradients of phosphorus concentrations in various portions of the wetland, and to the transient effects that accompany the changing storages of phosphorus within a FWS wetland system.

### Wetland Start-up

Observed wetland phosphorus removal performance may be influenced by transition from past history. Constructed treatment wetlands start from any one of several initial conditions, characterized by soil type, as well as vegetation type and density. These newly created wetlands are then exposed to a hydroperiod that is typically 100%, created by some form of wastewater. Under such conditions, it is obvious that some period of adaptation will ensue, during which soils and vegetation undergo changes dictated by ambient conditions of water depth and nutrient availability. For instance, FWS wetlands built on agricultural soils may have to deal with large stores of soil phosphorus that may temporarily result in high concentrations in floodwaters. For example, the adaptation period for the STA-1W FWS wetland in Florida extended over about 1½ years, during which time it was held in a no-discharge mode to accommodate the transition (Figure 10.13). This wetland was built on a former agricultural field, which apparently contained mobile legacy phosphorus.

In contrast, the Brawley, California, treatment wetland was built on the floodplain of the New River, on land that was not previously used for agriculture. Water from the New River was pumped to the wetland, and contained about 1.4 mg/L of TP. The wetland was built in the last half of 2000, with the planting occurring in mid-August. The transplants were healthy and robust by January 2001, but vegetative fill-in and establishment had just begun. Water quality monitoring commenced in January 2001. The wetland was operated with much higher flows in springtime, resulting in the cyclic pattern of removals seen in Figure 10.14. However, there was no trend in removal percentage with time on average over the first five years of the project life (linear model has $R^2 = 0$). The loading to this wetland (about 50 g P/m$^2$·yr) was far beyond that which could have been influenced by macrophyte establishment. Speculatively, the soils were already loaded with phosphorus matching the river water content (estimated $EPC_o$ = 1.4 mg/L), and no further start-up sorption was likely. This wetland therefore had no particular start-up transient.

Antecedent conditions also often influence current performance for existing, natural wetlands receiving new phosphorus inputs. In such a case, the wetland may initially display a larger removal capacity than after stabilization. The Houghton Lake, Michigan, natural treatment wetland began receiving treatment lagoon effluent in July 1978, at relatively low PLR, about 4 g P/m$^2$·yr. The adaptation of the wetland proceeded from a fully established, low-nutrient community to a nutrient-enriched replacement community. New, very high-standing crops of cattail (*Typha* spp.) incorporated more phosphorus in an area that expanded over an initial eight-year period (Kadlec, 1997a). During this period, phosphorus removal was enhanced by vegetative expansion.

Thus, it is seen that the initial period of performance of an FWS treatment wetland may exhibit lesser or greater phosphorus removal than in the long run. The deciding factors are the antecedent phosphorus loading of the soils and the importance of the vegetative cycle in comparison to the applied phosphorus loading.

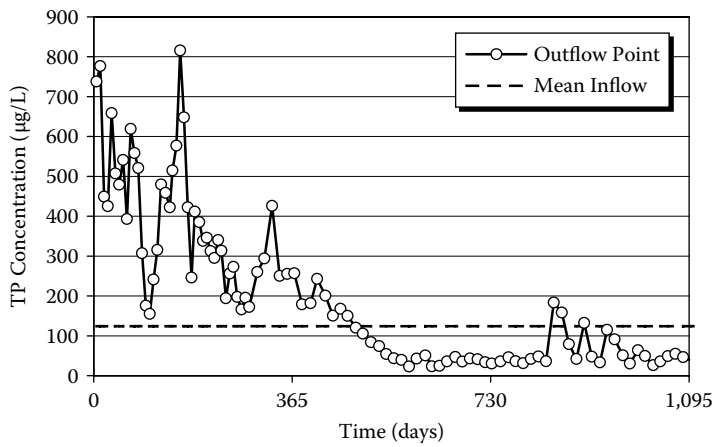



**FIGURE 10.13** Floodwater phosphorus during the start-up of STA-1W Cell 5 of the Everglades Protect project. Antecedent land use was agriculture. (Unpublished data from South Florida Water Management District.)

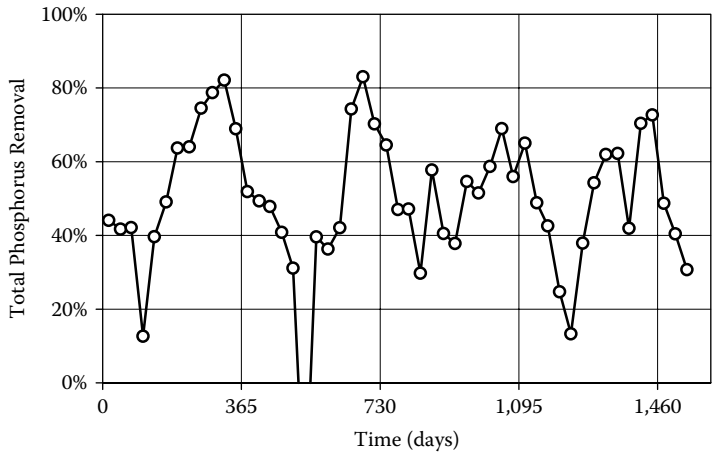



**FIGURE 10.14** Phosphorus removal at the Brawley, California, wetland. There is a cyclic trend that is driven by an annual cycle in the flow to the wetland, but no linear trend up or down. Planting occurred about 120 days prior to the data record, which begins on January 1, 2001.

## Phosphorus Gradients in FWS Treatment Wetlands

As for other pollutants that are reduced in treatment wetlands, there typically exists a decreasing profile of water concentrations from inlet to outlet. However, one must be very careful about interpretation of measurements along the direction of flow. It is well known that at any specific distance from the inflow point, there is a distribution of flows and concentrations across the width of the system (perpendicular to the flow direction). Further, it is difficult to know just what depth ought to be sampled, and internal wetland sampling is an art. Multiple sampling points across the wetland width are much better than a single point, but there is no guarantee that flow proportional samples will result. Such cross-gradient protocols provide "through the wall" sampling averages instead. Those spatial averages are quite different from flow-weighted concentrations, by as much as a factor of 2 (Kadlec, 1999d). Dierberg *et al.* (2005) sampled both slow and fast paths through a wetland, and found greatly different phosphorus gradients in the flow direction. The true flow-weighted mean gradient is obtained if all the water is collected at various points along the flow path, and a "mixing cup" concentration is obtained. This is achieved if the wetland is compartmentalized into several cells in the flow direction.

An example of such a compartmentalized sampling is typified by the Lakeland, Florida, system, which is divided into seven cells. Several of these have been tracer tested, and found to have differing degrees of short-circuiting (Keller and Bays, 2000). Nevertheless, the effect of compartmentalization is to provide the equivalent of plug flow. This treatment wetland is not efficient in phosphorus reduction, because it is built in an old phosphorus mine pit. There is a steady but gradual reduction in TP as water moves through the cells (Figure 10.15).

At very low phosphorus concentrations, there is evidence that TP cannot be reduced below a wetland background concentration ($C^*$) (Figure 10.16). Internal cycling plus atmospheric gains and losses of water and phosphorus combine to create such a background concentration. The profiles in Figure 10.16 result from transect sampling, and do not reflect

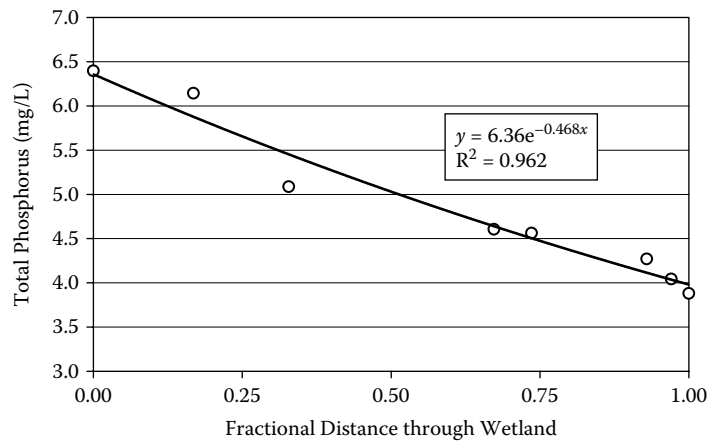



**FIGURE 10.15** Profile of phosphorus concentrations through the Lakeland, Florida, FWS treatment wetland. Data are from 13 years of operation. (Unpublished data from city of Lakeland.)

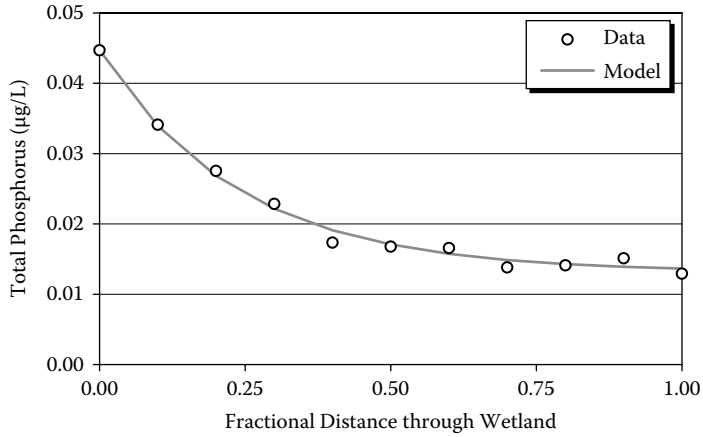



**FIGURE 10.16** The gradient of TP through the Boney Marsh FWS constructed treatment wetland. Data are the averages for two parallel transects in the flow direction, each sampled 13 times over 1978–1979. The model line is for $P = 12$, $k = 29$ m/yr, and $C^* = 12$ µg/L. (Unpublished data from South Florida Water Management District.)

flow-weighted averages. The mixing-cup inlet concentration for the period was 49 µg/L, in contrast to the interior inlet sample average of 44 µg/L. The mixing-cup outlet concentration for the period was 16 µg/L, in contrast to the interior outlet sample average of 13 µg/L.

The existence of phosphorus gradients within treatment wetlands means that it is not appropriate to use models that are configured as one well-mixed unit. *The existence of a low background concentration means it is not appropriate to use first-order models that carry the predicted concentrations down to zero.*

## Phosphorus Movement in FWS Wetlands: Fronts and Spiraling

Most FWS wetlands are flow through systems in which phosphorus is preferentially removed in the upstream sections or compartments. It has been shown above that there exist several mechanisms and storages that act to remove phosphorus, and that some of these have finite capacity. These effects combine to produce transitory, moving "fronts" along the flow direction, and these move at different speeds. Nevertheless, frontal movements do not last forever, because there exist mechanisms that cause their eventual cessation. It is important to understand something about the approximate time scales of these processes. We will here presume that the FWS wetland is newly constructed, and does not bear any legacy phosphorus.

Water moves through treatment wetlands in a matter of days. During that time frame, removal processes will be relatively stable, and variations will typically be due to random effects within the ecosystem. At any particular moment, there is almost certain to be a decreasing profile of water-phase phosphorus along the flow direction. This is the *water column phosphorus front.*

The front end of the system sees the most phosphorus. If phosphorus is the growth-limiting nutrient, a plant size gradient will likely occur. This and other growth-related phenomena are likely to last for only a year or two. During that same period, the sorption sites on the pre-existing soils will become fully occupied, and no further phosphorus removal will occur due to either sorption or plant establishment. However, the system will continue to form floc and new sediments, and phosphorus removal will continue at a reduced rate compared to the plant establishment phase.

The top layers of soils and sediments therefore build new storages of phosphorus, which are also allocated preferentially to the inlet region of the wetland. The system is building floc and soil phosphorus fronts, which are not coincidental with the water column phosphorus front. The establishment of a *soil phosphorus front* usually takes a matter of years, during which the vertical front of soil phosphorus profiles creeps upward due to soil building. Plant root zones can move to keep pace with this upward movement. Floc forms much more readily, and can move at speeds intermediate to the water speed and the soil phosphorus front. The rooted macrophytes respond to soil phosphorus, typically by the establishment of species that can better utilize the increased nutrient supply—unless the system was established with those more nutrient tolerant varieties. Species replacement lags the soil phosphorus front, and may take many years to develop and stabilize, although colonization may occur more rapidly. This results in a *vegetation replacement front.*

Supposing a stable and continuing input of water and phosphorus, the whole set of changes stops when a new set of vegetation has developed, and soil building is removing a fixed amount of phosphorus per unit time on average. The balance of phosphorus will then be exported, as a combination of continual outflows in the water, plus episodic (perhaps) outflows of floc, total suspended solids (TSS), or detritus.

However, even then the system will continue to evolve on the time scale of decades, because accretion (about 1 cm/year)

will eventually change the flow patterns and depths in the FWS wetland.

Although only a few projects have had the longevity to track these various fronts in full-scale situations, they have confirmed the concepts outlined here (DeBusk *et al*., 2001; Rutchey, 2006).

Apart from accretion, wetland solids form a large pool of phosphorus, some of which is available for exchange with surface waters and pore waters. As noted above, sorption and ion exchange are active processes in the wetland environment. These solid storages will stabilize under continuous operation of a treatment wetland, but are nonetheless active, and exchange compounds with their surroundings. Thus, the image of phosphorus compounds traveling with the flowing water is incorrect; phosphorus follows more of a "park and go" trajectory through the wetland.

### Short-Term Anomalies

There are transient effects related to short-term experiments on treatment wetlands. These transient events are different from the stable annual pattern of swelling and shrinking of the phytomass phosphorus storage. Results from transient studies must not be construed as being representative of long-term patterns. A case study is informative. Busnardo *et al.* (1992) operated FWS mesocosms vegetated with *Schoenoplectus* (*Scirpus*) *californicus*. The PLR to the mesocosms were 55 and 88 g P/m$^2$·yr for two consecutive seven-month periods. Approximately 52% of the phosphorus removed was found in plant growth. Although this experiment demonstrated that emergent macrophytes have the capacity to assimilate large quantities of phosphorus, Busnardo *et al.* (1992) speculated that plants would have a lesser effect in mature wetlands.

Other investigators have also compared species for phosphorus uptake in short-term experiments. For instance, Tanner (1996) examined eight species over a 124-day growth period. The phosphorus uptakes were variable, but very large. For instance, *Schoenoplectus validus* took up phosphorus at the rate of 11.8 g P/m$^2$·yr; *Glyceria maxima* 38.3 g P/m$^2$·yr; and *Zizania latifolia* 54.5 g P/m$^2$·yr. These values are enormous compared to sustainable growth requirements, and were not intended for the purpose of extrapolating long-term removal rates. There is a tempting suggestion for harvest contained in such numbers, but such studies must not be used as a removal rate basis for design of FWS wetlands.

## 10.5 PHOSPHORUS REMOVAL IN FWS WETLANDS

The performance of a treatment wetland for phosphorus removal is partly deterministic, describable by central tendency equations, and partly stochastic, describable by the distributions of random effects (Kadlec, 1997b). Both pieces of the description are important, and both may be well-quantified. Alternate approaches for the deterministic part involve loading graphs and rate constants, and both contribute to design in different ways. Phosphorus processing is also constrained by lower limits to the achievable TP concentration.

### Background Concentrations of Total Phosphorus

Constructed and natural treatment wetlands typically provide reductions in added chemicals of various types. Wetlands are most often configured as flow through systems, with water entering at one end and exiting at the opposite end. Along the flow path, there is typically an exponential decrease in concentration, from the inlet value to a lower nonzero exit value (Kadlec and Knight, 1996). If the wetland is large compared to the added water flow and chemical load, the concentration displays a plateau towards the outlet of the system, along which further reductions do not take place (Figure 10.16). The concentration along the plateau is the treatment wetland background concentration ($C^*$). The portion of the wetland that fully contains the gradient is the zone of total containment (ZTC). The gradientless region is unimpacted by the discharge in question, but may exhibit the effects of other current or previous impacts.

Background concentrations also exist for pristine wetlands that do not incur anthropogenic water and chemical inputs, although these are arguably rare in light of global atmospheric transport. Phosphorus cycling in the natural environment leads to low concentrations in surface waters, and it cannot be expected that treatment wetlands will reduce phosphorus below those ambient levels. The phosphorus concentrations in such unimpacted wetlands are also considered to be background levels. The median flow-weighted TP concentration in 85 relatively undeveloped basins of the United States is 0.022 mg/L (Clark *et al*., 2000). Areas of the Rocky Mountains and central plains generally had the highest background, up to 0.10 mg/L. The northeastern and southeastern regions typically have lower ambient background phosphorus concentrations, about 0.01 mg/L. Levels are very low in the Florida Everglades, often in the range of 0.006–0.010 mg/L.

### Mechanisms Creating Background Concentrations

In the treatment wetland situation, there are several possible reasons for the existence of a real or apparent nonzero background concentration for phosphorus. First, there may be some portion of the incoming phosphorus that is resistant to storage or conversion in the wetland environment. Some portion of the organic phosphorus may be highly resistant to uptake by the biogeochemical cycle. An extreme example would be an organophosphate pesticide, which is not readily degraded in wetlands. However, more benign sources may contain a biologically unavailable fraction, by virtue of the size and character of the molecules embodying the phosphorus (Proctor *et al*., 1999). Such phosphorus fractions may pass through the system untouched.

A second reason for a nonzero background concentration for TP is the association with particulates. Nonreactive particulate phosphorus (NRPP) is part of the total water column phosphorus (Lantske *et al*., 1999), and is an important part

of the phosphorus fractionation of wetland sediments (Reddy *et al*., 1993). Because phosphorus is associated with suspended particulate matter (TSS = total suspended solids), a nonzero background level of TSS entails a nonzero background level of phosphorus. For instance, at the Des Plaines, Illinois, wetlands, the export of 8 mg/L of TSS carried 16 µg/L of phosphorus, which is due to a phosphorus content of 0.2% in the exported TSS. Although TSS is notoriously difficult to measure inside wetlands, background levels of 5–10 mg/L are commonly found in densely vegetated systems (Kadlec and Knight, 1996).

In FWS wetlands with more open water surface, the presence of phytoplankton in the water column is enhanced because of high light levels. The presence of these organisms implies an accompanying suspended PP concentration. Classic examples of planktonic phosphorus for wetlands are Muskego Lake in Wisconsin (R.A. Smith & Associates, 1995) and the Lake Apopka Treatment Marsh in Florida (Coveney *et al*., 2001). Muskego Lake is about one meter deep, with a central area of dense SAV and a broad cattail fringe. Phosphorus loadings come from a mostly agricultural watershed, and internal loadings from storages created by discontinued sewage discharges. Central portions reach 600 µg/L TP, and a typical export concentration is 100 µg/L TP, which is nearly 100% particulate (planktonic). Lake Apopka contains about 175 µg/L TP, 95% of which is planktonic particulate. This water is passed through a treatment marsh, which accretes PP, but converts a small portion to dissolved phosphorus. The marsh background is about 60–90 µg/L TP.

A third reason for a nonzero background concentration is a set of wetland processes that provide inputs distributed across the entire areal extent of the system. Groundwater discharge and rainfall may bring phosphorus into all portions of a wetland (Raisin *et al*., 1999). Phosphorus may be utilized in the biogeochemical cycle, which is also distributed across the entire wetland area. That same cycle can produce return of the substance to the water column, usually by the processes of decomposition and leaching (Kadlec, 1997a). Two examples of rainfall-driven systems are the Houghton Lake natural wetland, with a background of 40 µg/L TP (Kadlec, 1997a), and the Loxahatchee National Wildlife Refuge in south Florida, with a background of 10 µg/L TP (McCormick *et al*., 1997). Groundwater effects are illustrated in the study of Reid's Wetland in northeastern Victoria, Australia (Raisin *et al*., 1999). Ninety percent of the water input was groundwater discharge, at about 80 µg/L. The resulting phosphorus concentration in the surface discharge was 63 µg/L.

A fourth factor influencing concentration gradients, and the possibility of plateaus, is hydraulic bypass of the reactive wetland environment. Bypassed water carries with it the inlet substances, which may reblend with treated water at downstream wetland locations (Kadlec, 2000). This process cannot create a true plateau or background, but may easily lead to an inferred background concentration, derived from extrapolation of gradients in the upstream portion of the system. For some chemicals, notably phosphorus, very few treatment wetlands extend beyond the ZTC, and such extrapolation, via curve fitting, is the norm rather than the exception. As shown in Kadlec (2000), the nature of internal flow patterns leads to a data-fitted background concentration, which varies strongly with the hydraulic loading rate (HLR) to the wetland. Higher loading rates lead to higher background concentrations for flow through wetlands.

### Loading Considerations

The FWS intersystem database that provides the basis for comparison includes information from 282 wetland cells, including 42 mesocosms. The remaining 240 wetland cells represent many phosphorus removal applications, at widely varying concentrations, in a variety of climatic zones in North America and several other countries. One hundred of these wetlands are documented in the NADB version 2 (NADB database, 1998) and the remaining 140 are documented in literature reports and project files. Some of the data sets are more reliable than others, in terms of the length of the period of record, frequency of sampling, and degree of hydraulic detail. Twenty-eight percent of these (79/282) have been tracer tested to determine hydraulic efficiency (number of tanks-in-series, *N*TIS).

Full-scale FWS treatment wetlands with phosphorus removal data span five orders of magnitude in size, from about 0.01 to 2,000 ha (see Table 10.10 for examples). They are loaded from less than 1 to over 10 cm/d. Inlet phosphorus concentrations range from less than 50 µg P/L to over 8,000 µg P/L. Because of the variability present in any of these data sets, it is necessary to utilize comparisons in terms of the primary forcing variables and model parameters.

The phosphorus concentration produced in FWS wetlands depends upon three primary variables (area, water flow, and inlet concentration), as well as numerous secondary variables (vegetation type, internal hydraulics, depth, event patterns, and others). It is presumed that the area effect may be combined with flow as the HLR because two side-by-side wetlands with double the flow should produce the same result as one wetland. Therefore, two primary variables are often considered: HLR and inlet concentration ($C_i$). Both mass removal models (e.g., the *k-C** model) and performance regressions are based upon these two variables (Kadlec and Knight, 1996).

An equivalent approach is to rearrange the primary variables, without loss of generality, by using phosphorus loading rate (PLR = HLR × $C_i$) and concentration ($C_i$). Thus, it is expected that the phosphorus concentration produced ($C_o$) will depend upon PLR and $C_i$. A graphical display has often been adopted in the literature (Kadlec and Knight, 1996; U.S. EPA, 2000a; Wallace and Knight, 2006). In the broad context, multiple data sets are represented by trends that show decreasing $C_o$ with decreasing PLR, with a different trend line associated with each inlet concentration (Figure 10.17). For any specific inlet concentration, or a narrow inlet concentration range, the slope of the data cloud is about 0.33 (Figure 10.17), but the resultant outlet concentration range moves upward to higher values. The right-hand asymptote of

**TABLE 10.10**
**Examples of Phosphorus Treatment Wetlands**

| Site | Location | Cells Monitored | POR | Area (ha) | HLR (cm/d) | TP In ($g/m^3$) | TP Out ($g/m^3$) | P Load ($g/m^2 \cdot yr$) | Reference |
|---|---|---|---|---|---|---|---|---|---|
| Warangal | India | All | 2001 | 0.012 | 4.28 | 7.375 | 1.742 | 115.21 | Jayakumar and Dandigi (2002) |
| Listowel 4 | Ontario | System 4 | 1980–84 | 0.13 | 1.95 | 3.165 | 0.624 | 22.57 | Herskowitz (1986) |
| Linköping | Sweden | All | 1994–98 | 0.3 | 4.00 | 0.281 | 0.078 | 4.11 | Unpublished data |
| Sacramento | California | 7 | 1994–97 | 0.59 | 6.32 | 2.438 | 2.287 | 56.20 | Nolte and Associates (1998a) |
| Tarrant | Texas | WC1 | 1993–96 | 0.75 | 8.88 | 0.870 | 0.576 | 28.20 | TCWCID (1995) |
| Tres Rios | Arizona | H1 | 1995–02 | 1.34 | 9.31 | 3.090 | 3.350 | 105.00 | Unpublished data |
| River Hebert | Nova Scotia | All | 1996–98 | 1.9 | 0.89 | 3.210 | 0.550 | 10.43 | Hanson (2002a) |
| Des Plaines | Illinois | 3 | 1989–92 | 2.39 | 7.00 | 0.103 | 0.035 | 2.59 | Hey *et al.* (1994) |
| Brighton | Ontario | All | 2000–02 | 6.2 | 4.52 | 0.525 | 0.265 | 8.67 | Unpublished data |
| Byron Bay | Australia | All | 1990–93 | 10.06 | 1.39 | 0.667 | 0.364 | 3.54 | Unpublished data |
| Boney Marsh | Florida | West | 1976–87 | 48.6 | 2.01 | 0.052 | 0.021 | 0.38 | Moustafa *et al*. (1996) |
| Apopka | Florida | All | 1990–97 | 73 | 10.96 | 0.175 | 0.090 | 7.00 | Coveney *et al.* (2001) |
| Orange County | Florida | 1–3 | 1988–96 | 88.7 | 0.75 | 0.110 | 0.070 | 0.30 | Schwartz *et al.* (1994) |
| OEW | Florida | Stratum 1 | 1988–00 | 95.5 | 5.61 | 0.280 | 0.106 | 3.08 | Unpublished data |
| Beaumont | Texas | All | 1993–95 | 263 | 3.14 | 6.080 | 2.320 | 69.68 | Unpublished data |
| Lakeland | Florida | All | 1989–93 | 498 | 0.52 | 8.365 | 4.181 | 15.89 | Unpublished data |
| ENRP | Florida | All | 1994–99 | 1,545 | 2.61 | 0.104 | 0.021 | 0.99 | Unpublished data |
| Kis Balaton | Hungary | All | 1985–90 | 1,800 | 3.02 | 0.499 | 0.232 | 5.50 | Pomogyi (1993) |

*Note:* Sizes span 5 orders of magnitude, while loadings and concentrations span 2 orders of magnitude.

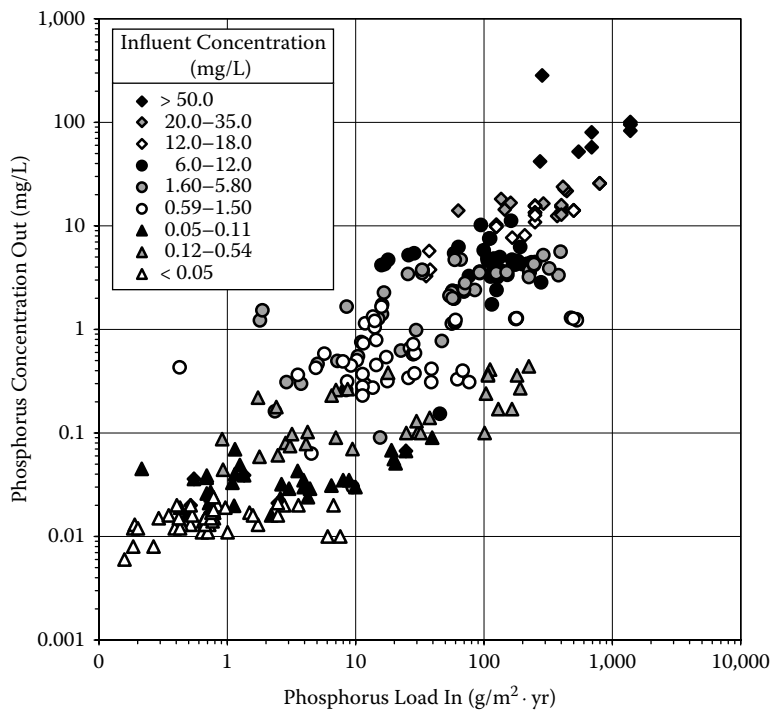



**FIGURE 10.17** Intersystem phosphorus responses for FWS treatment marshes. Each point represents the entire period of record for one wetland cell. Data span four orders of magnitude in both inlet and outlet concentrations and phosphorus loading, for the entire period of record for 282 individual systems. Data groups are coded in nine ranges to illustrate the effect of inlet concentration on the trends.

each data group, at very high phosphorus loading, is an outlet concentration equal to the inlet concentration—or in other words, no removal. *The fact that there exist data clusters for each inlet range indicates that there are at least two major factors influencing outlet phosphorus concentrations: inlet concentration ($C_i$) and PLR.*

If the entire set of points in Figure 10.17 is considered, thus ignoring the effect of inlet concentration, the general trend line has a slope of about 1.0. However, such a single variable plot is nonunique, and may be misleading. For instance, use of a small intersystem data set might result in use of left data points for high $C_i$, as well as right data points for low $C_i$, thus exaggerating the slope. Consequently, the $C_o$–PLR graph advocated in some literature (Richardson *et al*., 1997; U.S. EPA, 2000a) is inadequate. The *k-C** model spans the entire cloud of results shown in Figure 10.17, when exercised for various choices of $TP_i$, $k$, and $C^*$ (Kadlec, 1999c).

## Rates and Rate Constants

Despite the apparent complexity of the several removal mechanisms, data analysis shows that relatively simple equations can describe sustainable phosphorus removal processes. Profiles of phosphorus concentrations within wetlands typically show a decreasing trend that approaches the background concentration asymptote if the wetland is large enough. Global first-order removal rates characterize such removal profiles, but do not incorporate any biotic features.

The *k-C** model (Kadlec and Knight, 1996) embodies a first order, areal net uptake of phosphorus, coupled to a zero-order return flux from the static compartments of the wetland (soils, litter, biota), and is constrained to be applied to long-term average wetland performance. It is a two-parameter modified version of the first-order areal model, containing a lower limit on achievable phosphorus concentrations ($C^*$), proposed by Kadlec and Knight (1996). The long-term average phosphorus removal, locally variable, is calculated from:

$$J_{net} = k(C - C^*) \tag{10.8}$$

where

$C$ = phosphorus concentration, g P/L
$C^*$ = background phosphorus concentration, g P/L
$J_{net}$ = net phosphorus uptake, g P/m$^2$·d
$k$ = apparent rate constant, m/yr

This allows $C^*$ as a calibration parameter, with typical values in the 4–40 µg P/L range for FWS wetlands.

This rate is then combined with the appropriate water mass balance and internal mixing model to compute the performance of the entire wetland (Kadlec, 2003a), as previously discussed in Chapter 6:

$$\frac{C - C^*}{C_i - C^*} = \left(1 + \frac{k}{Pq}\right)^{-P} \tag{10.9}$$

where

$P$ = mixing and speciation parameter, dimensionless
$q$ = hydraulic loading rate, m/yr

In addition to describing intrasystem data (see Figure 10.16), this very simple model does a credible job of relating intersystem data sets. This is illustrated in Figure 10.18, in which two ranges of inlet concentrations are explored (0.1–0.6 and 1.0–6.0 mg/L). Performances of wetlands from the larger performance database were extracted, yielding 34 wetlands

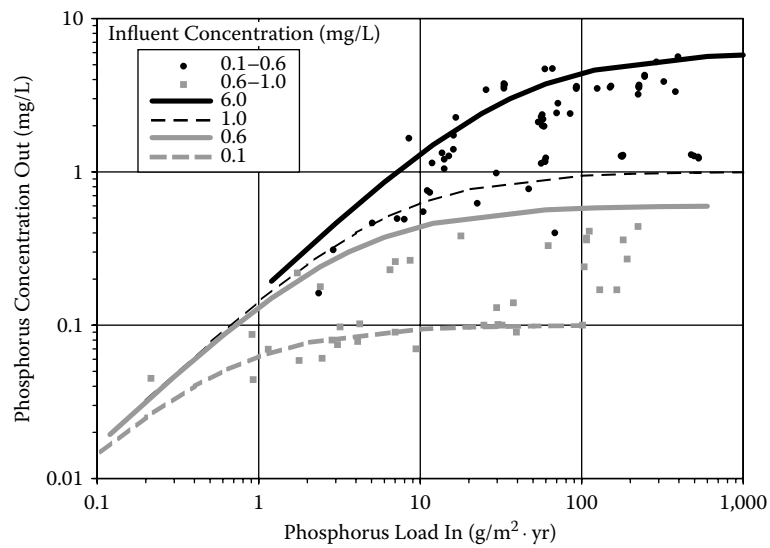



**FIGURE 10.18** Outlet concentration response to inlet loading for two ranges of inlet concentrations. Symbols represent life history data for FWS treatment wetlands, N = 34 for the 0.1–0.6 mg/L inlet range and N = 69 for the 1–6 mg/L inlet range. Lines represent bounding model calculations for $k$ = 6 m/yr, $C^*$ = 0, and $P$ = 1. Data sources include published papers, the North American Database (1998), and project reports.

in the lower range and 69 wetlands in the higher range. Equation 10.9 was then roughly calibrated, by matching the four model curves to the boundaries of the two data groups (Figure 10.18). Although lacking precision, this three-parameter ($k$, $P$, $C^*$) model goes a long way toward grouping variables to quantify performance.

More complex models have been formulated to provide performance forecasting. However, most of these have proven too cumbersome to calibrate or exercise in support of design explorations. Examples are discussed later in this chapter.

## Annual Rate Constants

Results across systems are given here, for measured $P$-values, or for the mean of those ($P = 3.4$) when not measured. The value $C^* = 0.002$ mg/L is used, and the remaining model parameter is the $k$-value, selected to fit Equation 10.9. The median annual rate constant was $k = 10.0$ m/yr (Table 10.11). The $10^{th}$–$90^{th}$ percentile range is 1.4–60 m/yr. There is sometimes a significant seasonal dependence of phosphorus $k$-values. Even on an average annual basis, temperature or climate may be an important determinant of the rate constant, and these factors are thus responsible for the some of the intersystem variability in annual $k$-values. Accordingly, it is necessary to examine intra-annual effects.

## Effects of Temperature and Season

Two factors are likely candidates for modifying phosphorus uptake over the course of a year: temperature and vegetation growth patterns. Water temperature would be expected to modify microbial processes, which are involved in phosphorus uptake and in phosphorus release during the decomposition of detritus from bacteria and algae, and from above- and belowground plant parts. Plant and algal growth requires substantial amounts of phosphorus, which is withdrawn from pore waters during the annual growth period. Warm-climate systems differ markedly from temperate wetlands, because the latter have a period of plant dormancy occasioned by winter conditions. Warm climate systems may also have periods of relative low vegetative activity, but they do not experience the complete growth stoppage seen in temperate systems.

It is preferable to utilize data from several replicated annual periods to establish trends that are valid against the background of other stochastic variations. Only minimal temperature trends have been previously found for phosphorus removal for emergent marshes (Kadlec and Reddy, 2001). Here the database has been expanded considerably, and split regionally into conditions of frozen winters (cold climate) and nonfrozen winters (warm climate) (Table 10.12).

**TABLE 10.11**
**Annual Reduction of Total Phosphorus in FWS Wetlands**

**Stipulations**

1. Period of record averages are used in calculations.
2. For $k$-value calculations, the following $P$-$k$-$C^*$ parameters are selected:
   a. $C^* = 0.002$ mg/L
   b. If not measured, $P = 3.4$ TIS (mean of 79 measured values); otherwise, the measured value.
3. Ranges of variables:

| | HLR (cm/d) | TP In (mg/L) | TP Out (mg/L) |
|---|---|---|---|
| **Mean** | 12.3 | 7.9 | 5.1 |
| **Median** | 4.66 | 1.03 | 0.48 |
| **Max** | 181.1 | 126.1 | 100.8 |
| **Min** | 0.132 | 0.007 | 0.006 |

**Results (N = 282 wetlands)**

| Percentile | Load Removed ($g/m^2 \cdot yr$) | Rate Constant (m/yr) |
|---|---|---|
| 0.0 | –31 | –6.9 |
| 0.1 | 0.22 | 1.4 |
| 0.2 | 0.48 | 2.8 |
| 0.3 | 1.59 | 4.5 |
| 0.4 | 3.48 | 6.6 |
| 0.5 | 6.00 | 10.0 |
| 0.6 | 11.3 | 13.1 |
| 0.7 | 26.6 | 16.7 |
| 0.8 | 45.6 | 25 |
| 0.9 | 92 | 60 |
| 1.0 | 474 | 347 |

**TABLE 10.12**
**Temperature Coefficients for Phosphorus Removal in FWS Wetlands**

| | Data (years) | HLR (cm/d) | TP In (mg/L) | TP Out (mg/L) | TPLI ($g/m^2 \cdot yr$) | $k_{20}$ (m/yr) | T Range (°C) | θ | $R^2$ |
|---|---|---|---|---|---|---|---|---|---|
| **Cold climate** | | | | | | | | | |
| Musselwhite, Ontario | 4 | 53.60 | 0.016 | 0.009 | 3.10 | 96 | 1–18 | 0.965 | 0.076 |
| Des Plaines, Illinois EW3 | 3 | 5.55 | 0.113 | 0.040 | 2.43 | 50 | 2–26 | 1.026 | 0.133 |
| Des Plaines, Illinois EW4 | 3 | 1.31 | 0.113 | 0.021 | 0.52 | 18 | 2–26 | 0.992 | 0.004 |
| Eskilstuna, Sweden | 3 | 16.32 | 0.225 | 0.110 | 13.20 | 87 | 1–17 | 1.058 | 0.235 |
| Linköping, Sweden | 3 | 4.04 | 0.281 | 0.078 | 4.11 | 18 | 2–17 | 0.980 | 0.050 |
| Commerce Twp., Michigan | 5 | 17.35 | 0.296 | 0.260 | 18.15 | 20 | 2–22 | 1.064 | 0.645 |
| Brighton, Ontario | 5 | 5.14 | 0.454 | 0.271 | 8.43 | 3 | 2–22 | 0.880 | 0.443 |
| Listowel, Ontario(Cell 3) | 4 | 1.34 | 0.820 | 0.410 | 4.04 | 13 | 1–18 | 1.040 | 0.331 |
| Estevan, Saskatchewan | 9 | 2.62 | 2.260 | 1.720 | 21.60 | 0 | 2–18 | 0.852 | 0.106 |
| Listowel, Ontario 4 | 4 | 1.74 | 2.880 | 0.480 | 18.44 | 14 | 1–18 | 0.980 | 0.044 |
| **Median (cold climate)** | | | | | | **18** | | **0.986** | **0.120** |
| **Mean (cold climate)** | | | | | | **32** | | **0.984** | **0.207** |
| **Warm climate** | | | | | | | | | |
| Boney Marsh, Florida | 7 | 2.80 | 0.054 | 0.021 | 0.55 | 13 | 18–31 | 1.017 | 0.088 |
| ENRP, Florida | 5 | 2.38 | 0.108 | 0.022 | 0.94 | 18 | 18–31 | 1.005 | 0.004 |
| Orlando Easterly, Florida | 10 | 6.03 | 0.183 | 0.084 | 4.04 | 19 | 8–28 | 1.006 | 0.001 |
| Titusville, Florida | 7 | 0.78 | 0.288 | 0.049 | 0.82 | 6 | 17–29 | 1.080 | 0.373 |
| New Hanover, North Carolina | 2 | 1.47 | 0.315 | 0.219 | 2.76 | 3 | 8–32 | 1.002 | 0.000 |
| Byron Bay, Australia | 3 | 1.24 | 0.918 | 0.364 | 3.16 | 5 | 10–28 | 1.000 | 0.000 |
| Imperial, California | 4 | 19.32 | 1.340 | 0.830 | 94.33 | 45 | 10–31 | 1.029 | 0.069 |
| Brawley, California | 4 | 8.87 | 1.400 | 0.710 | 45.45 | 18 | 10–31 | 0.992 | 0.017 |
| Lakeland, Florida | 7 | 1.02 | 5.080 | 3.030 | 18.26 | 2 | 16–28 | 1.086 | 0.064 |
| Warangal, India | 1 | 4.28 | 7.400 | 4.000 | 63.60 | 27 | 16–34 | 1.044 | 0.557 |
| **Median (warm climate)** | | | | | | **16** | | **1.012** | **0.041** |
| **Mean (warm climate)** | | | | | | **16** | | **1.026** | **0.117** |
| **Overall Median** | | | | | | **18** | | **1.006** | **0.073** |
| **Overall Mean** | | | | | | **24** | | **1.005** | **0.162** |

### *Cold Climate Wetlands*

Cold climate systems display two peaks of uptake, in spring and again in autumn (Figure 10.19). As a consequence, the modified Arrhenius temperature dependence of the uptake rate coefficient is not a good model. This is underscored by rate constants that decrease with increasing temperature (median $\theta = 0.986$), and only a small amount of the seasonal variance explained (median $R^2 = 0.012$). Monthly values of $k$ provide a better description.

The spring peak in phosphorus uptake is readily visualized as plant uptake during the peak growing season. The secondary peak in autumn is not as intuitive, but likely has its explanation in terms of a summer maximum in release due to decomposition, coupled with belowground phenomena in autumn. The visual early autumn senescence of aboveground plant parts can be misleading, because phosphorus accumulation in shoot bases appears to continue into December (Prentki *et al*., 1978). Further, roots and rhizomes accumulate phosphorus during autumn, which provides the storage needed to partially support the spring aboveground growth spurt. Translocation of phosphorus from aboveground sources to rhizomes can occur in treatment wetlands, ranging from zero (Tanner, 2001a) to 58% of the standing phosphorus crop (Mueleman *et al*., 2002). Additionally, studies have shown that both aboveground (Puriveth, 1980; Gessner, 2000) and belowground (van der Valk and Attiwil, 1983; Wrubleski *et al*., 1997) litter may increase in phosphorus mass during the cold season. In short, microbial and plant soil processing of phosphorus continues well after the senescence of aboveground plant parts is complete.

The return flux of phosphorus from decomposition processes may also contribute to the pattern of uptake observed in Figure 10.19. The rate constant is a parameter in a lumped model, which describes a net transfer, which is the difference between uptake processes and release processes to and from the active compartments in the ecosystem. Thus, in midsummer under warmest temperatures, microbial uptake may be operating rapidly, but so are leaching and decomposition processes. High returns of phosphorus from the static compartments can depress net rates of uptake, and help create the midsummer low in the overall rate constant.

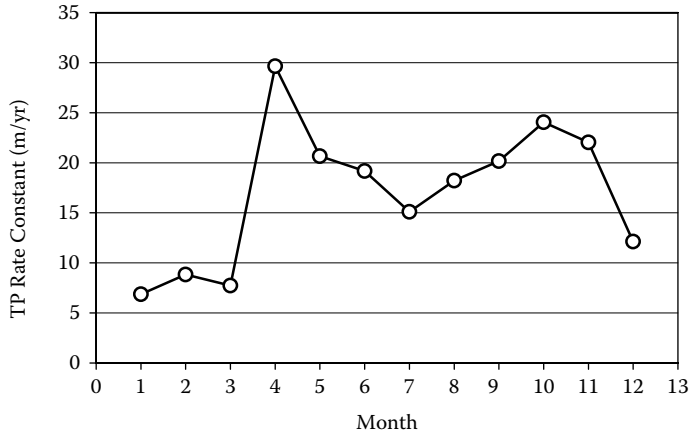



**FIGURE 10.19** Mean monthly phosphorus removal rate constants for northern FWS wetlands, by month (N = 17 wetlands). The average annual value is 18 m/yr.

### *Warm Climate Wetlands*

The examination of wetland phosphorus removal data for warm climates also shows only minimal seasonal effects (Table 10.12) according to the modified Arrhenius model. For instance, the model is calibrated by $\theta = 1.006$ for the Orlando Easterly Wetland in Florida, but accounts for none of the variance in rate constants ($R^2 = 0.001$) (Figure 10.20).

## Effects of Ecosystem Variables

### *Vegetative Cover*

It is difficult to find information on the phosphorus removal potential for unvegetated wetlands. Presumably, shallow facultative lagoons represent this end of the vegetation spectrum, but data for such systems are almost entirely absent (Water Environment Federation, 2001; Abis, 2002). In the design of lagoon systems, phosphorus removal is usually presumed to be minimal, and chemical addition is normally used when a phosphorus standard is to be met (Surampalli *et al*., 1995). Data for lakes show low removal compared to wetlands (Walker, 2003; Walker and Havens, 2003). Given the mechanisms discussed earlier, it is reasonable to presume that plants, either submerged or emergent, promote a major fraction of observed phosphorus removal in wetlands. Consequently, fully vegetated systems are to be preferred to unvegetated lakes and ponds.

Experience with SAV wetlands indicates that these are efficient phosphorus removal systems at low phosphorus concentrations (Gumbricht, 1993a; Dierberg *et al*., 2002; Knight *et al*., 2003). SAV has been shown to be more effective than emergent aquatic vegetation (EAV) for phosphorus removal at low concentrations (about 100 µg/L), in a subtropical climate, based upon model parameterization (Walker and Kadlec, 2005). However, SAV has been shown to be less effective for higher phosphorus concentrations (about 1,000 µg/L) (Toet, 2003).

### *Plant Establishment*

Initial establishment of plants in a constructed wetland requires more phosphorus than that needed to sustain the mature annual cycle. This establishment requirement occurs at rates which increase as the new plant density increases. For

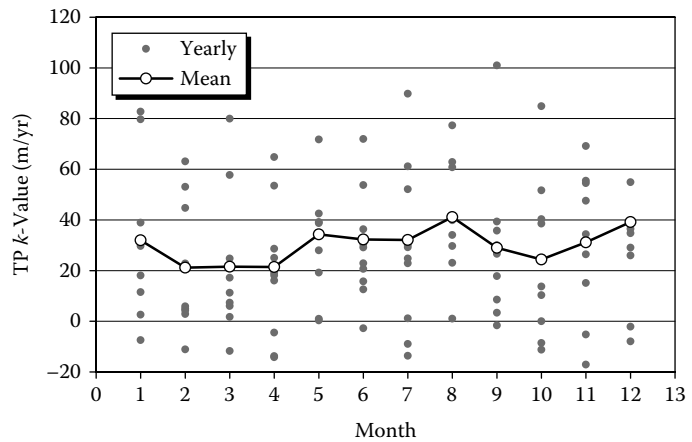



**FIGURE 10.20** Phosphorus removal rate constants for the Orlando Easterly Wetland, Florida. Ten years' monthly data for the first FWS cell were used. (Unpublished data from city of Orlando.)

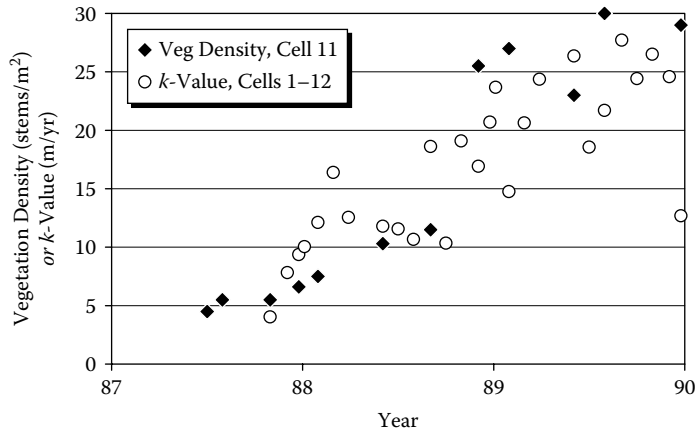



**FIGURE 10.21** Vegetation density and phosphorus rate constant at the Orlando Easterly Wetland, Florida, during start-up. (Unpublished data from city of Orlando.)

instance, Lakshman (1982) found phosphorus uptake strongly correlated with new stem density ($R^2 \approx 0.94$). A similar effect was present during the start-up of the Orlando Easterly Wetland (Figure 10.21).

### *Water Depth*

Water depth may also influence phosphorus removal by affecting the amount of submersed biota, and by affecting the hydraulics of flow. Hydraulics are more likely to be a factor in large wetlands that have considerable topographic irregularity, allowing "islands" and deeper zones, which may cause channelization or short-circuiting. In smaller wetlands with more uniform inundation, the amount of submerged "biology" is a more likely cause of short-circuiting.

Wetland data strongly support the idea of an areal rate model for small wetlands. For instance, the 17 Jackson Bottoms, Oregon, wetlands were constructed with varying proportions of deep and shallow areas, ranging from 12–96% in deep zones (SRI, 1991). These offered a side-by-side comparison of depth effects, and gave a strong denial of the first-order volumetric model for phosphorus removal. The volumetric rate constants decreased with increasing depth, as forecast from a constant areal uptake constant ($k$) according to:

$$k_V = \frac{k}{\varepsilon h} \tag{10.10}$$

where

$k_V$ = volumetric rate constant, 1/d
$k$ = areal rate constant, m/d
$h$ = water depth, m
$\varepsilon$ = hydraulic efficiency, dimensionless

The Jackson Bottoms data fit Equation 10.10, for areal rate constant ($k$) with $R^2 = 0.94$. Listowel, Ontario, Canada, systems 4 and 5 received the same water over a four-year period, and fit Equation 10.10 with $R^2 = 0.90$ (Figure 10.22).

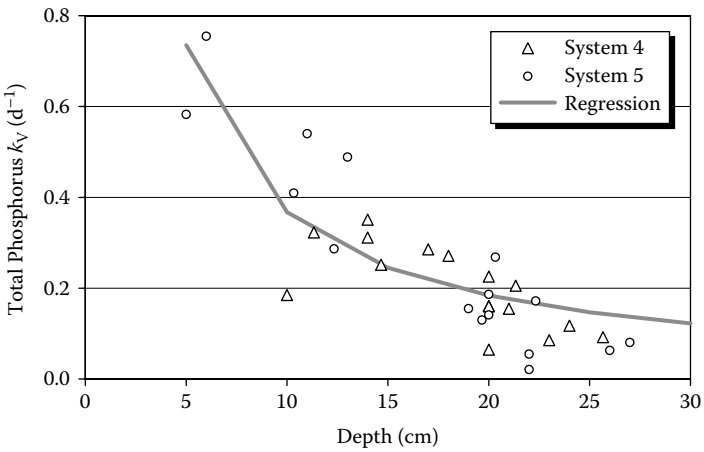



**FIGURE 10.22** Dependence of the volumetric phosphorus rate constant on depth for Listowel, Ontario, FWS Systems 4 and 5. The values selected were $P$ = 3 TIS and $C^*$ = 0.02 mg/L. Data span four years at quarterly frequency. The line is an inverse proportion between $k_V$ and depth. (Data from Herskowitz (1986) *Listowel Artificial Marsh Project Report.* Ontario Ministry of the Environment, Water Resources Branch, Toronto, Ontario.)

In contrast, data taken over a four-month period in south Florida on small side-by-side wetlands displayed ambiguous depth effects for $k_V$ and $k$. Although deeper water was less effective for cattails (SFWMD, 2004) and SAV (SFWMD, 2002) at higher concentrations (>50 µg/L), there was not such evidence at lower TP concentrations.

Accordingly, it may be beneficial to retain options for both areal rate constant ($k$) and volumetric rate constant ($k_V$) contributions, according to:

$$J_{net} = (k + hk_V)(C - C^*) \qquad (10.11)$$

### *Speciation*

TP is a lumped constituent in water entering a treatment wetland, comprised of several types of compounds. TP is comprised of PP, DOP, and SRP. Each of these analytical categories is in turn a mixture. PP is associated with a spectrum of particle sizes and types. DOP is associated with a suite of organic molecules of varying types and sizes. Even SRP contains components, such as mono- and dibasic phosphates, and polyphosphates. Each component of the overall TP mixture will have its own removal rate, and presumably these will be different. As water containing such a mixture passes through the wetland, not only is TP reduced, but its species composition changes because different fractions of the mixture are reduced at different rates. The mixture becomes *weathered*, a term originally coined to describe the selective stripping of light volatile materials upon exposure to outdoor environments. To date, only minimal attention has been focused on phosphorus speciation.

## 10.6 FWS WETLAND DETAILED PHOSPHORUS MODELING

### Aquatic Systems in General

Phosphorus modeling in aquatic systems has followed three relatively independent literature threads: lakes, rivers, and wetlands. Lake phosphorus models often utilize a first-order areal removal model, containing a "settling rate" as the calibration parameter (Vollenweider, 1975). Conceptually, phosphorus is removed by phytoplankton, which subsequently dies and settles. No stationary ecosystem compartments are considered, rather, removal is from a lumped solids-and-water phase. These simple models are conveniently applied in a single well-mixed unit hydraulic framework, for the lake, over an averaging period of months or years (Canfield and Bachman, 1981). Applications continue up to the present, with modifications to include dynamics (Walker and Havens, 2003). However, some situations have called for expanded models that include stationary compartments, such as detritus, as well as processes involving zooplankton and fish (Sagehashi *et al*., 2000, 2001).

Phosphorus assimilation by streams has also been modeled using a first-order model (Newbold *et al*., 1983). However, for these linear flowing water bodies, the underlying hydraulic model has presumptively often been taken to be plug flow, with or without dispersion. Further, the removal is taken to be volume-specific rather than area-specific, and hence the removal coefficient is expressed in units of inverse time. Still further, this coefficient is typically normalized to the water velocity, resulting in a reported parameter of phosphorus assimilation length (Reddy *et al*., 1999). However, more satisfying results were obtained by Simmons and Cheng (1985) when a two-path removal mechanism was used, corresponding to algal uptake and particulate settling. When the river is vegetated with macrophytes, there are accompanying epiphytes, and the phosphorus removal pathways increase in number (Wade *et al*., 2001).

Wetlands are perceived to differ from aquatic systems (lakes and streams) in terms of depth, velocity, and vegetation density. The standing stocks of emergent or submergent macrophytes, or of various kinds of algal communities, are typically larger. Therefore, early wetland models involved stationary storage compartments, including vegetation and soils (Dixon, 1974; Parker, 1974). Macrophyte biomass is observed to consist of live and dead aboveground and belowground parts. Algae are sometimes conceived to consist of phytoplankton (suspended), epipelon (bottom sediments), epiphyton (on submersed plant parts), and metaphyton (mats) (McDougal *et al*., 1997). However, the nonplanktonic forms are often termed periphyton. It is also generally agreed that the wetland sediments and soils can be categorized as consisting of an "active" component and an "inactive" component (Kadlec and Hammer, 1988; Mitsch and Gosselink, 2000a).

An example of a wetland compartment model is shown in Figure 10.23, modified from Wang and Mitsch (2000).

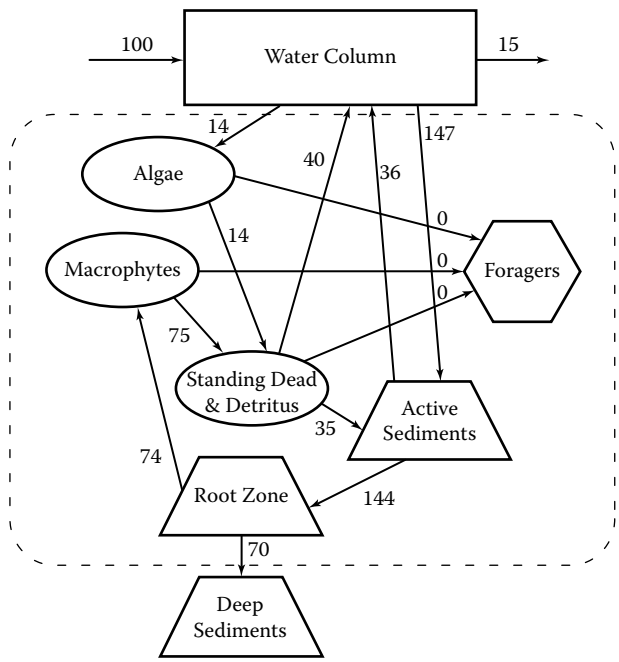



**FIGURE 10.23** Relative phosphorus movement for an average four Des Plaines River wetlands. The mean phosphorus loading was 1.2 g/m²·yr. The period of observation was two years, during which the systems were "not quite in steady state." The aggregation of storages into a single model compartment is indicated. (Adapted from Wang and Mitsch (2000) *Ecological Modelling* 126: 101–130. Reprinted with permission.)

Their dynamic one-tank model was calibrated to information drawn from four wetlands at the Des Plaines River wetland demonstration site, from operational years 1989–1991. During that time period, these wetlands were tracer-tested and found to behave as approximately four TIS (Kadlec, 1994a), and subsequently modeled in that hydraulic framework according to a first-order areal removal (Kadlec, 2001a).

### Phosphorus Models for Everglades Wetlands

#### Everglades Landscape Model (ELM)

The Everglades landscape model is a work-in-progress that targets predictions of water and phosphorus movement across the entire south Florida landscape (Fitz *et al*., 1996; Fitz and Sklar, 1999; SFWMD, 2003b). The ELM uses five stationary compartments: periphyton, floc, macrophytes, macrophyte detritus, and soil. It allows 12 community designations, and tracks carbon as well as phosphorus. It utilizes 2-D spatial discretization, but moves water and chemicals only by plug flow advection. It imports its hydrology from the stand-alone South Florida Water Management Model, which uses a relatively coarse grid ($3.2 \times 3.2$ km cells). Because of the multiple phosphorus transfers and the use of habitat-specific parameters, there are hundreds of coefficients that must be guessed or measured or determined from calibration. When the complex phosphorus cycling processes are combined with large numbers of grid cells, and multiple years of simulation, long run times on high-level computing platforms become necessary. These attributes make ELM inappropriate for exploring potential design and operational alternatives.

#### Wetland Water Quality Model (WWQM)

The need for a spatial–temporal model to describe phosphorus removal in the Stormwater Treatment Areas of the Florida Everglades (STAs) led to the development of this model, which specifically targeted the STAs (HydroQual, Inc., 1997, 1998; Moustafa and Hamrick, 2002). The WWQM uses two spatial dimensions, and allows for dynamic behavior. Hydrodynamics are calculated first, followed by chemical interactions in four stationary compartments: periphyton, macrophytes, aerobic sediments, and anaerobic sediments. It attempts to include calcium, iron, carbon, nitrogen, and sulfur in the intrinsic chemistry. As a consequence, the code solves over 100 equations requiring over 200 coefficients. The attendant computational times were found to be quite large. To date, this model has not been successfully implemented.

#### Everglades Phosphorus and Hydrology (EPH) Model

This model is spatially discretized into plug-flow cells in series, selected by the user to reflect landscape and water quality regions. It is simplified by consideration of only a few transfers, and by the selection of a monthly time step. Phosphorus removal is calculated using a first-order areal uptake equation, and return is taken to be zero order and operative only during dry periods (Munson *et al*., 2002). The EPH was used to theoretically investigate the results of different operational and design strategies for the STAs.

#### The Dynamic Model for Stormwater Treatment Areas (DMSTA)

A simple extension of the *P-k-C** design model is the addition of *P* storage in the biota of the wetland ecosystem. This extension has been applied to treatment wetlands in Michigan (Kadlec, 1997a), but that application did not include pulsed inflows. Treatment wetlands are now in use for controlling phosphorus in pulse-driven sequences of runoff that often occur in both agricultural and urban basins. During periods of high phosphorus availability, that storage will increase, and in periods of phosphorus famine, the storage will decrease. At all times, the ecosystem produces the residual sediments containing unavailable phosphorus, which characterized the WCA2A calibration data of the STA design model (Walker, 1995). This extension has been named the Dynamic STA Design Model (DMSTA) (Walker and Kadlec, 2005). DMSTA is an unsteady state model that removes phosphorus to permanent burial in proportion to the amount of labile phosphorus in storage. There is an implied labile pool of sedimentary and biomass phosphorus, in addition to the permanently buried phosphorus. That labile pool is presumed to be drawn down by a return flux, or bleed-back. Temporal variations in the water budget (flows, rainfall, ET, seepage, storage) are also simulated. DMSTA allows for spatial variability via cells in series or parallel, and accommodates *P*TIS for each wetland cell. DMSTA is part of a broader approach to phosphorus management modeling (Kadlec and Walker, 1999).

## 10.7 INTRASYSTEM PHOSPHORUS VARIABILITY IN FWS WETLANDS

### Stochastic Behavior

The framework for modeling treatment wetland performance, in general, is comprised of a deterministic component and a stochastic component characterized, respectively, by appropriate equations and one or more probability distributions (Kadlec, 1997b). Treatment wetland performance is typified by large coefficients of variation, and therefore both pieces of the description are necessary:

$$C_o = C^* + \frac{(C_i - C^*)}{\left(1 + \frac{k}{Pq}\right)^P} + E \qquad (10.12)$$

where

$E$ = stochastic component of outlet concentration, mg/L

Even if the *P-k-C** model is not used, there remains variation around the annual trendline. For example, there is often a sinusoidal variation through the year ($C_{trend}$), with an additive

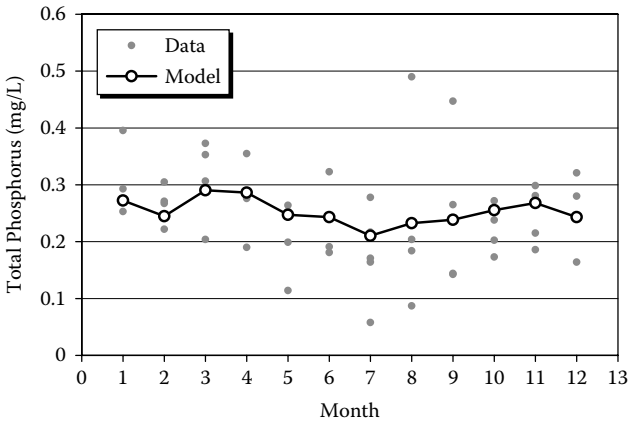



**FIGURE 10.24** Variability of outlet TP concentrations compared to a deterministic first-order model for the Commerce Township, Michigan, FWS wetland. (Unpublished data from Commerce Township, Michigan.)

random portion ($E$). The corresponding effluent concentration can be represented as:

$$C_o = C_{avg}\left[1 + A \cdot \cos\left(\omega\ (t - t_{max})\right)\right] + E = C_{trend} + E \quad (10.13)$$

where

$C_{avg}$ = mean concentration, mg/L
$A$ = amplitude of the concentration cycle
$\omega$ = annual cycle frequency = $2\pi/365 = 0.0172\ d^{-1}$
$t$ = time, Julian day
$t_{max}$ = time of maximum concentration, Julian day
$E$ = stochastic component of outlet concentration, mg/L

The random variation, $E$, can be captured through the use of a multiplier on the trend value, $C_{trend}$:

$$E = C_{trend}\Psi \quad (10.14)$$

where

$\Psi$ = stochastic portion of $C_o/C_{trend}$, dimensionless

Data from the Commerce Township wetland in Michigan provides an idea of the variability to be expected with respect to the mean long-term performance for phosphorus removal. This wetland received an average inlet concentration of 0.296 mg P/L for 1999–2003, at an average HLR of 17.1 cm/day (nominal detention time ≈ three days), and produced a mean outlet concentration of 0.260 mg/L. Figure 10.24 shows a considerable probability band around the mean cyclic trend performance for the monthly means of daily TP measurements. The maximum monthly value was 0.49 mg P/L at the outlet, about 90% higher than the annual average outlet concentration.

The corresponding frequency diagram for the Commerce Township Wetland is shown in Figure 10.25. The reader is cautioned that some literature promotes the use of the percentile points of a distribution of errors around the long-term mean value (see, e.g., Kadlec and Knight, 1996). The excursions in this latter approach, therefore, reflect both seasonal departures and the random variability. Figure 10.25, and the statistics drawn from the underlying analysis, uses the seasonal trend line as the basis, and therefore shows a slightly lesser variability.

## Anticipated Excursion Frequencies

Probabilistic effects are important in the utilization of design models for predicting phosphorus removal performance of treatment wetlands. Regulatory requirements vary from location to location, but often there is a standard other than a long-term average. There may be a maximum monthly concentration or a specified concentration not to be exceeded for more than a certain percentage of samples. To meet this type of regulatory requirement, the designer needs information on the variability of wetland outlet concentrations. Models

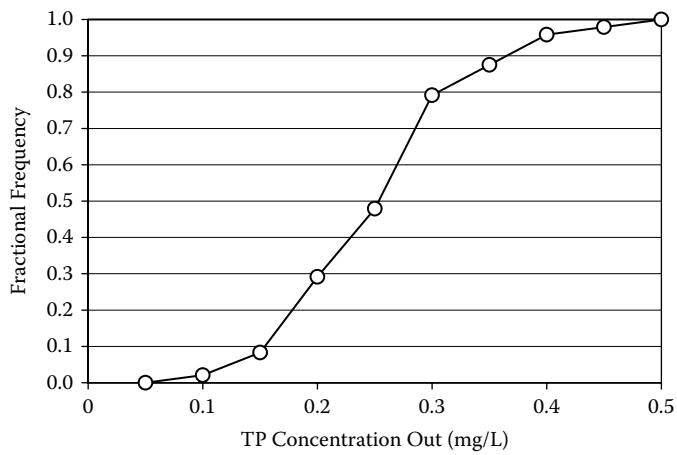



**FIGURE 10.25** Frequency distribution of the effluent total phosphorus concentration for the Commerce Township, Michigan, FWS treatment wetland. (Unpublished data from Commerce Township, Michigan.)

represent only the seasonal trends of effluent TP concentrations, leaving a considerable amount of probabilistic scatter in performance (see Equation 10.13). Monthly limits are the most common averaging period. Therefore, it is useful to examine the variability of monthly average outlet concentrations, here represented by a fractional addition to the trend value ($\Psi$), or the multiplier on the trend value ($1 + \Psi$).

Because designers are interested in preventing or controlling exceedances, the upper percentile points of that distribution are useful in design. For instance, the 90th percentile represents the fractional addition to the trend that may be expected to occur one time out of ten during the period of record. Note that these $\Psi$-values are not quite compliance percentiles, because of the finite number of months involved in a calibration set.

Table 10.13 lists percentiles of the monthly $\Psi$-distributions for phosphorus for some representative FWS wetlands. These data indicate that the median of the 95th percentile is an additional 128% above the trend, and that the median of the 90th percentile is an additional 94%. To incorporate this variability into design, the wetland designer must oversize the wetland. Suppose, for example, it is necessary to meet a monthly limit ($C_o$) of 0.3 mg/L of TP at least nine times out of ten (90th percentile). The designer should proceed in three steps:

1. Use seasonal variations (either temperature-driven or month-driven) to establish the bottleneck period, which is likely to be winter unless there are large reductions of inflow rates and concentrations in winter. The month with the highest expected outlet trend concentration is selected as the controlling month.
2. Account for the random part of the effluent concentration distribution during the controlling month. For instance, this could be accomplished by choosing $\Psi$ to be the median value of 94% (0.94) from Table 10.13.
3. Increase the wetland size so that the calculated seasonally variable effluent concentration ($C_{trend}$) does not exceed $0.3/(1 + 0.94) = 0.154$ mg/L during the bottleneck period.

*The reader should note that this correction is not a safety factor.* It is known from the nature of existing data sets that excursions will occur. It becomes necessary to plan for those excursions, so that potential exceedances occur at less than the predetermined frequency. Wetland data may sometimes contain sufficient intrasystem variability to place the 100th percentile above the median inlet concentration. Thus, it may

**TABLE 10.13**
**Trend Multipliers for Effluent Total Phosphorus Concentrations in FWS Wetlands**

| | | Excursion Frequency | | | |
|---|---|---|---|---|---|
| **Site** | **Years** | **50%** | **80%** | **90%** | **95%** |
| Lakeland 1 | 7 | 1.19 | 1.46 | 1.75 | 1.80 |
| Linköping | 3 | 1.08 | 1.39 | 1.97 | 2.14 |
| New Hanover | 2 | 1.00 | 1.61 | 1.71 | 2.04 |
| Musselwhite | 4 | 0.97 | 1.42 | 1.78 | 2.16 |
| Imperial | 4 | 1.44 | 2.86 | 3.80 | 4.94 |
| Brawley | 4 | 1.02 | 1.22 | 1.37 | 1.45 |
| Ekeby | 3 | 0.89 | 1.46 | 1.90 | 2.39 |
| OEW | 10 | 1.04 | 2.62 | 3.28 | 4.30 |
| Commerce Twp | 5 | 0.98 | 1.13 | 1.22 | 1.39 |
| Estevan | 6 | 0.81 | 1.27 | 3.56 | 6.10 |
| Boney Marsh | 9 | 0.88 | 1.21 | 1.52 | 1.90 |
| Listowel 3 | 4 | 0.99 | 1.86 | 2.75 | 3.00 |
| Listowel 4 | 4 | 0.96 | 1.85 | 2.63 | 3.19 |
| Listowel 5 | 4 | 0.98 | 1.71 | 2.94 | 3.40 |
| Brighton | 5 | 0.93 | 1.67 | 2.22 | 2.85 |
| **Median** | | **0.99** | **1.54** | **1.93** | **2.28** |
| **Mean** | | **1.00** | **1.65** | **2.29** | **2.87** |

*Note:* Data are analyzed on a monthly basis. Trend multiplier is $(1 + \Psi)$; see Equation 6.61. For instance, one month out of ten, we can expect a TP concentration 1.93 times the trend value computed from the *P-k-C** model, based on the median of the 15 wetlands.

not be possible to design a wetland to totally avoid the possibility of monthly exceedances.

## 10.8 LONGEVITY OF PHOSPHORUS REMOVAL IN FWS WETLANDS

There has been confusion about the life expectancy of FWS treatment wetlands for phosphorus removal. As discussed in this chapter, soil sorption and biomass expansion are processes that have finite phosphorus removal capacities. However, soil accretion is a sustainable mechanism that requires only a long hydroperiod to prevent oxidative release. Two lines of information allow confirmation of the long-term sustainability of phosphorus removal in FWS wetlands: the current performance of systems that have been receiving wastewater with phosphorus for extended (unmonitored) periods, and the life histories of wetlands that have had continuous monitoring.

### Long-Term Track Records

Some of the earliest treatment wetland projects have continued under continuous monitoring for up to 30 years. Two are reviewed here, to illustrate the long-term behavior of FWS wetlands for phosphorus removal.

### Houghton Lake, Michigan

This is a natural wetland project, but is included because of its long period of operation. The wetland was originally primarily a sedge meadow of modest hydroperiod (approximately 50%). Mesocosm studies (1973–1974) and a 360 $m^3$/d pilot project (1975–1977) indicated positive results for phosphorus removal. A 9,000 $m^3$/d full-scale project started in 1978, and continues to present. Because the impact zone is less than the full extent of the wetland, analysis focuses on a 100-ha subsystem around the discharge, which is defined by established sampling locations, and fully containing the visually impacted zone of the wetland (80+ ha). The incoming lagoon water averaged 3.21 mg/L, at a hydraulic loading of 0.2 cm/d for the first 28 years of operation. Removal was 2.08 g/$m^2$·yr, or 92% of the incoming mass, with no allowance for background concentration. The time sequence of events (Figure 10.26) shows a slight decreasing trend, to 88% mass reduction at year 30.

More interesting is the appearance of increased variability in the percent removal during the last ten years of operation. This scatter can be traced to variations in system hydraulics and hydrology, more so than to variations in inlet phosphorus concentrations. The outgoing load, and hence the mass removal, reflects the water loss to *ET*, which has been high and variable in recent years at the site.

Also of interest for this site are the forecasts that were made for treatment versus the actual events that took place at this site. For instance, Richardson and Marshall (1986) forecast that this system would become "saturated" after a cumulative loading of 1.0–1.5 g P/$m^2$. At present, after 30 years of operation, it is not "saturated" after a cumulative loading of 63 g P/$m^2$. It is instructive to examine the reasons for such an inaccurate forecast. Richardson and Marshall (1986) implicitly made some assumptions that have proven to be incorrect. First, they evaluated processes for the initial sedge community, which was entirely replaced by a much more productive *Typha* community. Second, they estimated the accretion of organic phosphorus in new sediments from short-term $^{32}P$ studies, rather than the long-term accretions that again reflect changed community types. Third, they assumed that sorption on the initial soils would in large measure govern the long-term uptake capacity of the ecosystem, but in fact accretion was dominant (Kadlec, 1997a). Fourth, they did not realize that wetland hydraulics would be greatly altered. The hydroperiod went to 100% for much of the impact zone, and a floating mat developed. Flow is now underneath the *Typha* mat in the majority of the treatment zone. The lesson from these incorrect projections is that phosphorus cycling and

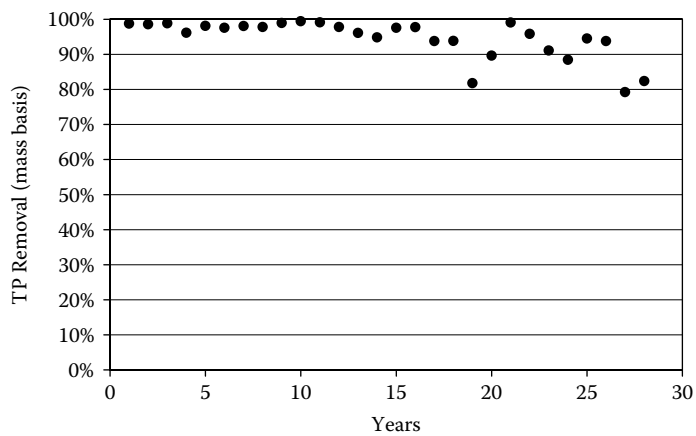



**FIGURE 10.26** The time series of percent mass removals of phosphorus for the Houghton Lake, Michigan, FWS treatment wetland. (From unpublished data.)

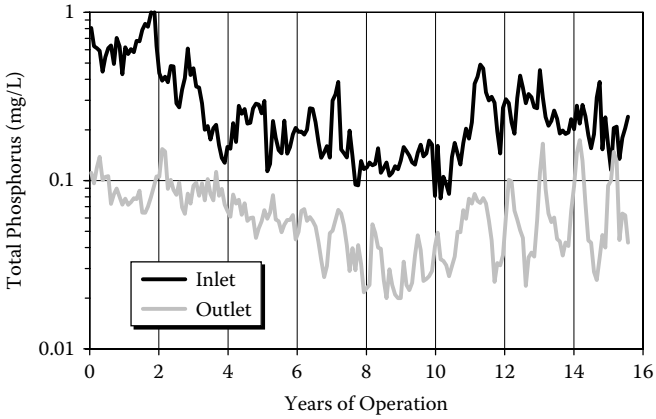



**FIGURE 10.27** Time series of inlet and outlet TP concentrations for the Orlando, Florida, Easterly Wetland. Flows to this FWS system were nearly uniform over the period of record. (Unpublished data from city of Orlando.)

ecosystem development are more complex than expected, and change over time periods of many years. It is not yet within the grasp of wetland science to synthesize short-term process measurements into long-time forecasts. The interpretation of long records is a better way to infer what may happen to phosphorus treatment wetlands over the course of time, and we now have such records.

### Orlando Easterly Wetland, Florida

The City of Orlando, Florida, employs a 490-ha FWS constructed wetland to polish tertiary wastewater. The wetland began operation in 1987, and received water at about 55,000 $m^3$/d, with TP concentration of 280 µg/L. The facility is comprised of 17 cells in series and parallel, 16 of which are mixed community marshes, plus one deep water cell (U.S. EPA, 1993a; Jackson and Sees, 2001; Wang *et al*., 2006a). TP concentrations were reduced to 64 µg/L over the period 1988–2003, at a HLR of 1.1 cm/d. The corresponding PLR was 1.16 g P/$m^2$·yr. Over the first ten years of its history, incoming TP concentrations dropped from about 800 to 200 µg P/L, and then fluctuated between 200 and 400 (Figure 10.27). During the entire history, there was a slight downward trend in percent removal, from 70–80%, possibly in response to a flow increase from 39,000 $m^3$/d to 69,000 $m^3$/d.

Although the annual average performance continued to be relatively stable, an amplified seasonal pattern began to emerge in year ten (Figure 10.27). This created winter monthly performances that were unacceptable to the city. Studies determined that sediment accretion in the front-end cells was causing hydraulic problems. Accordingly, sediment removal was conducted in the front-end cells during 2002–2003 (year 15 of operation) (Wang *et al*., 2006a). The front-end performance in the first year after the rehabilitation was much improved for hydraulics, and for TP removal (5 cells: $P$ = 12, $k$ = 30 m/yr, $C^*$ = 40 µg/L).

The lesson from this long-term project is that new sediment accretion may become a problem after a decade or two of operation. The accretions are not likely to be spatially uniform, with both gradient effects in the flow direction, and patchy development. The hydraulics of the system may be impaired long before the freeboard of the containment levees is jeopardized. Design features such as multiple controllable inlet points and adjustable water level controls are beneficial in the long-term control of performance.

## Historical Unplanned Projects

In past decades, it was not uncommon for wastewater treatment plants to discharge to existing marshes. Phosphorus removal was not part of pretreatment until recently, and hence considerable phosphorus loadings reached those marshes. Two such wetland discharges are reviewed here to illustrate the continuing ability of old wetlands to remove phosphorus.

### Brillion Marsh, Wisconsin

This 156-ha natural wetland was impacted by incoming wastewater beginning in 1923. The exact discharge quantities and changes over time are not known, but an intensive study was conducted over the two-year period, 1974–1975 (Spangler *et al*., 1976b; Fetter *et al*., 1977, 1978). The marsh was a uniform stand of *Typha latifolia* and *Typha angustifolia*, with a channel input dominated by treated wastewater. Outflow was also confined to a channel, and there were small tributary streams to the marsh conveying agricultural runoff. During the study, 16,300 $m^3$/d of influent carried 3.3 ± 2.2 mg/L of TP into the marsh. The HLR of 1 cm/d carried a load of 12.6 g/$m^2$·yr. The outgoing TP was 2.6 mg/L, for a concentration reduction of 22%. The load removed was about 2.8 g/$m^2$·yr. These studies indicate a continuing ability of this marsh to remove phosphorus.

### Great Meadows Marsh, Massachusetts

The Great Meadows National Wildlife Refuge in Concord, Massachusetts, consists of diked wetland areas constructed about 1930. Sewage treatment plants had discharged into the constructed wetlands, indirectly from 1930 to 1962, and directly from 1962 to 1978, with 1978 being the year of a hydrologic and water quality assessment (Yonika *et al*., 1979). Vegetation was *Phalaris arundinaceae* and *Typha latifolia* in shallower areas, and *Nuphar variegatum* and *Nelumbo lutea* in deeper zones. The 22-ha system received 2,090 $m^3$/d of secondary effluent, for a HLR of about 1 cm/d. The inlet TP was 2.00 ± 0.88 mg/L, producing a phosphorus loading of 6.9 g/$m^2$·yr. The outlet TP was 0.51 ± 0.40 mg/L, for a 75% concentration reduction. Phosphorus removal was seasonal, with an early spring maximum rate of 12 g/$m^2$·yr, and a midwinter release of 5 g/$m^2$·yr, and an average annual removal of 3.7 g/$m^2$·yr. This treatment wetland was functional for phosphorus removal even after several decades of operation.

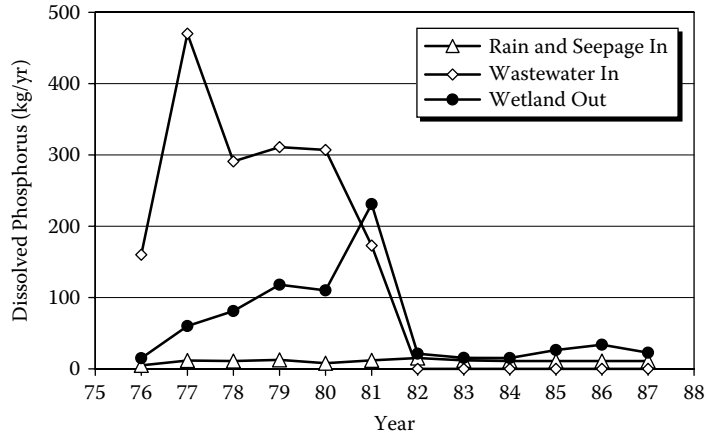



**FIGURE 10.28** Phosphorus load history of the Bellaire, Michigan, FWS treatment wetland #1. From 1976 to 1980, the source water was a facultative lagoon. In 1981, alum addition was used episodically in the lagoon. Discharges were diverted to another site in 1982. (Data from Kadlec (1983) *Wetlands*, 3: 44–63; and unpublished data.)

### Reasons for Low or Negative Reduction

There have been isolated instances of treatment wetlands that export more phosphorus than that which is contained in the influent. The discharge of antecedent loads of phosphorus from soils used in the wetland has already been discussed as one such potential cause. It has also been shown that wetland storages are partially mobile, and changing conditions in the incoming water may trigger net export, after start-up phenomena have disappeared.

An example of this "change over" phenomenon was seen at the Bellaire, Michigan, treatment wetland (Kadlec, 1983). This wetland system had an approximate area of 18 ha (Figure 10.28). From 1976 through 1980, the source water was effluent from a facultative lagoon. During the first five years, the wetland was on a (normal) track of losing its initial, saturable storage capacities, but removing considerable phosphorus (93% concentration reduction, 64% mass reduction). Because the outlet concentration was still trending upward, and was unacceptably high in 1980, alum addition was begun in the lagoon in 1981. This cut the phosphorus discharge to the wetland in half, but the "flywheel" effect of labile storage caused continued export of stored phosphorus. The amount of net export for that one year was 25% of the average load to the wetland for one earlier year. In 1982 and after, there were no further discharges to the wetland. There were also no further significant net exports of phosphorus, because the outflow mass was approximately the same as the estimated inflow and rain phosphorus load (Figure 10.28). These results have been interpreted by some as an indication that the wetland "wore out" or "became saturated." A more apt description is that the wetland "hiccupped" a small pulse of phosphorus as inlet loads were reduced and then discontinued.

The lesson here is that wetlands do not respond instantly to reduced loads, but take a period of one or more years to relax in response to the new reduced loading.

In summary, the forecasts of "phosphorus saturation" appear to have been ill-founded. However, the importance of alteration of hydrology and hydraulics may have been underestimated. The eventual appearance of erratic removal behavior may accompany the loss of detention time, possibly because of the development of short-circuits occasioned by accretion.

## 10.9 PHOSPHORUS PROCESSING IN SUBSURFACE FLOW WETLANDS

Similar to FWS wetlands, phosphorus entering subsurface flow (HSSF or VF) wetlands can be stored in plant biomass, the wetland bed media, or in accreted sediments. These storages have finite capacities, which place an upper limit on the amount of phosphorus that can be retained within HSSF or VF wetlands. One important removal mechanism in FWS wetlands, accretion, is less sustainable in HSSF and VF wetlands due to the loss of porosity and hydraulic conductivity that accompanies solids deposition.

Due to their typical use for on-site and small community wastewater treatment, HSSF and VF wetlands are constructed at a much smaller scale than FWS wetlands, and typically operated at higher PLR. Because of this, there is no equivalent to the body of knowledge developed over the last 15 years on lightly loaded FWS wetlands, as typified by the Florida systems. Instead, research to date has focused on maximizing short-term phosphorus removals within subsurface flow wetlands.

### Sorption

The material (aggregate) used to construct the HSSF or VF bed represents a reservoir of potential adsorption sites for removal of phosphorus. The amount of phosphorus that can be sorbed is a function of the inlet phosphorus concentration, and material properties of the bed media. Both the Langmuir

(Equation 10.2) and Freundlich (Equation 10.4) isotherms have been used to describe sorption, although the Langmuir isotherm is more common in the wetland literature.

Langmuir isotherm parameters, the maximum sorbed phosphorus ($S_{max}$) and the half-saturation concentration ($1/k$), are dependent on the mineralogy of the wetland aggregate. In most instances, aggregate is chosen based on local availability, hydraulic conductivity, and cost. Phosphorus sorption may or may not be a design objective. As a result, the range of phosphorus sorption capacities reported for aggregates in HSSF wetlands varies widely in the literature.

Phosphorus sorption is heavily influenced by the amount of iron, aluminum, calcium, and organic matter present in the bed aggregate (Rustige *et al*., 2003). This knowledge was applied even to the early German soil-matrix HSSF wetlands, where iron and aluminum oxides were recommended as soil amendments to increase phosphorus retention (Kickuth, 1982; Rustige *et al*., 2003). In Norway, kiln-fired lightweight expanded clay aggregate (LECA) was originally used as an alternative to gravel (Jenssen *et al*., 1994b). Assessment of the performance of these wetlands indicated that LECA had a high phosphorus retention capacity (Jenssen *et al*., 1996). The net result of these and other studies has been the assessment of the phosphorus sorption characteristics of a variety of potential HSSF bed aggregates. Following the nomenclature of Westholm (2006), these materials can be classified according to their source of origin, including minerals, soils, marine sediments, man-made aggregates, and industrial byproducts.

As phosphorus sorption reactions are endothermic (Jin *et al*., 2005), colder water temperatures will decrease the apparent sorption capacity of the bed aggregate. In a study of lightweight expanded clay (a type of manufactured aggregate), sorption capacity decreased 64% for the 2–4-mm size fraction when the water temperature was decreased from 20°C to 5°C (Zhu *et al*., 2003).

The reader is cautioned that there is a great deal of variability in phosphorus sorption studies, making direct comparison among laboratory studies difficult. There is also poor correlation between laboratory and field results, making the extrapolation of laboratory results to field applications problematic. The following items should be considered when comparing and contrasting field and laboratory studies.

- *Mineral properties of the aggregate*. Different substrates have different mineral properties. Phosphorus sorption is strongly influenced by the account of calcium carbonates, aluminum oxides, and iron oxides present in the material. As a result, comparisons for a generic material such as "soil" are generally not helpful unless the parent mineralogy (chemical composition) of the material is known (Westholm, 2006).
- *Historical mineralogy of the aggregate*. Phosphorus sorption is most strongly influenced by the surface chemistry of the aggregate (Johansson, 1997), and as this material ages and weathers over time, the surface chemistry will change (Hedström, 2006). Thus, not only the parent mineralogy, but the storage method and age of the material, are important.
- *Size of aggregate*. Finer materials have more surface area per unit volume. More surface area results in higher phosphorus sorption (Onyullo and McFarland, 2003). This logic path, taken far enough, results in the recommendation of bed aggregates so fine that, if left unchecked, would cause hydraulic failure in a subsurface flow wetland. Thus, other factors, such as hydraulic conductivity, may place constraints on aggregate size (and phosphorus sorption) for practical applications.
- *Batch flow versus continuous flow.* There exists no standardized method for measuring phosphorus sorption parameters. The maximum sorption capacity, $S_{max}$, and the half-saturation concentration, $1/k$, are a function of the concentration of the test solution (Drizo *et al*., 2002), and maximum batch concentrations, typically 320–10,000 mg/L (Zhu *et al*., 1997; Sakadevan and Bavor, 1998) used by different investigators varies widely. Of course, information at such high concentrations is meaningless in terms of treating wastewaters at 5–10 mg/L. Also, many batch experiments are conducted on shaker tables. This high degree of mixing energy results in abrasion of aggregate particles and may expose new adsorption sites (Drizo *et al*., 2002), situations that will not occur in field-scale wetlands.
- *Time scale of experiments.* Sorption parameters are typically measured using short-term batch tests. There are limitations to this approach, which typically result in overestimation of sorption capacities. As a result, long-term column studies have been proposed as a means to produce more realistic laboratory results (Drizo *et al*., 2002).
- *Laboratory versus field conditions.* Substantial differences occur between laboratory and field conditions. Biomat fouling and nonideal flow in field-scale reactors will lead to reduced phosphorus sorption capacity (Johansson, 1997). Drizo *et al.* (2002) suggested that a 50% reduction in phosphorus sorption capacity be used during design to account for these factors. In a recent study in Norway, the maximum sorbed phosphorus in a full scale HSSF wetland was only 38% of the theoretical sorption capacity (Ádám *et al*., 2006), indicating that further reductions may be prudent.
- *Speciation of phosphorus*. Laboratory measurements are typically done using pure phosphate solutions ($KH_2PO_4$). Thus, all of the phosphorus is soluble and inorganic. For full-scale applications, phosphorus in the wastewater is present in a variety of forms, including organic and particulate forms, which will affect the rate and degree of sorption.

### Mineral Aggregates

Mineral aggregates are the most common form of HSSF bed media used worldwide. These aggregates range from sands to gravels, and the source material may be fluvial deposits or crushed rock. Phosphorus sorption is dependent on the mineralogy of the parent mineral, and varies widely. Minerals that have assessed removal capacities include dolomite (Pant *et al*., 2001; Jenssen and Krogstad, 2003; Prochaska and Zouboulis, 2006), limestone (Drizo *et al*., 1999), wollastonite (Brooks *et al*., 2000), opoka (Johansson, 1997), bauxite (Drizo *et al*., 1999), calcite (Molle *et al*., 2002), zeolites (Drizo *et al*., 1999), iron-rich sands (Stuanes and Nilsson, 1987; Zhu *et al*., 1997; Kvarnström *et al*., 2004), and shale (Drizo *et al*., 1999).

Available information on Langmuir isotherm parameters for different bed media is summarized in Table 10.14 for SSF wetlands.

### Soils

Soils generally have hydraulic conductivities too low to support subsurface flow in HSSF wetlands and there has been a steady trend away from using soil beds since the late 1980s (Brix and Schierup, 1989a; Netter and Bischofsberger, 1990; Findlater *et al*., 1990; Coombes, 1990; Bucksteeg, 1990). Consequently, most soil wetlands are now used for plant rooting media in FWS wetlands (see Table 10.5). Nevertheless, some soils have favorable phosphorus retention characteristics due to their parent mineralogy. Laterite, a soil type that is most commonly found in tropical regions, is high in aluminum and iron oxides. Marl (seekreide) is a lime-rich mud or mudstone high in calcium carbonates that is derived from marine sediments. Spodosols (podzols) are weathered soils rich in aluminum and iron oxides that form in forested environments. All three soil types have been evaluated for use in constructed wetlands (Westholm, 2006). Use of soil materials requires that necessary constraints on hydraulic conductivity, soil compaction, and clogging (U.S. EPA, 2002c) are each addressed during design and construction.

### Marine Sediments

Marine sediments that have been investigated to date include maerl (algae deposits) (Gray *et al*., 2000b), oyster shells (Seo *et al*., 2005), and shell sand (Mæhlum and Roseth, 2000). These materials have a high proportion of calcium and magnesium carbonates, making them potentially attractive as phosphorus sorption aggregates. However, due to high shipping costs, utilization of these materials is likely limited to coastal areas.

### Man-Made Aggregates

Use of LECA in Norway has led to the creation of HSSF wetlands that are designed to retain phosphorus, albeit with a finite bed life (Jenssen *et al*., 1996). An updated version of this technology uses vertical flow as part of the initial treatment stage (Jenssen *et al*., 2002). Man-made aggregates vary widely in their parent material origin, and most are derived through kiln-firing shale or clay materials. These man-made aggregates are produced for the construction industry where light-weight concrete is needed (high-rise buildings, bridge decks, etc.). As a result, the constructed wetland market demand for manufactured aggregates is not significant in determining price or availability. At the present time, only two man-made aggregates, Filtralite-P® (Norway) and Utelite® (United States), are being specifically marketed as a phosphorus retention media.

### Industrial Byproducts

Due to their low cost, industrial byproducts have the potential to be used as wetland aggregates (Sakadevan and Bavor, 1998; Xu *et al*., 2006) provided that characteristics such as media size and hydraulic conductivity are suitable. Materials that have been assessed for phosphorus retention include blast furnace slag (Sakadevan and Bavor, 1998; Cameron *et al*., 2003; Silyn-Roberts and Lewis, 2003; Korkusuz *et al*., 2004, 2005; Xu *et al*., 2006; Drizo *et al*., 2006; Hedström, 2006), crushed brick (Ayres Associates, 2000), fly ash (Mann and Bavor, 1993), crushed concrete (Molle *et al*., 2002), burnt oil shale (Drizo *et al*., 1999), iron ochre (Heal *et al*., 2004), and treated wood chips (Eberhardt *et al*., 2006).

### Practical Limitations on Phosphorus Sorption

While considerable effort has been expended on assessing phosphorus retention for different types of construction materials, there are practical limitations on the use of sorption as a phosphorus removal tool, for several reasons. First of all, phosphorus sorption is limited by the finite extent of the reservoir of adsorption sites. Once the sites are saturated, phosphorus sorption will cease. Only a small fraction of the theoretical maximum sorption capacity ($S_{max}$) can actually be utilized under field conditions, because of low incoming concentrations compared to those that are needed to fill all available sorption sites.

If phosphorus removal via sorption is a design goal, this will generally control the sizing of the wetland. For instance, combined VF/HSSF wetlands in Norway typically use 8–12 $m^3$/PE of bed aggregate for a 15-year bed life, and this is for a lightweight aggregate (Filtralite-P) specifically manufactured for its phosphorus sorption capacity (Jenssen *et al*., 2002).

Due to the limited sorption capacity of most bed aggregates, HSSF and VF wetlands designed based on phosphorus sorption are likely to only be economically feasible in areas with ready access to a low-cost supply of aggregate material with very high phosphorus sorption characteristics. In most instances, the volume of bed material will be too large to be cost effective, and other means of phosphorus removal, such as chemical addition, will be more cost effective (Arias and Brix, 2004).

As the capacity of many sorbents can be exhausted in a matter of months or a few years, there is an implied step of

**TABLE 10.14**
**Summary of Bed Media Used for Phosphorus Retention in SSF Wetlands**

| System Type | Material Type | Material | Location | $d_{10}$ (mm) | $d_{60}$ (mm) | Bulk Density (g/cm$^3$) | Porosity | Clean Bed Hydraulic Conductivity (m/d) | $S_{max}$ (mg P/L) | Half Saturation Concentration, $1/k$ (mg P/L) | Reference |
|---|---|---|---|---|---|---|---|---|---|---|---|
| HSSF | Industrial byproduct | Blast furnace slag | Australia | 9.5 | 19 | — | — | — | 420 | — | Mann and Bavor (1993) |
| HSSF | Industrial byproduct | Blast furnace slag | NSW Australia | — | — | — | — | — | 44,247 | 22.6 | Sakadevan and Bavor (1998) |
| HSSF | Industrial byproduct | Blast furnace slag (fresh) | Sweden | 0.5 | 2 | 1.245 | 0.55 | 1,990 | 1,600 | — | Hedström (2006) |
| HSSF | Industrial byproduct | Blast furnace slag (weathered) | Sweden | 0.5 | 2 | 1.245 | 0.55 | 1,990 | 680 | — | Hedström (2006) |
| HSSF | Industrial byproduct | Burnt oil shale | Scotland | — | — | | 0.4 | 88.1 | 580 | 16.7 | Drizo *et al.* (1999) |
| HSSF | Industrial byproduct | Crushed concrete | France | 1.4 | 2.3 | 2.3 | 0.45 | — | 6,390 | 40 | Molle *et al.* (2002) |
| HSSF | Industrial byproduct | Fly ash | Australia | — | 0.5 | — | — | — | 260 | — | Mann and Bavor (1993) |
| HSSF | Industrial byproduct | Fly ash | Scotland | — | — | — | 0.453 | 24.2 | 860 | 14.3 | Drizo *et al.* (1999) |
| HSSF | Industrial byproduct | Furnace slag | Zhejiang, China | 0.05 | 2 | — | — | — | 8,890 | 38.5 | Xu *et al.* (2006) |
| HSSF | Industrial byproduct | Granulated blast furnace slag | Australia | 0.75 | 4.75 | — | — | — | 160 | — | Mann and Bavor (1993) |
| HSSF | Industrial byproduct | Polkemet mine ochre | United Kingdom | — | 0.6 | 1.8 | | 32 | 26,000 | — | Heal *et al.* (2004) |
| VF | Industrial byproduct | Red clay + 25% cinder | Yunnan, China | — | — | — | — | — | 1,234 | 6.7 | Zhang *et al.* (2005) |
| HSSF | Industrial byproduct | Steel furnace slag | NSW, Australia | — | — | — | — | — | 1,430 | 2.7 | Sakadevan and Bavor (1998) |
| HSSF | Manufactured | Ground plant detritus | South Korea | 0.27 | 1.3 | 0.19 | 0.64 | 320 | 1,001 | 44 | Seo *et al.* (2005) |
| HSSF | Manufactured | LWA | Norway | — | — | — | 0.404 | 60.5 | 420 | 10 | Drizo et al. (1999) |
| HSSF | Manufactured | LWA | Norway | — | — | — | — | — | 253 | 1.4 | Kvarnström *et al.* (2004) |
| HSSF | Manufactured | LWA | Texas | 0.72 | 1.7 | 0.728 | 0.594 | 92.2 | 971 | — | Forbes *et al.* (2004b) |
| HSSF | Manufactured | LWA | Utah | 0.5 | 1.6 | — | — | — | 4,530 | 20 | Onyullo and McFarland (unpublished) |
| HSSF | Manufactured | LWA | Utah | 3 | 4 | — | — | — | 3,530 | 17 | Onyullo and McFarland (unpublished) |
| HSSF | Manufactured | LWA | Utah | 6 | 9.5 | — | — | — | 2,960 | 20 | Onyullo and McFarland (unpublished) |
| HSSF | Marine | Ground oyster shell | South Korea | 0.9 | 1.5 | 0.83 | 0.57 | 1,728 | 834 | 104 | Seo *et al.* (2005) |
| HSSF | Mineral | Almind sand | Denmark | 1.40 | 3.7 | 1.46 | 0.43 | 770 ± 180 | 104 | 1.3 | Del Bubba *et al.* (2003); Arias *et al.* (2001) |
| VF | Mineral | Apatite | France | 0.5 | 0.9 | 2.48 | 0.46 | — | 4,760 | 6.7 | Molle *et al.* (2004b) |
| HSSF | Mineral | Aunsogard sand | Denmark | 0.24 | 1.0 | 1.86 | 0.3 | 22 ± 1 | 110 | 8.9 | Del Bubba *et al.* (2003); Arias *et al.* (2001) |
| VF | Mineral | Axios river sand | Greece | 0.21 | 0.49 | 1.63 | 0.33 | — | 121 | 16.7 | Prochaska and Zouboulis (2006) |
| HSSF | Mineral | Bauxite | Scotland | — | — | — | 0.345 | 51 | 610 | 3.8 | Drizo *et al.* (1999) |
| HSSF | Mineral | Bedsted sand 1 | Denmark | 0.28 | 1.1 | 1.79 | 0.32 | 143 ± 6 | 49 | 8.3 | Del Bubba *et al.* (2003); Arias *et al.* (2001) |

*(continued)*

**TABLE 10.14 (CONTINUED)**
**Summary of Bed Media Used for Phosphorus Retention in SSF Wetlands**

| System Type | Material Type | Material | Location | $d_{10}$ (mm) | $d_{60}$ (mm) | Bulk Density (g/cm$^3$) | Porosity | Clean Bed Hydraulic Conductivity (m/d) | $S_{max}$ (mg P/L) | Half Saturation Concentration, 1/$k$ (mg P/L) | Reference |
|---|---|---|---|---|---|---|---|---|---|---|---|
| HSSF | Mineral | Bedsted sand 2 | Denmark | 0.22 | 0.7 | 1.72 | 0.36 | 74 ± 7 | 42 | 10.4 | Del Bubba *et al.* (2003); Arias *et al.* (2001) |
| HSSF | Mineral | Bedsted sand 3 | Denmark | 0.25 | 0.9 | 1.66 | 0.38 | 77 ± 4 | 39 | 6.8 | Del Bubba *et al.* (2003); Arias *et al.* (2001) |
| HSSF | Mineral | Birkesig sand | Denmark | 0.32 | 1.4 | 1.83 | 0.31 | 67 ± 10 | 51 | 13 | Del Bubba *et al.* (2003); Arias *et al.* (2001) |
| HSSF | Mineral | Building sand 2 | Zhejiang, China | 0.15 | 1 | — | — | — | 130 | 29.4 | Xu *et al.* (2006) |
| HSSF | Mineral | Building sand 3 | Zhejiang, China | 0.25 | 0.75 | — | — | — | 260 | 58.8 | Xu *et al.* (2006) |
| HSSF | Mineral | Building sand 4 | Zhejiang, China | 0.1 | 1 | — | — | — | 190 | 43.5 | Xu *et al.* (2006) |
| HSSF | Mineral | Building sand 1 | Zhejiang, China | 0.15 | 0.5 | — | — | — | 290 | 13.9 | Xu *et al.* (2006) |
| HSSF | Mineral | Calcite | France | 1 | 1.6 | 2.75 | 0.47 | — | 8,750 | 29 | Molle *et al.* (2002) |
| HSSF | Mineral | Darup sand | Denmark | 0.61 | 3.4 | 1.71 | 0.36 | 360 ± 40 | 169 | 47.6 | Del Bubba *et al.* (2003); Arias *et al.* (2001) |
| VF | Mineral | Dolomite sand | Greece | 1.2 | — | 1.59 | 0.35 | — | 168 | 2.3 | Prochaska and Zouboulis (2006) |
| HSSF | Mineral | Farum sand | Denmark | 0.20 | 0.7 | 1.70 | 0.33 | 30 ± 6 | 79 | 23.8 | Del Bubba *et al.* (2003); Arias *et al.* (2001) |
| VF | Mineral | Fonthill sand | Ontario, Canada | — | — | 1.56 | 0.36 | — | 417 | 14 | Pant *et al.* (2001) |
| HSSF | Mineral | Griffith gravel | Australia | 3 | 5 | — | — | — | 48 | — | Mann and Bavor (1993) |
| VF | Mineral | Infiltration bed sand | Sweden | — | — | — | — | — | 82 | 0.9 | Kvarnström *et al.* (2004) |
| VF | Mineral | Infiltration bed sand/gravel | Sweden | — | — | — | — | — | 37 | 1.1 | Kvarnström *et al.* (2004) |
| HSSF | Mineral | Iron rich sand | Norway | — | — | — | — | — | 239 | 0.2 | Kvarnström *et al.* (2004) |
| HSSF | Mineral | Limestone | Scotland | — | — | — | 0.33 | 20.7 | 680 | 10 | Drizo *et al.* (1999) |
| VF | Mineral | Lockport dolomite sand | Ontario, Canada | — | — | 1.51 | 0.4 | — | 303 | 6 | Pant *et al.* (2001) |
| HSSF | Mineral | Logtved sand | Denmark | 0.21 | 0.6 | 1.67 | 0.36 | 69 ± 10 | 35 | 5.1 | Del Bubba *et al.* (2003); Arias *et al.* (2001) |
| HSSF | Mineral | Masonry sand | Texas | 0.11 | 0.15 | 1.67 | 0.304 | 17.3 | 59 | — | Forbes *et al.* (2004b) |
| HSSF | Mineral | Namhae gravel A | South Korea | 3.5 | 6.1 | 1.24 | 0.52 | 4,320 | 8 | 19 | Seo *et al.* (2005) |
| HSSF | Mineral | Namhae gravel B | South Korea | 1.5 | 3 | 1.58 | 0.37 | 1,234 | 12 | 8 | Seo *et al.* (2005) |
| HSSF | Mineral | Namhae sand C | South Korea | 0.4 | 1.6 | 1.59 | 0.31 | 66 | 23 | 5 | Seo *et al.* (2005) |
| HSSF | Mineral | Nymolle sand | Denmark | 0.80 | 3.2 | 1.40 | 0.44 | 1,130 ± 64 | 266 | 10.4 | Del Bubba *et al.* (2003); Arias *et al.* (2001) |

| | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| HSSF | Mineral | Quartz sand | Denmark | 0.29 | 0.9 | 1.74 | 0.32 | 111 ± 12 | 14 | 7.1 | Del Bubba *et al.* (2003); Arias *et al.* (2001) |
| VF | Mineral | Queenston shale sand | Ontario, Canada | — | — | 1.59 | 0.42 | — | 92 | 14 | Pant *et al.* (2001) |
| HSSF | Mineral | Richmond gravel | Australia | 5 | 10 | — | — | — | 26 | — | Mann and Bavor (1993) |
| HSSF | Mineral | Shale | Scotland | — | — | — | 0.377 | 86.4 | 650 | 1.6 | Drizo *et al.* (1999) |
| HSSF | Mineral | Soro sand | Denmark | 0.23 | 0.7 | 1.66 | 0.35 | 40 ± 4 | 60 | 14.7 | Del Bubba *et al.* (2003); Arias *et al.* (2001) |
| HSSF | Mineral | Vestergard sand | Denmark | 0.45 | 1.2 | 1.70 | 0.36 | 202 ± 18 | 41 | 7 | Del Bubba *et al.* (2003); Arias *et al.* (2001) |
| HSSF | Mineral | Zeolite | Scotland | — | — | — | 0.544 | 254 | 460 | 33.3 | Drizo *et al.* (1999) |
| VF | Mineral | Zeolite | NSW, Australia | — | — | — | — | — | 2,150 | 21.6 | Sakadevan and Bavor (1998) |
| VF | Soil | Loamy soil | Yunnan, China | — | — | — | — | — | 1,091 | 93.5 | Zhang *et al.* (2005) |
| HSSF | Soil | Longyou soil 1 | Zhejiang, China | — | — | — | — | — | 970 | 4.5 | Xu *et al.* (2006) |
| HSSF | Soil | Longyou soil 2 | Zhejiang, China | — | — | — | — | — | 340 | 24.4 | Xu *et al.* (2006) |
| VF | Soil | Red clay | Yunnan, China | 0.001 | 0.005 | — | — | — | 2,057 | 1.8 | Zhang *et al.* (2005) |

replacing the sorbent. If the sorption bed is the SSF wetland, the wetland must be rebuilt. However, there is no requirement that such a phosphorus sorption bed be located in the SSF wetland; it may be a pre- or post-treatment device. This option has been favored in a number of instances, such as Denmark (Arias *et al*., 2003a), Norway (Mæhlum, 1998), and Ontario (Cameron *et al*., 2003).

### Biomass Cycling

Wetland biota undergoes a cycle of growth, death, and partial decomposition. Plants uptake phosphorus during the growth cycle and release phosphorus during decomposition stages. This is the same cycle that occurs in FWS wetlands (Equations 10.5 and 10.6); however, in SSF wetlands phosphorus loadings are typically much higher, and as a result, the importance of biomass cycling (relative to the influent phosphorus load) is decreased.

A common "rule of thumb" is to size secondary HSSF wetlands at 5 $m^2$/PE for BOD and TSS removal (EC/EWPCA Emergent Hydrophyte Treatment Systems Expert Contact Group and Water Research Centre, 1990; ATV, 1998), which at 2 g/d phosphorus per person (Wallace and Knight, 2006), results in a phosphorus loading of 146 g/$m^2$·yr, or 12.6 g/$m^2$·mo. Danish systems average 150 g/$m^2$·yr (N = 71 wetlands). However, this is for so-called secondary HSSF systems. It does not necessarily apply to the hundreds of tertiary HSSF systems in the United Kingdom, for instance (Cooper *et al*., 2006a).

### Phosphorus Uptake during the Growing Season

The impacts of biomass cycling can be assessed by examining the size of the plant biomass compartment. *Phragmites*, a commonly used plant in European HSSF wetlands, typically has a standing crop of 1–2 kg/$m^2$, with a phosphorus content of about 0.2% (Vymazal *et al*., 1999). In temperate climates, the live aboveground biomass is replaced one to two times per year. This results in a maximum phosphorus uptake rate of about 8 g/$m^2$. Assuming that the growing season is four months long, the average uptake rate is 2 g/$m^2$·mo. Even if 50% of the phosphorus is taken up during the maximum growth month, this is only 4 g/$m^2$·mo, which is considerably less than the influent phosphorus load of 12.6 g/$m^2$·month. An example for a VF wetland is given in Figure 10.4, and the monthly phosphorus budgets are shown in Figure 10.10.

Vegetation growth and decay can impose a seasonal pattern of phosphorus uptake and release in the situation of lightly loaded SSF wetlands, as indicated by the potential for uptake of 100% of the applied phosphorus loading during the growing season. This occurs in reed beds treating gray water (Langergraber, 2005), and other polishing applications. More detailed analysis of the expectation of plant uptake as a contribution to phosphorus removal in HSSF wetlands may be found in the simulation studies of Langergraber (2005).

### Accumulation of Refractory Organic Material

There is one important difference between FWS wetlands and their HSSF and VF counterparts. In FWS wetlands, plant detritus (necromass) is eventually returned to the water column. Once in the water column, the supply of water needed to decompose plant detritus is not limited. Numerous wetland studies have shown that decomposition proceeds faster in oxygenated water than in dry air (see Chapter 3).

In the case of HSSF and VF wetlands, plant detritus (necromass) is not returned to the water column; instead it is deposited on top of the wetland bed. The top-of-bed environment may be unsaturated (in the case of HSSF wetlands) or intermittently saturated (in the case of intermittent downflow bed and tidal flow VF wetland configurations). In these situations, the rate of detritus (necromass) decomposition is slower than in the water in FWS wetlands. Therefore, the standing crop of litter is likely to be larger in SSF wetlands than in FWS wetlands, especially in arid climates, where the supply of water for decomposition above the bed is limited. In hot, arid climates, plant detritus will desiccate in the desert air and, in this form, is resistant to degradation. Arid climate SSF systems will thus accumulate a large standing crop of detritus.

It may be speculated that the phosphorus is leached out of the detritus and finds its way back to the water anyway, but at slower rates than in FWS systems. Because decomposition is not complete, there may be a buildup of the refractory, undecomposable fraction of the litter on top of the bed. This is expected to be larger in arid climates, and will contain residual phosphorus. These thatch layers that may necessitate removal to permit plant growth, either by mechanical removal (Nolte and Associates, 1998b) or by burning (Hanson *et al*., 2001). When the biomass is fire harvested, the majority of the accumulated phosphorus will return to the water column, having been leached from ash fall (Hanson *et al*., 2001). After burning, both FWS and SSF wetlands experience a sharp, temporary spike in phosphorus loading due to ash deposition.

### Plant Harvest to Remove Phosphorus

Generally speaking, the amount of phosphorus that can be recovered through harvesting of emergent wetland plants (e.g., *Phragmites*) is about 2–4.9 g/$m^2$·yr (Vymazal *et al*., 1999). This is a very small amount of phosphorus relative to primary influent phosphorus loadings, which typically are 150 g P/$m^2$·yr for HSSF wetlands and 300 g P/$m^2$·yr for VF wetlands. As a result, plant harvesting will remove only a tiny fraction of a primary influent phosphorus loading and is therefore not an effective method for phosphorus removal. For instance, the Lauwersoog, Netherlands, VF wetland is harvested, but only 1.9 g P/$m^2$·yr is recovered out of a loading of 34 g P/$m^2$·yr (5.6%) (Mueleman *et al*., 2002, see Figures 10.4 and 10.10). However, the same is not true for lightly loaded SSF systems, for which removals by harvest may reach nearly 40% (Vymazal, 2004).

### Chemical Precipitation

Precipitation reactions involving phosphorus are complex and computer models are typically used to estimate equilibrium conditions. Precipitation typically involves the reaction of phosphorus with metallic cations such as Fe, Al, Ca, or Mg, forming amorphous or poorly crystalline solids. These reactions typically occur at high concentrations of either phosphate or the metalloid cations (Rhue and Harris, 1999). A variety of cations can precipitate phosphate under certain conditions. Some important mineral precipitates in the wetland environment are (Reddy and D'Angelo, 1994):

| | |
|---|---|
| Apatite | $Ca_5(Cl,F)(PO_4)_3$ |
| Hydroxylapatite | $Ca_5(OH)(PO_4)_3$ |
| Variscite | $Al(PO_4)\cdot 2H_2O$ |
| Strengite | $Fe(PO_4)\cdot 2H_2O$ |
| Vivianite | $Fe_3(PO_4)_2\cdot 8H_2O$ |
| Wavellite | $Al_3(OH)_3(PO_4)_2\cdot 5H_2O$ |

In addition to direct chemical reaction, phosphorus can coprecipitate with other minerals, such as ferric oxyhydroxide, and the carbonate minerals, such as calcite (calcium carbonate—$CaCO_3$).

Several of the aggregates previously discussed are known to induce high-effluent pH concentrations, especially manufactured aggregates (LECA) and blast furnace slags. Effluent pH above 9.0 can result in measurable precipitation of soluble forms of phosphorus, although in practice, the nature and extent of precipitation reactions are dependent on the geochemistry of the water being treated (calcium, iron, and aluminum content), as well as the pH, temperature, and redox condition.

There is also not a clear-cut distinction between soluble and solid-phase precipitation reactions. It has been proposed that for blast furnace slag, this material provides nucleation sites that promote the adsorption → precipitation → crystallization of phosphorus in calcium, iron, and hydroxyl supersaturated pools (Drizo *et al*., 2002). This mechanism has been invoked to explain regeneration of phosphorus retention capacity in blast furnace slag after drainage and oxidation (Drizo *et al*., 2002).

An option for avoiding wetland media sorption as the phosphorus reduction mechanism is the use of chemical precipitation as pretreatment. Precipitants such as ferric chloride or alum are typically used. This technique is widely used as a final stage of large-scale mechanical treatment plants, and is being studied for use in small-scale systems (Arias and Brix, 2006).

### Accretion

In tropical (and humid temperate) climates, the annual rate of biomass decay approximates the rate of plant biomass production, and the accumulations of plant detritus seen in arid climates does not occur. In these climates, phosphorus uptake during the growing season will be returned through the system through biomass decay, generally within one annual cycle. The refractory (decay-resistant) portion of the plant biomass is generally in the range of 10–15% of the overall biomass production. In the case of the *Phragmites* example discussed earlier, this would result in the accumulation of 0.8–1.2 g/m$^2$·yr of phosphorus in refractory organic material. This rate of accumulation is small relative to influent phosphorus loads.

### Particulate Settling

As discussed in Chapter 7, internal environments in HSSF and VF wetlands are conducive to particulate settling and removal through a variety of mechanisms which result in the deposition of influent TSS in the inlet region of the wetland bed. This inlet region is typically a small percentage of the overall wetland, so the deposition of particulate matter (and associated phosphorus) is spatially nonuniform.

Solids that deposit within the wetland bed contribute to a loss of porosity and hydraulic conductivity, which left unchecked, leads to hydraulic failure and flooded conditions. There is a finite pore volume within the inlet region of the bed, and this can be used to estimate the limits of phosphorus retention via particulate settling within the wetland bed.

In a study of gravel-bed HSSF wetlands treating dairy wastewater in New Zealand, Tanner *et al*. (1999b) observed up to 170 g/m$^2$ of phosphorus retention in the inlet region after five years of operation. The wetland had a 0.4-m-deep bed, so the equivalent volumetric retention was 425 g/m$^3$. Accumulation of organic matter (and associated phosphorus) resulted in a 50% reduction in bed porosity (Tanner *et al*., 1998a), causing hydraulic problems and overland flow.

In VF wetlands, TSS accumulates in the inlet loading region as well. For intermittent downflow beds, this typically results in TSS accumulation in the upper portion, or on top of the wetland bed, as further discussed in Chapter 7.

## 10.10 PHOSPHORUS MOVEMENT IN SUBSURFACE FLOW WETLANDS

Phosphorus movement in subsurface flow wetlands is influenced by a number of factors. Phosphorus associated with particulate matter will be removed through settling, filtration, and interception mechanisms as discussed in Chapter 7. These three mechanisms result in accumulation of particulate matter within the inlet zone of the wetland. Soluble phosphorus will move with the water flow. As the hydraulics of subsurface flow wetlands are nonideal, there are typically short-circuits and dead zones within the wetland bed, which will affect the speed with which soluble phosphorus moves through the wetland. Phosphorus can also sorb to the bed aggregate, provided the sorption capacity of the bed is not exhausted. Lastly, phosphorus can be taken up by wetland

plants during the growing season and released after plant senescence.

The combined effect of these factors is that HSSF and VF wetlands display high phosphorus removals, until adequate phosphorus has been retained to support plant biomass cycling and until the bed sorption capacity is exhausted. Once these finite retention capacities have been consumed, the wetland will discharge effluent phosphorus concentrations close to the influent values, less phosphorus that is removed through chemical precipitation, accretion, and particulate settling.

### Effect of Bed Sorption on Phosphorus Movement

How phosphorus moves through the wetland bed is of considerable interest during the "short term" period when the aggregate is actively sorbing phosphorus, if the wetland is designed to maximize this finite removal mechanism.

In a theoretically ideal plug flow reactor, water will uniformly flow through the wetland bed. Soluble phosphorus will be adsorbed by the bed aggregate. Initially, the aggregate will begin to sorb phosphorus. As the bed begins to sorb phosphorus, the equilibrium phosphorus concentration ($C_{eq}$) (Equations 10.2 and 10.4) increases. Eventually, the equilibrium concentration and the influent concentration are the same ($C_i = C_{eq}$); the driving force for sorption ceases and this portion of the bed will remove (sorb) no more phosphorus unless there is an increase in influent concentration.

Phosphorus sorption is not an instantaneous process, because there must be transfer from the water to the surface of the solid bed particles, plus diffusional penetration into the interior of the particle (Ruthven, 1984). Additionally, bed aggregates have a distribution of binding (sorption) sites, which is dependent on the source mineralogy of the aggregate. Some binding sites are energetically more favorable for phosphorus sorption than others. Sites that are very favorable for sorption quickly bind phosphorus. Once the most energetically favorable sites are consumed, sorption occurs on less energetically favorable (slower) sites, until the driving force for sorption ceases ($C_i = C_{eq}$). Due to this distribution in binding energies, and the water-phase and solid-phase mass transfer resistances, it typically takes on the order of days to complete the sorption process (Kadlec and Hammer, 1981; Lantske *et al*., 1998).

The time to complete the sorption can be termed the *mass transfer time* ($t_{MT}$) and can be related to a *mass transfer zone length* ($L_{MT}$) as follows:

$$L_{MT} = t_{MT} \cdot v \tag{10.15}$$

where

$L_{MT}$ = mass transfer zone length, m
$t_{MT}$ = mass transfer time, days
$v$ = actual flow velocity, m/d

The $L_{MT}$ is the portion of the bed in which sorption is actively occurring. If this zone is very short, then a sorption front develops, behind which the bed particles are saturated with phosphorus, and ahead of which the bed has no phosphorus adsorbed. Such sharp sorption fronts are commonplace in chromatographic columns and other mechanical devices, but have never been observed in treatment wetlands.

Systems with slow-sorbing bed conditions, or high-flow velocities, can have an $L_{MT}$ that is longer than the bed length. Systems with an $L_{MT}$ that is longer than the bed length will report enfluent concentrations that gradually increase as the entire bed proceeds towards phosphorus saturation.

### Idealized Model of Sorption

Sorption processes in HSSF wetlands are complicated by the fact that both time and spatial variability are unavoidably involved. There is a large literature on sorption on packed beds, with a correspondingly large array of models of the sorption–saturation processes during flow through porous media (Ruthven, 1984). To a first approximation, it may be concluded that phosphorus sorption on HSSF wetlands is characterized by mass transfer control. That means that convective diffusion from the bulk flowing water, to the surface of the bed particles, is relatively slow (Figure 10.29). Under these conditions, the minimal equilibrium model includes advection, mass transfer, and accumulation in both water and solid phases.

$$v\frac{\partial C}{\partial x} + \frac{\partial C}{\partial t} = -ka_v(C - C^*) \tag{10.16}$$

$$M_s\frac{\partial C_s}{\partial t} = ka_v(C - C^*) \tag{10.17}$$

$$C_s = KC^* \tag{10.18}$$

where

$ka_v$ = mass transfer coefficient, $d^{-1}$
$C$ = water phase P concentration, $g/m^3$
$C^*$ = equilibrium P concentration, $g/m^3$
$C_s$ = soild phase P concentration, g/kg
$K$ = distribution coefficient, $m^3/kg$
$M_s$ = bulk density of aggregate, $kg/m^3$
$t$ = time, d
$v$ = actual water velocity, m/d
$x$ = longitudinal distance, m

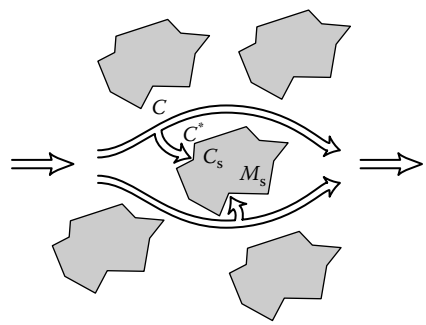



**FIGURE 10.29** Mass transfer and sorption on HSSF bed media.

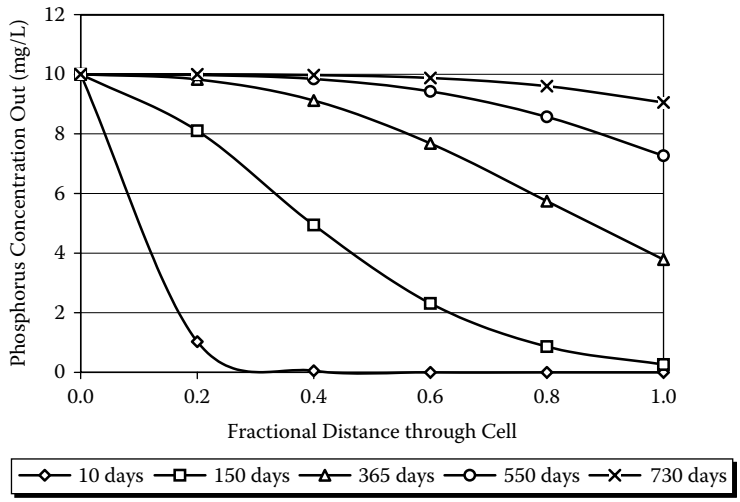



**FIGURE 10.30** Progression of phosphorus profiles in a hypothetical HSSF wetland during the exhaustion of phosphorus sorption capacity. At two years, 60% of the incoming phosphorus has been retained. The model parameters were: $K$ = 100 L/kg; $\tau_n$ = 2 d; $k_{aV}$ = 2 $d^{-1}$; 40% porosity; solid density = 1,500 kg/$m^3$; $C_i$ = 10 mg/L.

In practice, it is more realistic to solve these equations for a number of distance increments corresponding to the number of TIS that represents the HSSF wetland (see Chapter 6; Ruthven, 1984).

As an example of the general character of results to be expected, the TIS version of Equations 10.16 and 10.17 is solved for conditions approximating those observed in a typical HSSF wetland. Model parameters are given on Figure 10.30. Starting from a condition of no phosphorus sorbed, the initial water concentration profile drops rapidly in the front end of the bed (Figure 10.30). As time progresses, the bed loads with sorbed phosphorus, preferentially in the front end, until at about 150 days, breakthrough begins to occur (Figure 10.31). Until that time, 100% of the incoming phosphorus was sorbed in the bed. However, the total is only 33% of that which can ultimately be sorbed. Breakthrough ends at about 1,095 days (three years). The amount of phosphorus captured is 41% of the amount fed to the bed over the three-year period.

These model results may be compared to measured values from HSSF treatment wetlands. Longitudinal profiles are typically decreasing curves that do not go to zero phosphorus at the bed exit, but rather to some major fraction of the inlet concentration (Figure 10.32). This is the case for numerous profiles measured at Minoa, New York; Grand Lake, Minnesota; NERCC, Minnesota; Baxter, Tennessee; and Benton, Kentucky. It is also the result observed for small mesocosms packed with candidate HSSF bed material (Figure 10.33). However, anomalous profiles, with more phosphorus near the wetland exit, have been reported (McCarey *et al*., 2004). Although there are but few life history time series of HSSF phosphorus outlet concentrations, these also display breakthrough curves as expected from the model example (Figure 10.34; see also Rustige *et al*., 2003; Alas *et al*., 2006).

Of course, operating HSSF systems have other processes that complicate the analysis of phosphorus removal well beyond idealized sorption calculations. Plant uptake and

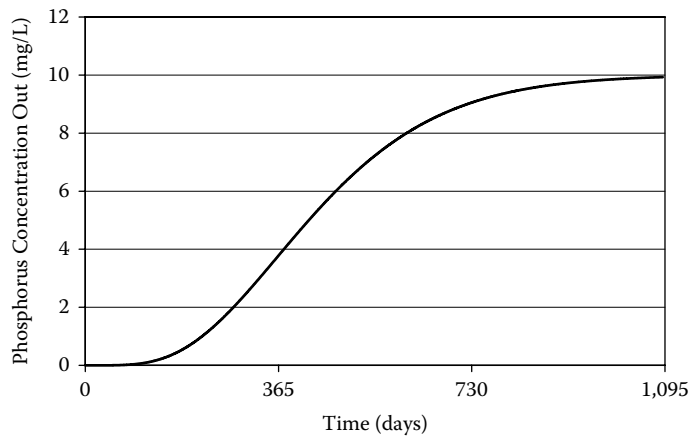



**FIGURE 10.31** Breakthrough curve for the hypothetical HSSF wetland of Figure 10.30. This model result is comparable to the data in Figure 10.34.

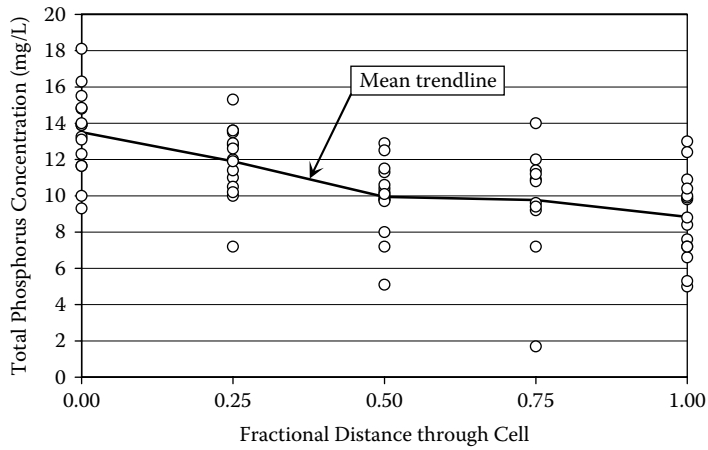



**FIGURE 10.32** Internal profile of total phosphorus for the NERCC wetland near Duluth, Minnesota. Data represents 15 transects taken down the length of the HSSF wetland bed between February and October, 1998. (From unpublished data.)

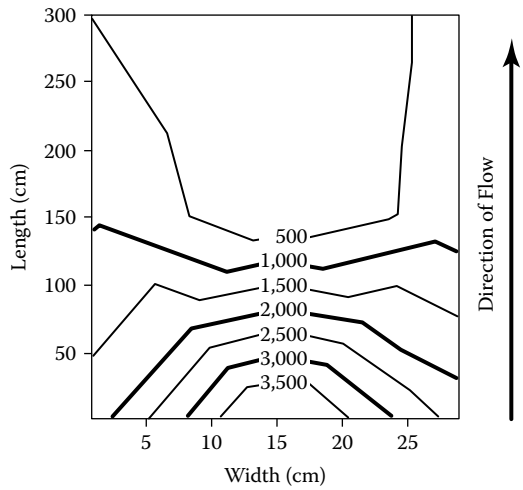



**FIGURE 10.33** Example of nonuniform spatial distribution of phosphorus sorption in a mesoscale HSSF wetland. Contour lines represent mg/kg of sorbed phosphorus. (From Ádám *et al*. (2006) *Water Research*, 40: 1143–1154. Reprinted with permission.)

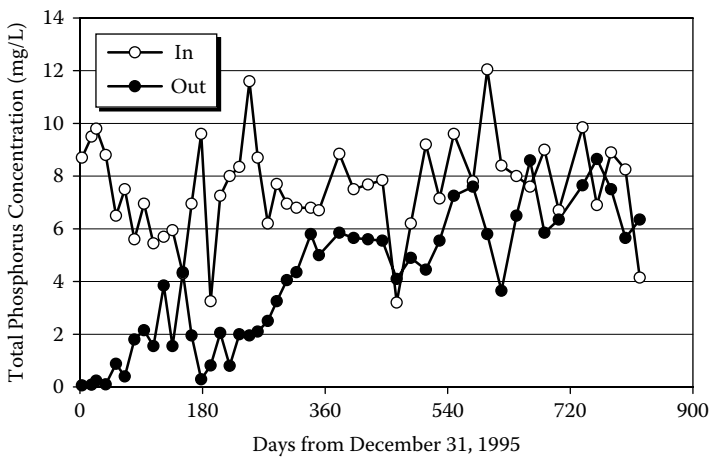



**FIGURE 10.34** Phosphorus concentrations entering and leaving the Grand Lake, Minnesota, HSSF wetland over a two-year period.

cycling may be factors in lightly loaded systems. The development of biological solids within the bed leads to a companion storage to sorption, which may contribute to phosphorus removal. Mineralization of sorbed phosphorus may free new sorption sites. Very importantly, some forms of phosphorus, such as DOP, do not sorb as readily as phosphates. Further, particulate forms of phosphorus are likely to be physically removed, and may or may not subsequently decompose and contribute to the soluble phosphorus fractions.

The sorption isotherms for materials that have been exposed to phosphorus loadings in SSF beds change characteristics as the bed becomes loaded. According to Equation 10.2, the slope of the isotherm decreases, and the equilibrium phosphorus concentration in the water ($EPC_o$) increases. This is indeed the field observation described by Pant *et al.* (2001) for three different media used in the Niagara-on-the-Lake, Ontario, VF wetlands. For example, for Fonthill sand, the initial $EPC_o = 0.02$ mg/L, while after 33 months of use, $EPC_o = 2.89$ mg/L. The slope of the isotherm went from an initial value of 85 L/kg down to 24.3 L/kg after 33 months.

## 10.11 PHOSPHORUS REMOVAL IN SUBSURFACE FLOW WETLANDS

The performance of a treatment wetland depends on both deterministic and stochastic tendencies (Kadlec, 1997b). Deterministic trends can be described as a response to influent load or influent concentration, and may or may not have a seasonal component. Stochastic variability can be described in terms of a multiplier on central trends. Both stochastic variability and deterministic trends are important elements for describing system performance in terms of permit compliance. Phosphorus removal is also limited by the presence of nonzero background concentrations.

### Background Concentrations of Total Phosphorus

Wetland systems report nonzero background concentrations for all constituents, including phosphorus. For natural basins that have no anthropogenic inputs (other that atmospheric deposition), background concentrations for phosphorus are very low, about 0.02 mg/L (Clark *et al.*, 2000). Subsurface flow wetlands, in contrast, typically experience influent phosphorus loadings that are much higher than natural wetlands. In highly loaded systems, nonzero background concentrations can occur for several reasons, including:

1. *Biomass cycling.* Biomass cycling results in the return of phosphorus to the water column through decomposition of plant detritus.
2. *Nonideal flow.* Nonideal flow and short-circuiting can deliver some fraction of the influent flow (untreated wastewater) to the outlet zone.
3. *Sampling errors.* Sampling errors that produce elevated concentrations of effluent TSS will also produce low-level concentrations of phosphorus.
4. *Breakthrough.* As documented above, the removal capability of the SSF wetland is primarily from sorption. When that capacity is used up, phosphorus passes through the system, and appears at the wetland outlet. Parameter estimation procedures will unavoidably characterize part of this pass through as a wetland background.

As a result of the breakthrough curve, the apparent $C^*$ value will depend upon system age, as well as on the incoming phosphorus concentration.

### Seasonal Effects

The growth and die-back of emergent wetland plants imposes a cycle of phosphorus uptake and release on subsurface flow wetlands. Although the phosphorus biomass cycle tends to be small relative to influent phosphorus loadings, most HSSF wetlands display some seasonal trend, as illustrated in Figure 10.35.

After the wetland aggregate has been saturated with sorbed phosphorus, weakly seasonal trends can be observed for a number of HSSF wetland systems. Table 10.15 summarizes results for two HSSF wetlands in the United Kingdom.

### Input–Output Relationships

Unlike FWS wetlands, HSSF and VF wetlands are rarely operated at low enough PLR that internal processes, such as plant biomass cycling and accretion, control effluent phosphorus concentrations. Most HSSF wetlands are designed for biochemical oxygen demand (BOD) and TSS removal, and most VF wetlands are designed for BOD, TSS, and ammonia removal. Phosphorus removal is usually not a primary design consideration, and examination of existing data sets reflects this.

#### HSSF Wetlands

The intersystem database that provides the basis for comparison includes annual performance summaries from 86 HSSF wetlands, representing 193 system-years of data. The most useful presentations involve effluent concentration (mg/L) versus influent loading ($g/m^2 \cdot d$), as shown in Figure 10.36.

The long-term effluent phosphorus concentrations produced in HSSF wetlands are a function of three primary variables (area, hydraulic loading, and influent concentrations). However, the sorption characteristics of the media are important factors that can be dominant mechanisms in treatment performance over the initial stage of operation until the sorption capacity is saturated. Sorption phenomena will be most pronounced in systems that use materials such as LECA or blast furnace slag where phosphorus retention is a design goal. However, as the period of record is relatively short for many HSSF wetlands, it is likely that sorption effects result in part of the "data scatter" represented in plots such as Figure 10.36.

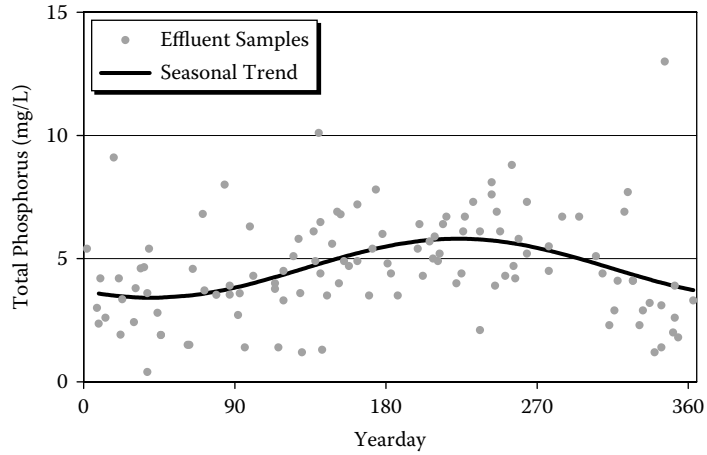



**FIGURE 10.35** Seasonal effluent phosphorus trends for a HSSF wetland near Londonderry, Northern Ireland. Figure represents six years of data; sample frequency is once every three weeks. (Data from CWA database (2006) *Constructed Wetlands Interactive Database, Version 9.02*. Compiled by G.D. Job and P.F. Cooper. United Kingdom Constructed Wetland Association (CWA), Gloucestershire, United Kingdom.) *Note*: U.K. site names are approximate.

An underlying assumption is that it is acceptable to combine the HLR and the influent concentration ($C_i$) to generate the phosphorus mass load (PLR), and that the PLR is related to the effluent phosphorus concentration ($C_o$). A graphical display has often been adopted in the literature (U.S. EPA, 2000a), sometimes with statistical intervals to represent variability in treatment performance (Wallace and Knight, 2006).

There is a severe potential drawback to combining data over many inlet concentration ranges to produce PLRs; this is illustrated in examining Figure 10.36 in more detail. Aggregation of the entire data set produces a data cloud that has a general trend line with a slope of about 1.0 (on a log–log plot). However, when PLRs are classified by influent concentration, as done in Figure 10.36, a different picture emerges. For any specific influent concentration range, the slope of the data cloud is about 0.2. This indicates that both $C_i$ and PLR affect effluent concentrations ($C_o$). At high PLRs, $C_o = C_i$ and there is no apparent removal. As the PLR is decreased (by making the wetland larger), $C_o$ decreases, but only within the context of the data subset defined by the $C_i$ range. In other words, wetlands that receive high influent concentrations ($C_i$) will still return high effluent concentrations ($C_o$), even at low PLRs. Wetlands with lower influent concentrations ($C_i$) will return lower effluent concentrations ($C_o$) when compared to systems with a higher $C_i$ range, even when they are loaded at the same PLR.

Therefore, it can be seen that PLR is not an independent variable to predict effluent phosphorus concentrations in HSSF wetlands. Ignoring the effect of $C_i$ when calculating PLRs leads to a potential design pitfall; combining all PLR data (regardless of $C_i$) leads to the assumption that low PLRs will result in low effluent concentrations; however, if the wetland in question receives a high $C_i$, it is unlikely that the expectation of low $C_o$ values will be achieved after the initial phase of active bed sorption.

Given the low slopes seen in the $C_o$–PLR relationship when it is broken down by influent concentration ranges ($C_i$), the conservative assumption at this stage of the technology development is that long-term, sustainable

**TABLE 10.15**
**Annual Trends for Influent and Effluent TP for Two HSSF Wetland Systems**

| System Location | Waste Stream | $C_{mean}$ | Amplitude, $A$ | $t_{max}$ | $R^2$ |
|---|---|---|---|---|---|
| Londonderry, Northern Ireland | Influent | 4.5 | 0.27 | 199 | 0.21 |
| | Effluent | 3.9 | 0.26 | 219 | 0.17 |
| Leicestershire, England | Influent | 6.6 | 0.32 | 158 | 0.10 |
| | Effluent | 3.2 | 0.08 | 226 | 0.02 |

*Note:* The trend in each time series is presumed to be sinusoidal according to

$$C = C_{mean}\left[1 + A \cdot \cos\left(\omega(t - t_{max})\right)\right] + E$$

(see text for notation); site names for U.K. systems are approximate.

*Source:* From CWA database (2006) *Constructed Wetlands Interactive Database*, *Version 9.02*. Compiled by G.D. Job and P.F. Cooper, United Kingdom Constructed Wetland Association (CWA), Gloucestershire, United Kingdom.)

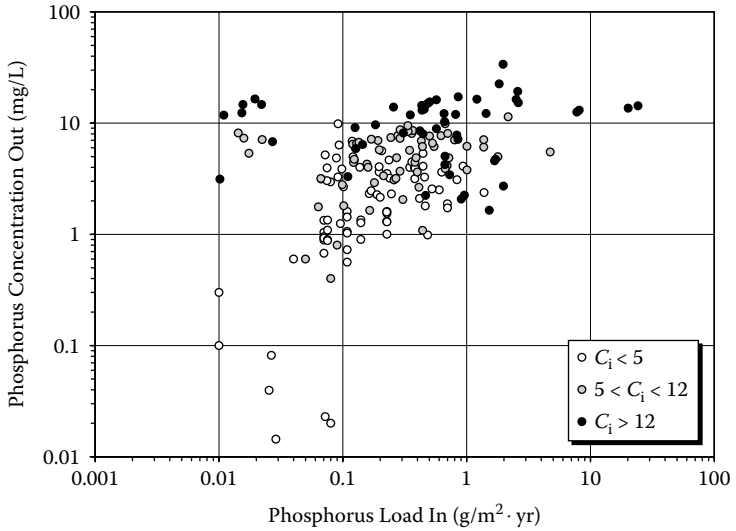



**FIGURE 10.36** Intersystem phosphorus responses for HSSF treatment wetlands. Each point represents one system-year of data (86 systems, 193 system-years total). Note that the influent phosphorus concentration ($C_i$) is a strong determinant of effluent concentrations, regardless of the phosphorus mass loading rate (PLR). Many of these systems were not at equilibrium conditions, and the bed materials were still actively sorbing phosphorus; this is the speculative explanation for low effluent concentrations ($C_o < 1$ mg/L) on this chart.

phosphorus removal in SSF wetlands is essentially zero ($C_o = C_i$).

If phosphorus retention is a performance expectation, this needs to be designed into the system using a bed media with favorable sorption characteristics (Table 10.14) and a modeling approach that approximates the hydraulic performance of the wetland (Figure 10.30 and Figure 10.31). It should be noted that SSF wetlands have been successfully used for phosphorus retention, especially in Norway (Jenssen and Krogstad, 2003), but they all employ the bed sorption approach just described. Chemical precipitation remains a speculative means to sustainable removal (Molle *et al*., 2005b), but has yet to be demonstrated in full-scale SSF wetland designs.

### VF Wetlands

At present, there is insufficient data to represent phosphorus removal in VF wetlands. As VF wetlands operate at even higher hydraulic and influent loading rates than HSSF wetlands, the short-term, finite phosphorus reservoirs, such as aggregate sorption and biomass cycling, play an even smaller role in VF systems. Because long-term, low-rate phosphorus removal mechanisms between the two wetland types are similar, use of Figure 10.36 is recommended until a more complete database on VF wetland performance can be established.

#### Stochastic Variability

Similar to FWS wetlands, there are two components to the effluent phosphorus concentration, a long-term trend plus a stochastic component. This stochastic variability ($E$) can be represented in the effluent by a multiplier on the effluent trend concentration (Equation 10.14). Because effluent phosphorus concentrations are strongly related to internal processes, random physical and biological events have pronounced effects on effluent concentrations. These effects, by definition, are random and cannot be predicted based on the current state of the art in wetland science.

However, stochastic variations typically display a limited range of variability. By combining a large number of observations on a given wetland system, statistics can be used to quantify the range of variability within the wetland system. Similarly, by comparing system performance between wetland systems, statistics can be used to assess intrasystem performance.

Treatment wetlands are commonly regulated systems, and have a specified effluent concentration, and an allowable excursion frequency, determined by permit. Quantifying the range of variability displayed by stochastic events is very useful in wetland design, as this allows the designer to assess what range of variability the effluent data is likely to display. Trend multipliers ($1 + \Psi$) for four HSSF wetlands are summarized in Table 10.16. These data indicate that the mean of the 95% percentile is a 1.93 multiplier on the long-term median effluent concentration. This information may be needed in design.

## SUMMARY

Constructed and natural wetlands are capable of absorbing new phosphorus loadings, and in appropriate circumstances can provide a low-cost alternative to chemical and biological treatment. Phosphorus interacts strongly with wetland soils, bed aggregates, and biota, which provides both short-term and sustainable long-term storage of this nutrient. Soil

**TABLE 10.16**
**Trend Multipliers for Effluent TP Concentrations in HSSF Wetlands in the United Kingdom**

| Location | Median Inlet (mg/L) | Median Outlet (mg/L) | Excursion Frequency 80% | 90% | 95% |
|---|---|---|---|---|---|
| Londonderry, Northern Ireland | 4.5 | 3.9 | 1.46 | 1.66 | 1.93 |
| Londonderry, Northern Ireland | 4.3 | 4.0 | 1.49 | 1.85 | 2.16 |
| Leicestershire, England | 6.6 | 3.2 | 1.21 | 1.36 | 1.69 |
| North Yorkshire, England | — | 6.9 | 1.36 | 1.75 | 1.94 |
| **Mean** | | | **1.38** | **1.66** | **1.93** |

*Note:* Trend multiplier distributions represent 295 measurements over four wetland systems; site names for U.K. systems are approximate. Trend multiplier is $(1 + \Psi)$; see Equation 6.61.

*Source:* From CWA database (2006) *Constructed Wetlands Interactive Database*, *Version 9.02*, Compiled by G.D. Job and P.F. Cooper, United Kingdom Constructed Wetland Association (CWA), Gloucestershire, United Kingdom.)

sorption may provide initial removal, but this partly reversible storage eventually becomes saturated. Uptake by biota, including bacteria, algae, and duckweed, as well as macrophytes, forms an initial removal mechanism. Cycling through growth, death, and decomposition returns most of the biotic uptake, but an important residual contributes to long-term accretion in newly formed sediments and soils. Despite the apparent complexity of these several removal mechanisms, data analysis shows that relatively simple equations can describe the sustainable processes. A first-order removal rate model can characterize removal in FWS wetlands, but does not incorporate any biotic features.

Phosphorus removal has been documented for over 250 FWS wetlands, for inflow concentrations in a wide range, from below 20 µg/L to over 100 mg/L. As inflow concentrations increase, the phosphorus load removal also increases, from less than 0.1 to well over 100 $g/m^2 \cdot yr$. Both transient and sustainable removal processes occur, with periods of initial adaptation of up to several years. Similarly, a wetland may exhibit a slow response (one to two years) to changing inlet conditions.

Wetland water temperature appears to be only a minor modifying factor on phosphorus sequestration, and in many cases, seasonality is also of small consequence. Nevertheless, in some instances, winter performance is lower compared to warmer seasons, for reasons that are not fully understood. The absence of vegetation lessens removal capability. Further, the community type in FWS wetlands (EAV, SAV) has significant effects, but not in all concentration ranges. However, species composition of a given community type appears to be a relatively minor factor.

Simple, highly aggregated models can explain a good deal of both intrasystem and intersystem variations. However, such deterministic models are not capable of describing the purely stochastic variations in phosphorus removals. That probabilistic component of behavior can be adequately described using statistical descriptors.

Experiences of the last three decades indicate that FWS wetlands can fulfill a useful role in phosphorus reduction in many situations. Improvements in water quality for secondary and tertiary effluents are possible, but there is perhaps an even greater role in controlling nutrients in urban and agricultural runoff. This positive connotation must, however, be tempered by the large land requirement that usually accompanies the use of a passive wetland to immobilize phosphorus. Because phosphorus is a nutrient, there is not a perceived threat due to toxicity within a treatment wetland of this sort. However, the eutrophication caused by trapping phosphorus within a wetland means that not all biodiversity goals will necessarily be met for that wetland.

In contrast, subsurface flow (HSSF and VF) wetlands are rarely designed with phosphorus retention as a primary performance objective. The long-term, sustainable mode of performance is one of essentially zero phosphorus removal. When phosphorus retention is a design goal, researchers have focused on bed materials (lightweight aggregates, industrial byproducts) that have high phosphorus sorption capacities. This maximizes the short-term, finite sorption reservoir within the wetland. Because the bed has a finite sorption capacity, wetlands designed using this approach are by nature sacrificial and will require periodic excavation and bed replacement as a result.

Once short-term phosphorus retention reservoirs have been saturated, phosphorus removal proceeds, albeit at a much lower rate, through the sustainable mechanisms of chemical precipitation, accretion, and particulate settling. These low-rate mechanisms contribute to the accumulation of solids within the wetland bed, which can lead to a loss of pore volume and hydraulic conductivity. As a result, the wetland bed characteristics (media size, porosity, and hydraulic conductivity) set upper limits in the extent of phosphorus storage through the accumulation of particulate matter.

# 11 Halogens, Sulfur, Metals, and Metalloids

In addition to the pollutants discussed in earlier chapters, wastewaters typically contain many other substances. Some of these elements can cause problems when discharged to receiving waters, and their removal must be considered during design. These additional materials include salts, acids, bases, macronutrients, micronutrients, and heavy metals, and may be categorized in a number of ways. Salts include compounds that readily dissociate in water to form charged ions that may or may not be used as nutrients for plant and animal growth. Common examples of salts are sodium chloride (NaCl) and gypsum ($CaSO_4$). Acids release a hydrogen ion when they dissociate (e.g., hydrochloric acid, HCl), and bases release a hydroxyl ion (e.g., ferric hydroxide—$Fe(OH)_3$). Specific environmental conditions determine whether the cations (positively charged ions) and anions (negatively charged ions), formed when a salt, acid, or base is dissolved in water, are chemically or biologically active. Collectively, ionic materials contribute to the electrical conductivity (EC) of the water. When ionic materials are combined with dissolved nonionic materials, the result is the total dissolved solids (TDS) content of the water.

Nitrogen and phosphorus, discussed in Chapters 9 and 10, are examples of macronutrients, which have strong biogeochemical cycles in a wetland. Sulfur also is typically present in variable but potentially high concentrations, and has just as powerful influences on wetland functioning. The magnitude of these influences is just emerging as a controlling factor on wetland performance for a number of other pollutants. Most obvious is the role of sulfides in immobilizing trace metals.

Iron, aluminum, and manganese are ubiquitous in wetlands, but are present at elevated concentrations in mine drainage waters and the wetlands constructed to treat them. A trace metal can be either a required micronutrient or toxic, depending on the concentration. For example, copper and zinc are essential elements for plants and animals at low concentrations, but they are toxic to some organisms at elevated concentrations. However, for some trace metals, such as cadmium and lead, essentiality for plants or animals has never been found. In this chapter, many of the important elements are discussed, but the list does not include all the elements that may be found in waters, or all those that might require treatment. The use of a wetland treatment system to modify the concentration of elements depends on how the elements interact with the wetland environment and on the wetland designer's knowledge of design factors that can enhance or diminish these processes. There is a rapidly growing body of knowledge about how wetland treatment systems affect specific trace elements. A thorough, updated review of the scientific literature is recommended for project design.

A number of substances are considered measures of water quality, but are seldom of concern as pollutants to be treated in constructed wetlands. These include common metals (e.g., sodium, potassium, calcium, and magnesium) as well as halogens (e.g., fluorine, chlorine, bromine, and iodine). Together with sulfate, these compounds often dominate the total ion content of natural waters and wastewaters. In total, they form the major part of EC and TDS. Sulfate is of special importance, because of its active biogeochemical cycle, and interactions with trace metal removal. In some treatment wetlands, several of these collective water quality parameters have become important in their own right.

## 11.1 HALOGENS

Chloride and bromide are widely regarded as being "conservative" in wetland environments, meaning that they interact with the ecosystem to a very limited extent. Therefore, they can be used as tracers of water movement in the wetland. Usually, chloride is present at concentrations that preclude its use as an injected tracer, but it sometimes serves as a means of confirming the wetland water budget. Fluoride is usually a very minor trace constituent in aquatic systems, but there are industrial effluents that contain relatively high concentrations. The aluminum industry is one such source, including leachates from solid waste disposal sites. Bromide is often present at low background concentrations, and injected bromide may then be used to trace internal water movements. Very little is known about the fate and transport of iodine in freshwater systems.

### Chloride and Chlorine

U.S. EPA (2002a, 2006) sets a criteria maximum concentration (CMC), which is an estimate of the highest concentration of a material in surface water to which an aquatic community can be exposed briefly without resulting in an unacceptable effect. The value for chloride is 860 mg/L, and for chlorine is 19 µg/L. U.S. EPA (2002a, 2006) also sets a criterion continuous concentration (CCC), which is an estimate of the highest concentration of a material in surface water to which an aquatic community can be exposed indefinitely without resulting in an unacceptable effect. The CCC value for chloride is 230 mg/L, and for chlorine is 11 µg/L.

The chlorine content of wetland plant tissues has not been measured often. Results from two projects are shown in

**TABLE 11.1**
**Chlorine and Fluorine Concentrations in Plant Tissues at Oxnard, California**

| | Chlorine | | | Fluorine | | |
|---|---|---|---|---|---|---|
| **Plant** | **Water** | **Shoot** | **Root** | **Water** | **Shoot** | **Root** |
| *Typha latifolia* | 300–340 | 8,910 | 7,650 | 2.5–2.8 | 352 | 155 |
| *Scirpus americanus* | 300–340 | 19,100 | 2,920 | 2.5–2.8 | 232 | 343 |
| *Juncus balticus* | 300–340 | 6,040 | 2,490 | 2.5–2.8 | 280 | 453 |
| *Anemopsis california* | 300–340 | 28,600 | 7,080 | 2.5–2.8 | 3,290 | 160 |
| *Jaumea carnosa* | 300–340 | 36,500 | 5,330 | 2.5–2.8 | 690 | 389 |
| *Distichlis spicata* | 300–340 | 5,820 | 4,470 | 2.5–2.8 | 336 | 518 |
| *Potemogeton pectinatus* | 300–340 | 4,910 | 9,910 | 2.5–2.8 | 134 | 389 |
| *Salicornia virginica* | 300–340 | 67,300 | 7,570 | 2.5–2.8 | 402 | 110 |
| *Monothochlore littoralis* | 300–340 | 4,400 | 1,360 | 2.5–2.8 | 204 | 213 |

*Note:* Units are mg/kg for tissues, and mg/L for water.

*Source:* Data from CH2M Hill (2005) *Additional testing for the Membrane Concentrate Pilot Wetlands Project.* Report to the City of Oxnard Water Division, Oxnard, California, United States.

Tables 11.1 and 11.2. The Oxnard, California, systems were exposed to high chloride, and developed high leaf tissue concentrations (5–40 g/kg dry weight, or 0.5–4.0%). Presumably, much of the chlorine associated with the dry matter was originally in solution in the plant water content, which is 70–80% of the wet weight. Therefore sap concentrations would be about five times lower. Interestingly, *Salicornia* spp. is a hyperaccumulator of chlorine (Table 11.1). It is also notable that roots contain much less chlorine than the shoots. Standing dead and litter of *Typha latifolia* were found to contain much less chlorine than live leaves at the Houghton Lake, Michigan, treatment wetland (Table 11.2).

Chlorine is biologically interactive in wetland ecosystems. It is influential in the osmolality salinity balance, but metabolic utilization does not usually cause changes in water concentrations (Wetzel, 1983). However, there are circumstances in which utilization can be measured. Xu *et al.* (2004) measured chloride and sulfate profiles in vertical flow mesocosms with *Typha latifolia* in sand, during growth of the plants. The hydraulic loading rate was low (0.66 cm/d), and consequently, transpiration was an important effect (about 0.3 cm/d). Sulfate was added at concentrations far in excess of any potential plant requirements (80 mg/L $SO_4$-S). As the water traversed the root zone, sulfate concentrations increased to about double their inlet value, which was strictly attributed to transpiration losses. No increase occurred in unvegetated controls. However, the profiles of chloride were very different: virtually all of the inlet chloride was absorbed in the mesocosms, from a starting concentration of about 5.0 mg/L. Given the biomass increase of the plants, chloride removal would have produced a chloride content of the cattails of about 4,000 mg/kg, which is at the low end of the range measured for *Typha* (Tables 11.1 and 11.2).

The more typical situation is an overabundance of chloride entering the wetland. Because of the relatively low biological demand for chloride, the total chloride mass is usually relatively constant between the inflows and outflows and storages of a treatment wetland (Table 11.3). Therefore, the wetland chloride mass balance can be used to confirm the water budget.

**TABLE 11.2**
**Halogen Content of Biomass Compartments for *Typha latifolia* in the Houghton Lake, Michigan, Treatment Wetland in 1991**

| | Water | Live Leaves | Standing Dead | Litter | Roots + Soil |
|---|---|---|---|---|---|
| **Control Zone** | | | | | |
| Chlorine | 5–15 | 13,300 | 195 | 727 | 300 |
| Bromine | — | 25 | 2 | 43 | 37 |
| Iodine | — | — | 3 | 13 | 9 |
| **Discharge Zone** | | | | | |
| Chlorine | 100–125 | 34,700 | 369 | 1,105 | 858 |
| Bromine | — | 46 | 11 | 50 | 51 |
| Iodine | — | — | — | 17 | 11 |

*Note:* Units are mg/kg for tissues, and mg/L for water.

*Source:* Unpublished data.

**TABLE 11.3**
**Examples of Conservative Materials Entering and Leaving FWS Treatment Wetlands**

| System | Wetland | Years | HLR (m/yr) | EC In (µS/cm) | EC Out (µS/cm) | Chloride In (mg/L) | Chloride Out (mg/L) | TDS In (mg/L) | TDS Out (mg/L) | Reference |
|---|---|---|---|---|---|---|---|---|---|---|
| **Pass-Through** | | | | | | | | | | |
| Tres Rios, Arizona | Hayfield 1 | 8 | 41.2 | 1,559 | 1,497 | 259 | 257 | 902 | 906 | Unpublished data |
| Tres Rios, Arizona | Hayfield 2 | 8 | 55.8 | 1,558 | 1,511 | 254 | 253 | 913 | 924 | Unpublished data |
| Tres Rios, Arizona | Cobble 1 | 8 | 119.5 | 1,500 | 1,494 | 264 | 272 | 934 | 940 | Unpublished data |
| Tres Rios, Arizona | Cobble 2 | 8 | 65.2 | 1,500 | 1,488 | 263 | 264 | 934 | 949 | Unpublished data |
| Brawley, California | All | 4 | 33.9 | 4,503 | 4,713 | — | — | — | — | Unpublished data |
| Imperial, California | All | 4 | 24.9 | 2,718 | 2,642 | — | — | — | — | Unpublished data |
| Orlando Easterly, Florida | WP1–MM7 | 10 | 11.5 | 550 | 522 | 66 | 65 | — | — | Unpublished data |
| Estevan, Saskatchewan | All | 10 | 10.5 | 2,332 | 2,457 | 198 | 201 | 1,644 | 1,722 | Unpublished data |
| Isanti-Chisago, Minnesota | All | 3 | 36.5 | 1,367 | 1,147 | 79 | 76 | 770 | 760 | Kadlec (2003c) |
| ENRP, Florida | All | 6 | 11.3 | 1,033 | 1,037 | 160 | 167 | — | — | Unpublished data |
| Connell, Washington | Full scale | 1 | 21.8 | 2,890 | 2,740 | — | — | — | — | Unpublished data |
| Purdue University, Indiana | 15 wetlands | 2 | 15.5 | 4,658 | 3,788 | — | — | — | — | WERF database (2006) |
| Newton, Mississippi | 12 wetlands | 2 | — | — | — | — | — | 710 | 639 | WERF database (2006) |
| Richmond, New South Wales | *Myriophyllum* | 2 | 26.8 | 772 | 707 | — | — | — | — | Bavor *et al.* (1988) |
| Benton, Kentucky | Cattail | 1 | 19.7 | 324 | 362 | — | — | — | — | TVA (1990) |
| Benton, Kentucky | Woolgrass | 1 | 20.8 | 324 | 348 | — | — | — | — | TVA (1990) |
| Albright, West Virginia | All | 10 | 26.4 | — | — | — | — | 1,334 | 1,164 | Hoover *et al.* (1998) |
| Springdale, Pennsylvania | All | 10 | 35.8 | — | — | — | — | 1,818 | 1,828 | Hoover *et al.* (1998) |
| Oxnard, California | Train 1 | 1 | 19.8 | 4,500 | 4,650 | 215 | 255 | 4,200 | 4,200 | CH2M Hill (2005) |
| Pensacola, Florida | 6 pulp and paper | 2 | 18.4 | 2,125 | 1,931 | — | — | 1,458 | 1,385 | NADB database (1998) |
| Boney Marsh, Florida | River | 8 | 7.4 | — | — | 16.5 | 15.9 | — | — | Unpublished data |
| **Anomalies** | | | | | | | | | | |
| New Hanover, North Carolina | Full scale | 1 | 0.79 | 6,256 | 1,468 | 2,753 | 411 | 5,742 | 1,286 | Unpublished data |
| Incline Village, Nevada | Full scale | 5 | 0.54 | — | — | 39 | 155 | 269 | 2,464 | Kadlec *et al.* (1990) |
| Ouray, Colorado | Full scale | 5 | 95 | — | — | — | — | 326 | 685 | HDR/ERO (2001) |

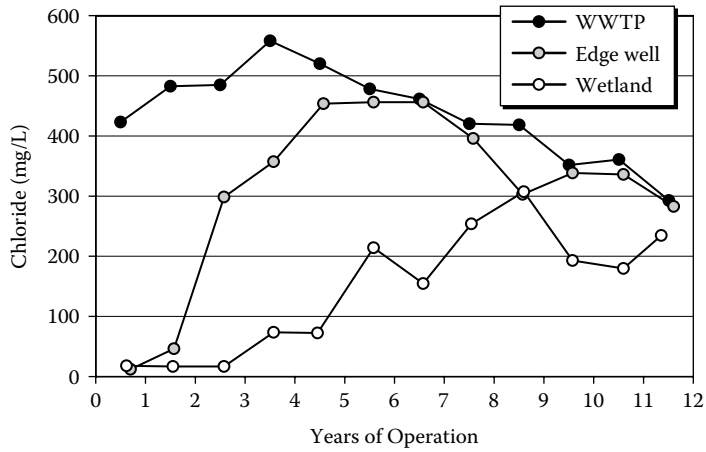



**FIGURE 11.1** Progress of chloride from the infiltration beds of the wastewater treatment plant (WWTP), underground to a well on the wetland edge, and then out into the wetland at Genoa-Oceola, Michigan. (From unpublished data.)

Chloride can serve as a tracer of water movement, especially in the analysis of the very slow underground movement of water. For instance, the progress of a chloride front from rapid infiltration basins, underground to a monitoring well on the edge of the receiving wetland, and then out into that wetland, is shown in Figure 11.1. The wastewater treatment plant at Genoa-Oceola, Michigan, received very high chlorides (about 400–550 mg/L) because of the widespread use of water softeners in the region. The treated water was discharged onto rapid infiltration basins on a hilltop adjacent to a wetland. The hydrologic gradient moved the water slowly toward the wetland, and the high chloride arrived in wells at the wetland edge after about three years. After about six years, the wetland surface waters reflected the chloride of the wastewater, with some dilution from the other flows in the aquifer.

## Disinfection: Chlorine in Wetlands

Free chlorine is toxic to most life forms, and is one of the most frequently used wastewater disinfectants. There are implications for treatment wetlands that receive chlorinated effluents, because the residual toxicity may negatively influence the microbial communities within the system.

Some of the free chlorine added during disinfection is converted in solution to chlorides or chloramines, the latter being regarded as an undesirable pollutant. The products that result from disinfecting water by the addition of chlorine are:

- Free residual chlorine—the portion of chlorine remaining as molecular chlorine, hypochloride (HOCl), or hypochlorite ion ($OCl^-$)
- Combined residual chlorine—the portion of chlorine that combines with ammonia or nitrogenous compounds, forming chloramines
- Total residual chlorine (TRC)—the sum of free residual plus the combined residual chlorine

Chlorine reduction in wetlands can occur via several pathways, such as photodegradation, and reaction with organic and inorganic material in the water, as well as ammonia. Wetlands have enough organic matter to promote formation of trihalomethanes (THM) (Gallard and von Gunten, 2002), but other organic halides may also form. Total organic halides (TOX) and halo-acetic acids (HAA) may also form (Rostad *et al.*, 2000). The photochemical process is initiated by production of oxidants such as peroxides. These oxidants then oxidize the chlorinated compounds. Volatilization, adsorption, and interactions with aquatic plants and the soil system may also contribute to the decay of residual chlorine.

Studies of the loss of TRC were conducted in the Tres Rios, Arizona, FWS wetlands (Wass, Gerke, and Associates, 2004). Disappearance was approximated by first-order behavior. The rate constants were calculated to be 0.86 $d^{-1}$ for the Cobble C1 and Hayfield H1 wetlands, based on transect data. Thus there would be more than 90% reduction in TRC for a three-day detention time. Surveys of organo-chlorine compounds in the wetlands showed decreasing gradients from inlet to outlet for TOX, THM, and HAA (Table 11.4).

### Bromide and Bromine

Bromide is not commonly measured as a constituent of natural freshwaters or wastewaters. It is a common choice for a water movement tracer, and a number of studies have therefore determined the background bromide in treatment wetland waters. Example values were 0.2–0.3 mg/L at Tres Rios, Arizona; 0.13–0.18 mg/L at Orlando Easterly, Florida; 0.05–0.15 mg/L at Hillsdale, Michigan; and 0.3–0.4 mg/L at Des Plaines, Illinois. Bromine is, therefore, also found in only minor trace amounts in vegetation. Concentrations of 25–46 mg/kg dry mass were measured in *Typha latifolia* at the Houghton Lake, Michigan, treatment wetland. Parsons et al. (2004) added a uniform sudden dose of bromide to

**TABLE 11.4**
**Inputs and Outputs of Chlorination Byproducts in the Tres Rios, Arizona, Demonstration Wetlands (µg/L)**

| Location | TOX | THM | HAA |
|---|---|---|---|
| Input | 170 | 11.0 | 31.0 |
| C1 Out | 76 | 0.3 | 7.2 |
| C2 Out | 97 | 1.4 | 8.2 |
| Input | 140 | 11.0 | 31.0 |
| H1 Out | 101 | 1.8 | 8.7 |
| H2 Out | 127 | 1.8 | 11.0 |
| Input | 178 | 11.0 | 70.0 |
| R1-R12 Out | 123 | 1.4 | 10.9 |

*Note:* TOX = total organic halides; THM = trihalomethanes; HAA = halo-acetic acids

*Source:* Adapted from Rostad *et al.* (2000) *Environmental Science and Technology*, 34: 2703–2710.

establish an initial surface water concentration of 100 mg/L in a prairie pothole in Saskatchewan. The wetland had no surface inflow or outflow, and thus the dose remained confined to the system except for infiltration. They measured 20–90 mg/kg dry mass in aboveground tissues of *Heracleum lanatum*, *Polygonum* spp., and *Carex* spp., with localized values ranging up to 12,700 mg/kg dry mass. The fraction of the dose retained in vegetation was estimated to be 8.7%, as determined by areal averaging for the diverse species.

Bromide sorbs to soils to about the same extent as nitrate (Clay *et al.*, 2004), which is negligible in most situations. Bromine does not have a role in plant metabolism; however, bromide can be taken up by plants, to help satisfy the anionic component of the charge balance in the plant internal water. In that respect, bromide competes with chloride, as documented by Xu *et al.* (2004) in *Typha* and *Phragmites* systems. Plant uptake is, therefore, presumably greatest during the growing season, during which new plant water is building within the wetland.

### Fluoride and Fluorine

Fluorine in water exists primarily in the form of sodium and calcium salts. Calcium fluoride is used as a flux in steel manufacturing. Sodium fluoride is used as a drinking water additive for prevention of dental cavities (tooth decay). The recommended optimum level ranges from 0.7 mg/L for warmer climates to 1.2 mg/L for cooler climates. Median concentrations are 0.2 mg/L in surface water and 0.1 mg/L in groundwater (U.S. EPA, 2002a). The current Maximum Contaminant Level set by the U.S. EPA in 1986 is 4 mg/L. Fluoride levels typically range from 0.03 to 0.57 mg/L in eastern U.K. rivers (Neal *et al.*, 2003a).

Fluoride differs from chloride and bromide, because it has been a target of treatment wetland design. The aluminum industry relies upon molten salt electrolytes that contain fluorides. Solid wastes from the industry are usually landfilled, and produce leachates that contain elevated concentrations of fluoride (up to 100 mg/L).

Fluoride partitions more strongly to soils and sediments than do bromide and chloride. The Langmuir adsorption capacities of soils ranges from 100–400 mg/kg for silts and loams (Bower and Hatcher, 1967). However, the oxyhydroxides of iron and aluminum have much higher binding capacities, 30,000–50,000 mg/kg. This property causes fluoride to be a poor tracer. For instance, LeBlanc *et al.* (1991) found: "Fluoride was abandoned as a tracer early in the test because fluoride concentrations were rapidly attenuated by adsorption…" in the mineral soils of the site under study. Similarly, Jamieson *et al.* (2002) found only 57% recovery on a fluoride tracer test of a dairy wastewater treatment wetland in Nova Scotia.

Fluorine is taken up by plants to a moderate extent (see Table 11.1), with tissue concentrations typically in the 100–500 mg/kg range. But because it is not a macronutrient or a micronutrient, it is probable that such uptake is driven by the plant water ionic balance. The result of limited uptake and sorption is a limited overall reduction of fluoride in treatment wetlands (Table 11.5). Data are too sparse to determine whether seasonal effects are present, or to elucidate possible differences between wetland types or plant varieties.

**TABLE 11.5**
**Example Performances of Treatment Wetlands for Fluorine (mg/L)**

| Location | Water Treated | Inlet | Outlet | Reference |
|---|---|---|---|---|
| Brookhaven, New York | Domestic | 0.24 | 0.24 | NADB database (1998) |
| Tucush, Peru | Metal mine | 0.24 | 0.17 | Unpublished data |
| Imperial, California | Agricultural runoff | 0.54 | 0.56 | Unpublished data |
| Brawley, California | River | 0.71 | 0.83 | Unpublished data |
| Oxnard T1, California | Backwash | 2.80 | 2.90 | CH2M Hill (2005) |
| Oxnard T2, California | Backwash | 2.80 | 3.10 | CH2M Hill (2005) |
| Alcoa, Tennessee | Aluminum waste leachate | 5.80 | 4.90 | Gessner *et al.* (2005) |
| Russelville, Kentucky | Aluminum processing | 15.4 | 8.50 | Rowe and Abdel-Magid (1995) |
| Lambton, Ontario | Gypsum leachate | 16.0 | 15.3 | Unpublished data |
| Australia | Power station | 11.4 | 11.7 | Jensen *et al.* (2006) |

## 11.2 ALKALI METALS

Sodium, potassium, calcium, and magnesium are rarely the object of regulatory concern, because under most circumstances they do not pose any toxicity threat. Nevertheless, each of these has a role in wetland functioning, and can yield valuable information about pollutant processing and the wetland water budget.

### Sodium

Sodium is important in plant and animal physiology. Sodium ions help to regulate osmotic pressure in cells, and therefore affect the diffusion of all essential growth nutrients between the external environment and the protoplasm of the living cells. A "sodium pump" fueled by the conversion of energy-bearing adenosine triphosphate (ATP) maintains internal cell sodium concentrations at optimal levels. The sodium content of wetland plant aboveground tissues ranges from <0.05% to more than 1.3% dry weight (Table 11.6). The median across the 13 species is 0.28%, or 2,800 mg/kg.

Because most freshwater wetland species have low sodium requirements, the dissolved sodium content of wastewater passing through wetlands changes little (Table 11.7). Thus, sodium concentrations can be used as a conservative tracer for calculating dilution and concentration and for tracking groundwater discharges from wetlands. For instance, the concentration of sodium in arid land treatment wetlands is likely to increase during the summer season due to evaporative concentration.

Sodium is useful as a marker for added salt (NaCl), which may enter wastewater treatment systems because of its use in water softening and road de-icing. For example, the Cumberland County, Pennsylvania, data in Table 11.7, show a large increase in sodium during a spring flushing event, which was also accompanied by a large pulse of chloride (data not shown: $C_i$ = 14 mg/L and $C_o$ = 140 mg/L). The origins of the sodium may have been the accumulation of road de-icing salt, contributed by the highway runoff the wetland was designed to treat. At the Genoa-Oceola, Michigan, site, discussed in the section on chloride, the water softener salt also had elevated sodium (150 mg/L), compared to the wetland background of about 5 mg/L. The underground plume reached the wetland after about six years, at which time the wetland surface water sodium had increased to 95 mg/L.

### Potassium

Ionic pumping maintains potassium levels in plants at concentrations of 1.0–4.0%. Potassium regulates the opening and closing of stomata on plant leaves. Stomata are the valves that allow gases inside the plant to be exchanged with the atmosphere. Potassium also is used as an enzyme activator in protein synthesis in most cells. Potassium typically comprises about 2.6% of the dry weight of wetland plants. Potassium concentrations in water of surface flow treatment wetlands are typically between 1.0 and 40 mg/L (Table 11.7), with an average world river concentration of about 3.4 mg/L (Hutchinson, 1975). Potassium has not been the target of treatment wetland design. In general, there is not much change in potassium from wetland inlet to outlet (Table 11.7).

### Calcium

Calcium is biologically active because it is used as a nutrient by invertebrates and vertebrates, and because of its role in the carbonate cycle. Calcium is required by, and present in sizeable amounts in, angiosperm plants (Vymazal, 1995). The median concentration in a variety of wetland plants is

**TABLE 11.6**
**Examples of Major Ion Content of Wetland Plants**

| Plant Type | Sodium (% dw) | Potassium (% dw) | Calcium (% dw) | Magnesium (% dw) |
|---|---|---|---|---|
| *Typha latifolia* | 0.28 | 2.65 | 0.76 | 0.15 |
| *Juncus effusus* | 0.40 | 0.89 | 0.38 | 0.11 |
| *Phragmites australis (W)* | 0.06 | 2.60 | 0.17 | 0.08 |
| *Glyceria maxima (W)* | 0.12 | 0.81 | 0.48 | 0.15 |
| *Scirpus americanus* | 0.09 | 2.83 | 0.50 | 0.22 |
| *Sagittaria latifolia* | 0.14 | 4.04 | 0.55 | 0.18 |
| *Nymphaea odorata* | 1.35 | 1.28 | 1.06 | 0.14 |
| *Nelumbo lutea* | 0.66 | 0.99 | 1.79 | 0.26 |
| *Ceratophyllum demersum* | 1.16 | 4.01 | 0.77 | 0.42 |
| *Najas guadalupensis* | 0.61 | 3.49 | 0.98 | 0.47 |
| *Myriophyllum heterophyllum* | 1.30 | 3.03 | 0.88 | 0.26 |
| *Eichhornia crassipes* | 0.01 | — | 6.12 | 3.90 |
| *Pistia stratiotes* | 0.02 | — | 4.46 | 5.25 |
| **Median** | **0.28** | **2.65** | **0.77** | **0.22** |

*Note:* The letter "W" denotes a treatment wetland.

*Sources:* Data from Boyd (1978) In *Freshwater Wetlands: Ecological Processes and Management Potential*. Academic Press, New York, 155–167; and Vymazal (1995) *Algae and Nutrient Cycling in Wetlands*. CRC Press/Lewis Publishers, Boca Raton, Florida, 1995.

**TABLE 11.7**
**Examples of Major Cations Entering and Leaving Treatment Wetlands**

| | | | | Sodium | | Potassium | | Calcium | | Magnesium | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| System | Wetland | Years | HLR (m/yr) | In (mg/L) | Out (mg/L) | In (mg/L) | Out (mg/L) | In (mg/L) | Out (mg/L) | In (mg/L) | Out (mg/L) | Reference |
| Estevan, Saskatchewan | All | 10 | 10.5 | 353 | 364 | 25.7 | 23.5 | 87 | 90 | 65 | 68 | Unpublished data |
| ENRP, Florida | All | 6 | 11.3 | 110 | 119 | — | — | 84 | 69 | — | — | Unpublished data |
| Oxnard, California | Train 1 | 1 | 19.8 | 405 | 465 | 20 | 20 | 490 | 475 | 210 | 245 | CH2M Hill (2005) |
| Imperial, California | All | 4 | 24.9 | 371 | 348 | 11.2 | 8.9 | 182 | 160 | 82 | 75 | Unpublished data |
| Brawley, California | All | 4 | 33.9 | 698 | 774 | 18.6 | 18.7 | 181 | 183 | 88 | 93 | Unpublished data |
| Columbia, Missouri | All | 3 | 49.0 | — | — | 36.6 | 32.8 | — | — | — | — | Unpublished data |
| Musselwhite, Ontario | All | 4 | 190 | 55 | 71.2 | 19.0 | 22.5 | 93.8 | 112.5 | 9.63 | 10.5 | Bishay and Kadlec (2005) |
| Monroe County, New York | FWS | 2 | 3.5 | 410 | 430 | 269 | 196 | 180 | 70 | 160 | 150 | Eckhardt *et al.* (1999) |
| Hidden River, Florida | Urban runoff | 2 | 3.8 | 0.477 | 0.828 | 0.069 | 0.106 | 7.08 | 8.35 | 0.094 | 0.349 | Carr and Rushton (1995) |
| Pensacola, Florida | 6 cells pulp and paper | 2 | 18.4 | — | — | — | — | — | — | 9.3 | 7.3 | NADB database (1998) |
| Brookhaven, New York | MMP | 2 | 5.48 | 19.97 | 19.05 | 4.42 | 3.16 | 25.52 | 16.69 | 4.58 | 4.76 | Hendrey *et al.* (1979) |
| Boney Marsh, Florida | River | 8 | 7.4 | 9.79 | 10.28 | 1.2 | 1.26 | 13.83 | 13.77 | 2.73 | 2.4 | Unpublished data |
| Norco, Louisiana | Refinery, West Cell | 1 | 17.8 | 360 | 430 | 8.4 | 9.1 | 140.7 | 46.7 | 23.4 | 16.7 | Hawkins *et al.* (1997) |
| West Lafayette, Indiana | Urban stormwater | 1 Event | — | 110 | 85 | 2.5 | 2.7 | 95 | 54 | 28 | 15 | Harbor *et al.* (2000) |
| Cumberland County, Pennsylvania | Highway runoff | 1 Event | — | 28.1 | 62.7 | 6.54 | 7.93 | 16.6 | 16.2 | 1.15 | 1 | Nelson *et al.* (2002) |
| San Tomé, Argentina, FWS | Tool factory | 3 | 18.3 | — | — | — | — | 175 | 86 | 17.2 | 16.6 | Maine *et al.* (2006) |
| Mořina, Czech Republic, HSSF | Municipal sewage | 1 | 9.3 | 127 | 102 | 19 | 20 | 98 | 89 | 21 | 19 | Unpublished data |
| Břehov, Czech Republic, HSSF | Municipal sewage | 1 | 24.6 | 39 | 31 | 62 | 49 | 42 | 33 | 18 | 13 | Unpublished data |
| Slavošovice, Czech Republic, HSSF | Municipal sewage | 1 | 9.8 | 43 | 16 | 36 | 18 | 41 | 17 | 16 | 7.4 | Unpublished data |
| Australia | Power station | 6 | 24.3 | 238 | 245 | — | — | — | — | — | — | Jensen *et al.* (2006) |

0.77% dry mass (see Table 11.6), and is similar in freshwater planktonic algae. However, levels in floating plants and filamentous green algae range upward to 5–7% dry mass (see Table 11.6; Vymazal, 1995). The photosynthetic organs of plants and algae may develop calcium carbonate (calcite) encrustations in hard water environments. Because there is generally an excess of calcium in surface water and wastewater, calcium concentration does not change appreciably in many wetland treatment systems (see Table 11.7).

In some treatment wetlands, there is an iron deficiency, and calcium biogeochemistry is dominant. When this occurs, the wetland sediments contain a high proportion of calcium carbonate, which is referred to as calcitic mud or marl. The southern Everglades contain extensive areas of these calcitic muds, which form under conditions of shorter hydroperiod, as a result of calcium carbonate precipitation mediated by periphyton. These materials are very dense, low in organic content, and are typically low in phosphorus content. Calcareous periphyton in the south Florida environment contributes to high soil calcium, with concentrations ranging from 3–4% in peats to 20–40% in calcitic wetland sediments (Reddy *et al*., 1991; DeBusk *et al*., 2004).

Calcium is also important in constructed wetlands receiving some types of leachates. Municipal landfills may contain construction materials including gypsum wallboard (calcium sulfate), and the waste piles from phosphate fertilizer manufacture contain mostly calcium sulfate as well.

### Magnesium

Magnesium is an essential micronutrient because of its role in phosphate energy transfer and because it is a structural component in the chlorophyll molecule (Wetzel, 1983). Because magnesium concentration of surface water almost always exceeds the requirements for plant growth, elevated magnesium concentrations are not affected when wastewater travels through wetland treatment systems. Magnesium is more soluble than calcium, and precipitate formation does not occur (see Table 11.7). Plant tissue concentrations are approximately 0.25% dry weight, but may be higher for floating plants and filamentous algae (see Table 11.6).

## 11.3 COLLECTIVE PARAMETERS

### Hardness

Hardness measures the concentrations of divalent cations in a water sample. The prevalent divalent ions in most surface waters are calcium and magnesium. Rainwater typically has low hardness (soft water) with a calcium concentration between 0.1 and 10 mg/L, a magnesium concentration of about 0.1 mg/L, and a hardness value less than 30 mg/L as $CaCO_3$. Surface water hardness is variable, depending on the soil and rock concentrations of calcium and magnesium, and on the degree of contact with rocks, soils, and pollution. Inland surface water hardness varies from 10 to 300 mg/L as $CaCO_3$, with a calcium concentration between 0.3 and 70 mg/L and magnesium concentration between 0.4 and 40 mg/L.

### Total Ion Content

Two chemical parameters are commonly used to indicate the collective concentrations of dissolved substances: TDS and specific conductance. These parameters do not specify the distribution of contributing ions and organic compounds that contribute, but they are helpful in support of the wetland water budget. Further, the TDS content of water is sometimes a regulated parameter, especially in arid regions, where salt buildup is a water quality concern.

#### Total Dissolved Solids

TDS is used to quantify the degree of pollution in many industrial wastewater effluents, including textile wastes, food processing wastes, and pulp and paper wastes. When discharged to surface or groundwaters, these dissolved solids may represent a significant pollution source. The total quantity of dissolved solids in a water sample is measured by filtration followed by sample evaporation. This quantity contains both inorganic ions and organic compounds. TDS is nearly as conservative in wetlands as specific conductance and chloride. Because TDS concentrations are high in many wastewaters and the individual components of these solids greatly exceed the biological requirements for growth, wetlands generally have a negligible effect on this parameter (see Table 11.3).

#### Electrical Conductivity

EC, also called specific conductance, of an aqueous solution is the reciprocal of the resistance between two platinum electrodes, 1 cm apart and with a surface area of 1 $cm^2$. The reciprocal of EC is equal to resistance, and is a function of the total quantity of ionized materials in a water sample. Specific conductance usually is reported at a temperature of 25°C and in units of µS/cm, or µmhos/cm. Measurements can be made with pocket-portable, inexpensive meters. Specific conductance is nearly proportional to the TDS in many surface waters and is a convenient measure of the salt content of wastewaters.

Total ionic salts in wetlands, as measured by specific conductance, may be somewhat altered by biological conditions in wetlands, but physical processes of dilution and evaporation represent the major influences. Therefore, EC is a relatively accurate indicator of dilution and concentration effects by rainfall and runoff and evapotranspiration in wetland treatment systems.

Treatment wetlands are usually, but not always, dominated by the introduced flows. Rainfall and evapotranspiration are minor in comparison, except possibly for the duration of extreme events. Therefore, in the long run, EC of the inlet and outlet waters are close to the same. For the 17 pass-through systems of Table 11.3, the outlet EC averages

**TABLE 11.8**
**Pore Water Concentrations of Alkalinity, Calcium, and Electrical Conductivity in Cell 1 of the ENRP Wetland in Florida**

| Community | Location | EAV | SAV | FAV | Mean |
|---|---|---|---|---|---|
| Alkalinity (mg/L) | Surface water | 213 | 204 | 209 | 209 |
| | Pore water | 634 | 598 | 631 | 621 |
| Calcium (mg/L) | Surface water | 67 | 63 | 67 | 66 |
| | Pore water | 152 | 136 | 146 | 145 |
| Conductivity (μS/cm) | Surface water | 1,038 | 1,008 | 1,045 | 1,030 |
| | Pore water | 1,830 | 1,899 | 1,740 | 1,823 |

*Note:* EAV = emergent aquatic vegetation; SAV = submerged aquatic vegetation; FAV = floating aquatic vegetation.

*Source:* Data from DBE (2003) *Assessment of hydraulic and ecological factors influencing phosphorus removal in Stormwater Treatment Area 1 West.* Final report to Florida Department of Environmental Protection, Contract No. WM795, April 2003.

98% ± 2% of the inlet EC. This represents long-term mean performance, over an average of five years. Those wetlands receive an average annual hydraulic loading of 31 m/yr, which is far greater than precipitation or evapotranspiration, which is about 1 or 2 m/yr.

There are circumstances in which conductivity, chloride, and TDS change from inlet to outlet, and data then present a challenge for finding the cause. Three such anomalies are listed in Table 11.3. The first of these is data from the first 17 months of operation of the New Hanover County, North Carolina, system. This FWS wetland treats landfill leachate that has a high EC (>6,000 μS/cm), and during the start-up period produced an average outlet EC = 1,468 μS/cm. Chloride and TDS exhibit similar large decreases. The reason in this case is a very low hydraulic loading (0.79 m/yr), coupled with large rainfall. However, the larger effect was the long time required to replace the low conductivity water used to initially fill the wetland. Both factors produce large dilution of the incoming leachate.

The second illustration is for the Incline Village, Nevada, wetland. This arid-climate system does not receive water in the summer, because it is used to irrigate fodder crops. The design of the wetland was for disposal, primarily (≈90%) to evaporative losses, and secondarily (≈10%) to infiltration. The hydraulic loading is very small, and large concentration increases in chloride and TDS are produced as the water moves through the sequence of cells (Kadlec *et al.*, 1990).

The third illustration is the Ouray, Colorado, wetland, which exhibits twofold increases in TDS over a five-year averaging period (HDR/ERO, 2001). This is a strong signal of secondary sources of water entering the treatment wetland. This has been identified as a maintenance issue: "The wetland system experiences an outside flow problem with sulfates, which concentrate.…"

Conductivity has also been used as a diagnostic tool for internal processes in treatment wetlands in a number of ways. For example, EC is often much higher in the pore waters of FWS wetland soils and sediments than it is in overlying waters. DBE (2003) measured the EC of surface waters and pore waters in Cell 1 of the ENRP in 2001. The pore water EC was higher than that of the overlying surface waters (Table 11.8). Among the potential reasons is that rooted plants extract their transpiration requirement from pore water, but reject some or all of the associated salts. The result is an upward positive gradient in EC in the top soil layer. That gradient may continue into the overlying water, and produce stratification of the EC of the water column. The pattern of these results indicates an internal recycle loop, in which dissolved substances are drawn down into the root zone by transpiration or other flows, only to be rejected by the plants to avoid buildup of TDS within their tissues. This creates an upward gradient, which causes diffusive solute movements back into the water column from the pore water.

### *Density-Induced Vertical Stratification*

Conductivity is also a tool to understand the phenomenon of density stratification in constructed and other wetlands. Wetlands are typically too shallow to stratify due to thermal gradients, but the same is not true for density segregation, which may exist due to the character of incoming wastewater, rainfall, or added tracers. Two examples will serve to illustrate the potential for vertical stratification.

### *Salt Plumes in FWS Wetlands*

Various salts, such as sodium bromide and lithium chloride, are convenient tracers for water movement. Considerable quantities are needed, and consequently it is tempting to add concentrated solutions, in order to deal with manageable tracer solution volumes for addition. However, when dense solutions are introduced into the bottom of wetlands, there may be a strong energy barrier to vertical mixing, resulting in the dense material remaining on the bottom of the wetland. Concentrations used in a South Florida Water Management District (SFWMD, 2002) tracer study were 7.5% LiCl (density = 1.04 g/cm$^3$) (Söhnel and Novotny, 1985). Concentrations of sodium bromide in a Tres Rios, Arizona (Whitmer, 1998), tracer study were about 20% NaBr (density = 1.16 g/cm$^3$)

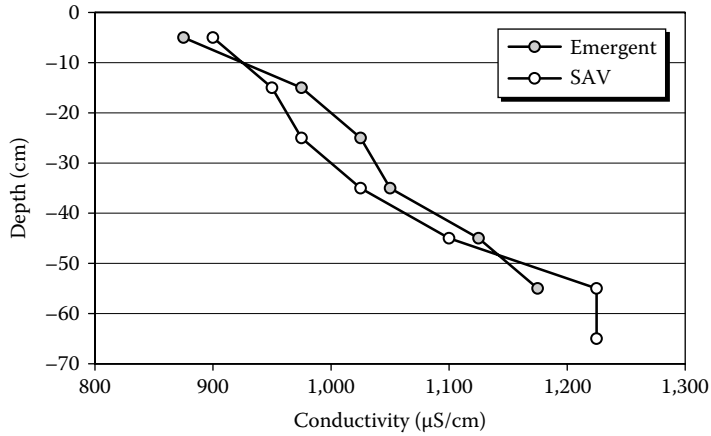



**FIGURE 11.2** Vertical stratification of electrical conductivity in the ENRP, Florida constructed FWS wetland. Points represent the averages for 141 dates during May 1995 to October 1997. (Data from Chimney *et al.* (2006) *Ecological Engineering* 27(4): 322–330.)

(Söhnel and Novotny, 1985). It is noteworthy that demonstration of stratification in limnology laboratory courses utilizes densities of 1.105 and 1.05 g/cm$^3$ (Wetzel and Likens, 1991). More directly, the work of Schmid *et al.* (2003, 2004b) shows that both tracer injections would lead to stable, unmixed layers of tracer on the wetland bottom. Both tracer studies showed very poor tracer recovery, suggesting that the tracer "got stuck" in the wetland sediments.

Further evidence of vertical stratification was reported by Chimney *et al.* (2006), for a constructed wetland in the Florida Everglades (Figure 11.2). In this case, a likely cause is the back-diffusion of salts from the concentrated pore water, as described in Table 11.8.

### *Stratification in HSSF Wetlands*

The presence of a gravel matrix in a HSSF wetland serves to exacerbate the potential for vertical stratification. It is well known that flows through clean porous media are susceptible to layering. For example, Wood *et al.* (2004) found that density effects occurred when the invading solution concentration was greater than approximately 13,000 mg/L. It is not surprising that Drizo *et al.* (2000) found that tracer bromide at 20,000 mg/L sank to the bottom of HSSF wetlands. Rash and Liehr (1999) have also reported stratification effects in HSSF wetlands.

The Grand Lake, Minnesota, wetland exhibited large vertical stratification during a start-up period of two years, as evidenced by much higher EC in the bottom water samples than in the top samples (Figure 11.3; Kadlec *et al.*, 2003). These wetlands were initially filled with water from an adjacent natural bog, which had lower EC than the incoming wastewater. The stratification persisted, as a result of the very low flow over the summer and fall of 1996. The stratification was mitigated in the fall of 1996 by pumping bottom water to the surface, although it restratified the next spring but not as strongly. After about two years of operation, only mild stratification existed, with 21% higher EC in the bottom water samples than in the top samples.

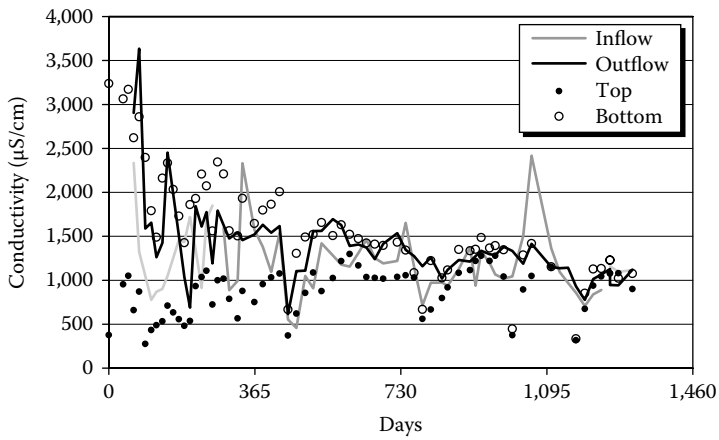



**FIGURE 11.3** Electrical conductivity for Grand Lake, Minnesota, HSSF wetland cell #1. Introduction and removal were at the cell bottom. High conductivity water persisted at 45 cm depth for about two years, and low conductivity water persisted at 15 cm depth. The influence of snow melt can be seen in the lower conductivity values on the top in each early spring. (Adapted from Kadlec *et al.* (2003) In *Constructed Wetlands for Wastewater Treatment in Cold Climates.* Mander and Jenssen (Eds.), WIT Press, Southampton, United Kingdom, pp. 19–52.)

## 11.4 SULFUR

Sources of sulfur include geochemical weathering of minerals, wind-blown sea salt, and emissions from fossil-fuel combustion (Wetzel, 1983). Large quantities of sulfur enter the atmosphere from natural and industrial sources, and return to earth as acid precipitation containing sulfates (sulfuric acid). Treatment wetlands receive these atmospheric inputs as well as sulfur compounds that may be included in the chemicals in the water to be treated. Municipal wastewater contains sulfur compounds, originating from the potable water supply, and augmented by waste products. Drinking water standards are strict for sulfide (2 μg/L), but less so for sulfate (250 mg/L). Hydrogen sulfide is a reactive and toxic gas with problematic side effects, including a rotten egg odor, corrosion, and acute toxicity.

The processing of sulfur in wetland ecosystems is represented by interconversions of several sulfur compounds in the different micro-regions of the ecosystem (Figure 11.4). Oxidized forms, such as sulfite, sulfate, and thiosulfate, are found in the oxygenated portion of the FWS water column. Reduced forms, including sulfide, bisulfide, and elemental sulfur, are found in the soils and sediments under conditions of low redox potential. Ionic and molecular forms are prevalent. Hydrogen sulfide and methylated sulfur compounds are volatile, and may be lost from the wetland to the atmosphere. Sulfate is an essential nutrient because its reduced, sulfhydryl (-SH) form is used in the formation of amino acids. Because there is usually enough sulfate in surface waters to meet the sulfur requirement, sulfate rarely limits overall productivity in wetland systems.

Although seldom a water quality target in its own right, sulfur is an important part of the chemical processing in wetlands. From a treatment perspective, sulfur has a critical role in the formation and storage of metal sulfides. In this section, the principal reactions of sulfur in the environment are explored, together with the treatment and storage potential.

Sulfur concentrations in wetland plant tissues typically range from 0.1–0.6% dry mass, but algal concentrations may be considerably larger (Table 11.9). Belowground tissues have not often been measured, but are considerably higher than aboveground plant part concentrations. Treatment wetland sediments contain sulfur at 0.1–1.0% dry mass.

### Dissimilatory Sulfate Reduction

Aerobic organisms excrete sulfur as sulfate. However, upon death and sedimentation, heterotrophic bacteria release the sulfur in the reduced state, which can result in the accumulation of high levels of hydrogen sulfide in wetland sediments. A second process that transforms sulfate and other oxidized sulfur forms (sulfite, thiosulfate, and elemental sulfur) to hydrogen sulfide in anaerobic sediments, dissimilatory sulfate reduction, is mediated by anaerobic, heterotrophic bacteria such as *Desulfovibrio* and *Desulfotomaculum*, which use sulfate as a hydrogen acceptor (Castro *et al.*, 2002;

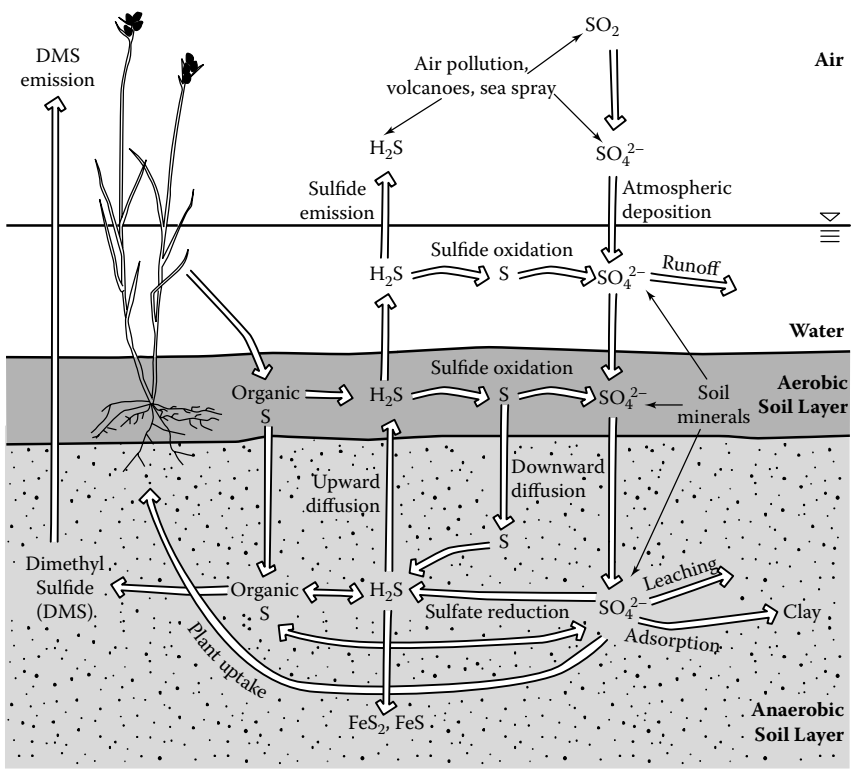



**FIGURE 11.4** Sulfur pathways and forms in FWS wetlands. (From Mitsch and Gosselink (1993), *Wetlands*. Second Edition, Van Nostrand Reinhold Company, New York. Reprinted with permission.)

**TABLE 11.9**
**Sulfur Content (mg/kg) of Example Wetland Plants and Sediments**

| Component | Water (mg/L) | Above (mg/kg) | Below (mg/kg) | Reference |
|---|---|---|---|---|
| **Plants** | | | | |
| *Phragmites australis* | 124 | 780 | 2,244 | Winterand Kickuth (1989b) |
| *Phragmites australis* | 113 | 3,880 | — | Samecka-Cymerman and Kempers (2001) |
| *Phalaris arundinacea* | 170 | 1,775 | — | Samecka-Cymerman and Kempers (2001) |
| *Typha latifolia* | 500 | 2,495 | 6,026 | Ye *et al.* (2001a, b) |
| *Typha latifolia* | — | 1,500 | — | Boyd (1978) |
| *Typha latifolia* | 17 | 9,800 | — | Kadlec and Alvord (1989) |
| *Typha angustifolia* | 67 | 3,800 | — | Samecka-Cymerman and Kempers (2001) |
| *Scirpus lacustris* | 67 | 3,430 | — | Samecka-Cymerman and Kempers (2001) |
| *Salix* spp. | 1 | 1,600 | — | Kadlec and Alvord (1989) |
| *Juncus effusus* | 500 | 3,853 | 6,885 | Ye *et al.* (2001a, b) |
| *Juncus effusus* | — | 2,600 | — | Boyd (1978) |
| *Scirpus americanus* | — | 5,900 | — | Boyd (1978) |
| *Hydrocotyle* spp. | — | 1,600 | — | Boyd (1978) |
| *Nuphar advena* | — | 3,200 | — | Boyd (1978) |
| *Ceratophyllum demersum* | — | 3,000 | — | Boyd (1978) |
| *Myriophyllum heterophyllum* | — | 2,400 | — | Boyd (1978) |
| *Potamogeton diversifolius* | — | 5,000 | — | Boyd (1978) |
| *Chara vulgaris* | — | 19,300 | — | Vymazal (1995) |
| *Cladophora glomerata* | — | 12,800 | — | Vymazal (1995) |
| **Sediments** | | | | |
| B Horizon | 124 | — | 343 | Winter and Kickuth (1989b) |
| Natural fen at water table | 16 | — | 440 | Bayley (1986) |
| A Horizon | 124 | — | 665 | Winter and Kickuth (1989b) |
| Natural cedar swamp | 6 | — | 5,766 | Spratt *et al.* (1987) |
| 10–15 cm | 500 | — | 7,300 | Ye *et al.* (2001a, b) |
| 5–10 cm | 500 | — | 7,900 | Ye *et al.* (2001a, b) |
| 0–5 cm | 500 | — | 10,300 | Ye *et al.* (2001a, b) |
| 0–10 cm discharge | 17 | — | 2,900 | Houghton Lake (unpublished) |
| 0–10 cm control | 1 | — | 3,200 | Houghton Lake (unpublished) |
| 30–50 cm (11 lakes, Poland) | 16–227 | — | 80–2,890 | Samecka-Cymerman and Kempers (2001) |

*Note:* Water concentration (mg/L) is for sulfate.

Lloyd et al., 2004). The presence of decaying organic matter in the wetland sediments and soils depletes oxygen and creates acid pore waters. Organic matter fuels sulfate reduction. For example:

$$SO_4^{2-} + 2CH_2O + 2H^+ \rightarrow H_2S + 2H_2O + 2CO_2 \quad (11.1)$$

$$SO_4^{2-} + 2CH_2O \rightarrow HS^- + H^+ + 2HCO_3^- \quad (11.2)$$

where $CH_2O$ represents the organic substrate required by the microbes.

Equation 11.1 is favored at low pH, while Equation 11.2 dominates at higher pH. As ferrous sulfide (FeS) is highly insoluble, hydrogen sulfide does not tend to accumulate until the reduced iron is removed from solution. When ion concentrations are low, or when sulfate and organic matter concentrations are high, significant hydrogen sulfide concentrations can occur. Several other metal sulfides are also very insoluble, including ZnS, CdS, and others.

Sulfate is mildly sorbable on soils. For example, Fumoto and Sverdrup (2001) found Freundlich isotherm parameters of $0.17 < K_F < 0.44$ [units: (mol/kg) × (mol/L)$^{-n}$] and $n = 0.078$ for mineral soils. For water at 20 mg/L, the resultant sorbed amounts were measured in the range 250–500 mg S/kg.

## Hydrogen Sulfide

Hydrogen sulfide exists in water solution as un-ionized ($H_2S$) or singly or doubly ionized (bisulfide, $HS^-$, or sulfide, $S^{2-}$), depending on water temperature and pH. The two dissociation reactions are:

$$H_2S \rightleftharpoons H^+ + HS^- \quad (11.3)$$

$$HS^- \rightleftharpoons H^+ + S^{2-} \quad (11.4)$$

The equilibria for these reactions in water are:

$$(\log_{10} K_1) = \left( \log_{10} \frac{C_{HS^-}}{C_{H_2S}} \right) - \text{pH} = \left( 2.6198 + \frac{1303}{T + 273.16} - \text{pH} \right) \quad (11.5)$$

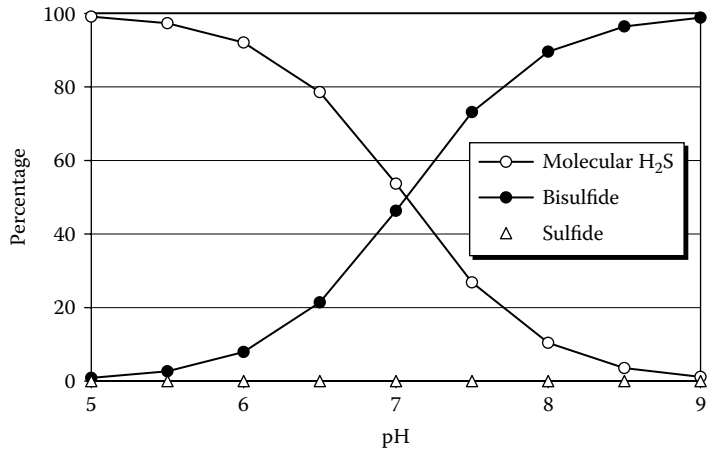



**FIGURE 11.5** Aqueous equilibrium concentrations of sulfide and bisulfide at 25°C. Based on dissociation constants from Lange's *Handbook of Chemistry* (1985).

$$(\log_{10} K_2) = \left( \log_{10} \frac{C_{S^{2-}}}{C_{HS^-}} \right) - \text{pH}$$
$$= \left( 4.2797 + \frac{2567}{T + 273.16} - \text{pH} \right) \qquad (11.6)$$

where

$C_{S^{2-}}$ = sulfide ion concentration, mol/L
$C_{HS^-}$ = bisulfide ion concentration, mol/L
$C_{H_2S}$ = un-ionized hydrogen sulfide concentration, mol/L
$K_1$ = first dissociation constant, dimensionless
$K_2$ = second dissociation constant, dimensionless
$T$ = water temperature, °C

At equilibrium, the un-ionized form is predominant at low pH, and bisulfide is dominant at high pH in aqueous systems (Figure 11.5). However, equilibrium is not necessarily attained in wetland systems, because of continual influxes of sulfate, and a large number of microbially mediated processes that may occur. In the presence of sulfate in aqueous solution, oxidation prevails at $E_h > -300$ mV at circumneutral pH (Pankow, 1991). In wetlands, the sulfate reduction zone occupies the range $-200 < E_h < -100$ mV (Reddy and D'Angelo, 1994). Nonetheless, there may be large fractions of un-ionized $H_2S$, which is volatile and may be lost to the atmosphere.

Volatilization of hydrogen sulfide requires mass transport to the air–water interface, followed by transfer into the air, and follows rules analogous to those discussed for ammonia volatilization (see Chapter 9). The Henry's law constant for $H_2S$ at 25°C is 3.49 mg/L·atm, which indicates large volatility. At other temperatures, the Henry's law constant, in units of mol fractions in the liquid and gas, is given by Lide (1992):

$$\ln_e H = -24.912 + \left( \frac{3477}{T + 273.16} \right) + 0.3993 \ln_e (T + 273.16)$$
$$+ 0.0157(T + 273.16) \qquad (11.7)$$

The volatility of $H_2S$ means that losses in the wetland environment are controlled by mass transport processes in the water phase.

## Hydrogen Sulfide in Municipal Wastewater Treatment Wetlands

The Listowel, Ontario, system studied five wetlands for four years, including the $H_2S$ content of the incoming and outgoing waters (Table 11.10) (Herskowitz, 1986). The sulfate content was only infrequently monitored, but was typically in the range of 170–200 mg/L. Alum additions to wetlands 1, 2, and 3 accounted for 24 mg/L of the incoming sulfate. There were marked differences between wetlands of high aspect ratio (Systems 1, 3, and 4), which had lower outlet $H_2S$, and those of low aspect ratio (Systems 2 and 5), which had much higher outlet $H_2S$. All systems produced hydrogen sulfide in the warm months.

As a point of reference, the rate of emission of $H_2S$ was measured in anaerobic ponds in the Mediterranean climate at Meze, France (Paing *et al.*, 2003). The pond reduced sulfate

**TABLE 11.10**
**Hydrogen Sulfide in the Listowel, Ontario, Constructed Wetlands, 1980–1984, in mg/L**

| Total $H_2S$ | System | 1 | 2 | 3 | 4 | 5 |
|---|---|---|---|---|---|---|
| Annual | In | 1.81 | 1.81 | 1.81 | 0.23 | 0.23 |
| | % Un-ionized | 72% | 72% | 72% | 57% | 57% |
| | Out | 1.19 | 4.44 | 1.00 | 1.28 | 3.10 |
| | % Un-ionized | 76% | 67% | 65% | 63% | 69% |
| Jan-Feb-Mar | In | 3.93 | 3.93 | 3.93 | 0.46 | 0.46 |
| | Out | 0.75 | 3.41 | 0.72 | 0.66 | 3.22 |

*Note:* The period of ice cover was typically November through March.

*Source:* Herskowitz (1986) *Listowel Artificial Marsh Project Report*. Ontario Ministry of the Environment, Water Resources Branch, Toronto, Ontario.

from 165 mg/L to 57 mg S/L, but produced sulfides, with an increase from 3.8 to 19.2 mg S/L. The $H_2S$ emission rates were found in the range of 20–576 mg S/m$^2$·d (mean = 172). This led to atmospheric concentrations as high as 5 ppm, which is well above the human odor threshold of about 0.05 ppm. Almasi and Pescod (1996) also found high sulfides in ponds for warm (25°C) and cool (10°C) conditions, in the range 15–60 mg/L, leading to $H_2S$ concentrations of 2–12 mg S/L.

### Oxidation of Sulfur and Sulfides

When it is exposed to air or oxygenated water, hydrogen sulfide may be oxidized back to sulfate. This may occur via sulfur bacteria such as *Beggiatoa*, which promotes the oxidation of hydrogen sulfide to elemental sulfur:

$$H_2S + \frac{1}{2} O_2 \rightarrow S^\circ + H_2O \qquad (11.8)$$

*Thiobacillus* can subsequently cause oxidation of elemental sulfur to sulfate:

$$S^\circ + \frac{3}{2} O_2 + H_2O \rightarrow SO_4^{2-} + 2H^+ \qquad (11.9)$$

Photosynthetic bacteria, such as purple sulfur bacteria, use hydrogen sulfide as an oxygen acceptor in the reduction of carbon dioxide, resulting in partial or complete oxidation back to sulfate:

$$H_2S + CO_2 \rightarrow S^\circ + CH_2O \qquad (11.10)$$

where $CH_2O$ represents organic matter.

Under some circumstances, treatment wetlands have been observed to turn purple, as happened in a FWS treating high-strength potato processing wastewater (P. Burgoon, personal communication).

In any case, these microbially mediated reactions suggest that elemental sulfur may be found in treatment wetlands. Anecdotal reports of elemental sulfur have been made for the Houghton Lake, Michigan, system; the Tres Rios, Arizona, wetlands; the Brighton, Ontario, system; and the Nučice, Czech Republic, HSSF system. In extreme situations, a whitish colloid or adhering whitish precipitate is seen in the outflow from the treatment wetland (Figure 11.6).

Gammons *et al.* (2000a) comment about this phenomenon in connection with the Butte, Montana, mine drainage treatment wetlands:

> Large quantities of excess $H_2S$ produced in the anaerobic cells created a foul smell and also resulted in unsightly precipitates of colloidal sulfur in downstream aerobic waters. It is evident from the above observations that optimal wetlands performance is in some respects a delicate balancing act. Too much BSR [biological sulfate reduction] activity results in an undesirable accumulation and release of $H_2S$, whereas too little results in decreased metal attenuation.



**FIGURE 11.6 (A color version of this figure follows page 550)** This HSSF wetland outlet structure at Tamarack, Minnesota, has become coated with elemental sulfur.

Winter and Kickuth (1989b) reported about 36% of the removed sulfur in a HSSF wetland treating textile industry wastewater was stored in the form of elemental sulfur.

### Organic Sulfur

Organic sulfur compounds account for a good share of the sulfur found in wetland sediments. For instance, 84–88% of the total sulfur in a New Jersey peat was organic sulfur (Spratt *et al.*, 1987), and over 90% in a West Virginia peat (Wieder and Lang, 1988). In treatment wetlands, the storage of sulfur is also in major part associated with humic materials. For instance, Winter and Kickuth (1989b) reported about 30% of the removed sulfur was in humic materials.

Additionally, there are several low molecular weight organic sulfur compounds that may be found in wastewater. Methanethiol ($CH_3SH$) and dimethyl sulfide (DMS or $(CH_3)_2S$) are perhaps the most common, and are quite volatile (Faulkner and Richardson, 1989; Lomans *et al.*, 2002). Both are extremely odiferous. There are several mechanisms that can produce and destroy these volatile organic sulfur compounds (see Lomans *et al.*, 2002). Kiene and Hines (1995) found both were formed in natural fen peat at the same rate of 40 nmol/L·d (256 μg/m$^2$·d in the top 20 cm of soil). Wood *et al.* (2000) measured DMS removals of 80% (152–28 mg/L) in SSF wetlands treating swine wastewater, and attributed the loss to mineralization and oxidation. Domestic wastewater contains lesser amounts of DMS, and reductions are not significant for anoxic HSSF systems. Huang *et al.* (2005b) found small removals, averaging 20% (2.24 μg/L down to 1.79 μg/L). Their studies involved eight wetlands, with average redox of –35 mV, and reductions of sulfate of 60% (72.5 mg/L down to 29.4 mg/L).

### Phytotoxicity

Lamers (1998) documents that sulfate has negative effects on the growth rate of *Carex nigra*, *Juncus acutiflorus,* and *Gallium*

*palustre*, at concentrations of 64 and 128 mg S/L. Koch and Mendelssohn (1989) report that 32 mg S/L of sulfide produced negative effects in *Panicum hemitomon* and *Spartina alterniflora*. The presence of sulfide is coupled with anaerobic conditions in the root zone, but the effects of sulfide go beyond mere anoxia (Koch *et al.*, 1990). Hydrogen sulfide apparently inhibits the activity of alcohol dehydrogenase, thereby limiting the ability of plants to avail themselves of alternative anoxic energy pathways. This effect was confirmed by measuring a reduced $^{15}N$ uptake rate in the presence of sulfide. However, the availability of free sulfide is strongly mediated by the presence of iron, because of the formation of iron sulfides.

Phytotoxicity was found to be very serious at the 45-mg S/L level in *Phragmites australis* (Armstrong *et al.*, 1996). These authors found that aeration pathways became blocked, interfering with the diffusive connection to the atmosphere, and thus reducing the plant's ability to oxygenate the rhizosphere. Smolders and Roelofs (1996) found for *Stratiotes aloides,* an aquatic macrophyte characteristic for mesotrophic freshwater marshes, that levels of 320 mg S/L were toxic to the roots). Lamers *et al.* (2002) found root parts, growing in 1.7–3.4 mg S/L of sulfate into the peaty sediment, clearly showed sulfide toxicity by becoming black, slimy, and unfit for nutrient uptake from the sediment. Free sulfide could not be detected in the surface water. They concluded that only roots in the surface water would survive. *Nuphar lutea* did not propagate in the sulfate-treated enclosures. However, the sensitivity of a wetland plant species to free sulfide not only depends on the actual sulfide levels in the rhizosphere, but also on detoxification mechanisms like radial oxygen loss. As noted above, high sulfide concentrations in freshwater sediments may also lead to higher fluxes of volatile organic sulfur compounds to the atmosphere due to the microbial methylation of hydrogen sulfide.

## Performance of Wetlands for Sulfur Removal

Since sulfate inputs in surface flow wetland treatment systems frequently exceed the biological requirements of wetland biota, wetlands generally are not as effective for removal of sulfur as for other contaminants (Wieder, 1989). Although microbial routes provide for gaseous losses of $H_2S$ and DMS, these require the very low redox potentials usually found only in deeper wetland sediments. Metal sulfide precipitation often blocks much of the gaseous loss by immobilizing sulfides in the sediments. Plant storage is minimal, for instance plant uptake was estimated at 1.5 g S/m$^2$·yr in a natural bog (Hemond, 1980). Winter and Kickuth (1989a,b) reported only 1% taken up by plants in a HSSF treatment wetland. Consequently, the majority of sulfur removal will generally be to organic, elemental, and metal sulfide forms in the wetland sediments.

As a result, the median-observed concentration reduction is only 14% for 32 wetlands (Table 11.11). Only a few mine water wetlands show more than 50% reduction, and that may be attributed to the anaerobic mode of operation in some cases. Subsurface horizontal flow wetlands also sometimes satisfy this anoxic condition. Winter and Kickuth (1989a, b) reported that a root-zone, soil-based treatment system receiving textile wastewaters from a facility in Bielefeld, West Germany, removed from 80–85% of the sulfur mass at an hydraulic loading rate of 1.14 cm/d for a removal rate of 9.6 kg/ha·d. These authors reported that the majority of this sulfur was largely stored in the wetland soil as elemental sulfur (31%) and organic sulfur (25%), and that only a small fraction was released by volatilization to the atmosphere. Huang *et al.* (2005b) observed 24–88% reduction for hydraulic loadings of 2.0–4.5 cm/d, corresponding to sulfate–sulfur loadings of 5–10 kg/ha·d. In Table 11.11, results from five HSSF constructed wetlands in the Czech Republic are presented. The median removal is 51%, indicating anaerobic conditions in the beds with horizontal sub-surface flow. However, Vymazal and Kröpfelová (2006) pointed out that despite removal of sulfates, elimination of ammonia may occur at the same time. The question remains to be answered whether this removal is due to "conventional" aerobic nitrification, which may proceed in aerobic microzones adjacent to plant roots, or due to Anammox, i.e., anaerobic ammonia oxidation.

Sulfate removal should not be viewed as a process that is independent of other wetland chemistry and processes. For instance, studies by Wiessner *et al.* (2005a) determined that sulfate reduction was strongly dependent on wastewater biochemical oxygen demand (BOD), presumably acting as a carbon source, in a manner analogous to denitrification. BOD of 200 mg/L led to 100% removal of sulfate. Conversely, ammonia reduction decreased from 75 to 25% as sulfate reduction increased. Wiessner *et al.* (2005b) conclude that the importance of the sulfur transformation processes inside the rhizosphere of constructed wetlands, even in the case of treatment of domestic wastewater, has been underestimated. Extreme variations of removal processes in large-scale treatment wetlands may reflect this fact. The sensitivity of nitrification, for example, could be due to nutrient or oxygen limitations, but could also have been additionally or exclusively caused by products of sulfur transformation. Wiessner *et al.* (2005b) suggest that, in view of the interesting high application potential of simple wetland systems for the removal of metals by sulfide precipitation or the treatment of sulfate-rich wastewaters such as acid mine drainage, the dynamics of the sulfur cycle in the rhizosphere should be understood in more detail.

## Sulfur-Induced Eutrophication

Interactions also may exist between sulfur processing and phosphorus removal in treatment wetlands. In some systems, but not all, a fraction of the accreted phosphorus is bound by iron-containing substances. Other fractions are bound in calcium-rich materials, or in organic components of soils and sediments. Two effects have been reported: (1) a reduction of phosphorus uptake due to sulfide toxicity, and (2) sulfide binds iron and interferes with that component of phosphorus storage that relies upon the iron–phosphorus link (Lamers *et al.*, 1998).

**TABLE 11.11**
**Example Performances of Treatment Wetlands for Sulfate Reduction**

| System | Wetland Cells | Years | Inlet (mg/L) | Outlet (mg/L) | Reduction (%) | Reference |
|---|---|---|---|---|---|---|
| **Mine Water** | | | | | | |
| Woodcutters, Australia | All | 1 | 370 | 160 | 57 | Noller *et al.* (1994) |
| Butte, Montana | WDP1 | 3 | 399 | 367 | 8 | Gammons *et al.* (2000) |
| Widows Creek, Alabama | All | 1 | 539 | 210 | 61 | Ye *et al.* (2001a, b) |
| Albright, West Virginia | All | — | 870 | 723 | 17 | Hoover *et al.* (1998) |
| Northeastern United States | 109 Wetlands | 1 | 1,194 | 997 | 16 | Wieder (1989) |
| Springdale, Pennsylvania | All | — | 1,334 | 1,288 | 3 | Hoover *et al.* (1998) |
| Lick Run, Ohio | All | 1 | 1,672 | 1,286 | 23 | Mitsch and Wise (1998) |
| Idaho Springs, Colorado | Cell A | 1 | 1,700 | 1,680 | 1 | Machemer *et al.* (1993) |
| Greenville, Kentucky | 5 Wetlands | 1 | 3,132 | 2,786 | 11 | Wieder (1992) |
| Northumberland, United Kingdom | Shilbottle | 1 | 8,000 | 2,000 | 75 | Batty and Younger (2004) |
| West Virginia | All | 1 | 1,330 | 1,125 | 15 | Girts *et al.* (1987) |
| Whittle, United Kingdom | All | 1 | 3,110 | 2,564 | 15 | Batty *et al.* (2005) |
| Quaking Houses, United Kingdom | All | 1 | 811 | 446 | 45 | Batty *et al.* (2005) |
| Jones Branch, Kentucky | All | 1 | 3,034 | 1,352 | 55 | Barton and Karathanasis (1999) |
| **Leachate** | | | | | | |
| Monroe County, New York | All | 1 | 23 | 20 | 13 | Eckhardt *et al.* (1997) |
| Isanti-Chisago, Minnesota | All | 3 | 80 | 74 | 8 | Kadlec (2003c) |
| Lambton, Ontario | All | 1 | 1,845 | 1,533 | 17 | Unpublished data |
| **Agricultural Runoff** | | | | | | |
| STA5, Florida | All | 1 | 10 | 5 | 50 | Unpublished data |
| STA6, Florida | All | 1 | 27 | 24 | 9 | Unpublished data |
| ENRP, Florida | All | 6 | 70 | 54 | 22 | Unpublished data |
| STA1W, Florida | All | 1 | 90 | 88 | 2 | Unpublished data |
| STA2, Florida | All | 1 | 103 | 77 | 24 | Unpublished data |
| Brawley, California | All | 4 | 675 | 731 | –8 | Unpublished data |
| Imperial, California | All | 4 | 722 | 676 | 6 | Unpublished data |
| **Industrial** | | | | | | |
| Santo Tomé, Argentina | All | 1 | 1,444 | 929 | 36 | Hadad *et al.* (2006) |
| Santo Tomé, Argentina | All | 3 | 1,222 | 647 | 36 | Maine *et al.* (2006) |
| **Municipal Wastewater** | | | | | | |
| OEW, Florida | WP1-MM7 | 10 | 54 | 47 | 13 | Unpublished data |
| Lakeland, Florida | All | 3 | 138 | 111 | 20 | NADB database (1998) |
| Tres Rios, Arizona | Cobble 1 | 8 | 172 | 181 | –5 | Unpublished data |
| Tres Rios, Arizona | Cobble 2 | 8 | 173 | 175 | –1 | Unpublished data |
| Tres Rios, Arizona | Hayfield 1 | 8 | 177 | 181 | –2 | Unpublished data |
| Tres Rios, Arizona | Hayfield 2 | 8 | 181 | 185 | –2 | Unpublished data |
| Estevan, Saskatchewan | All | 10 | 636 | 633 | 0 | Unpublished data |
| **Median** | | | | | **14** | |
| **Municipal HSSF** | | | | | | |
| Břehov, Czech Republic | All | 2 | 59 | 29 | 51 | Vymazal and Kröpfelová (2006) |
| Ondřejov, Czech Republic | All | 1 | 86 | 25 | 71 | Vymazal and Kröpfelová (2006) |
| Trhové Dušníky, Czech Republic | All | 1 | 80 | 60 | 25 | Vymazal and Kröpfelová (2006) |
| Cistá, Czech Republic | All | 1 | 72 | 68 | 6 | Vymazal and Kröpfelová (2006) |
| Mořina, Czech Republic | All | 1 | 154 | 73 | 53 | Vymazal and Kröpfelová (2006) |
| **Median HSSF** | | | | | **51** | |

The response of wetlands to new sources of sulfate, and ultimately sulfide in the sediments, therefore, differs considerably depending upon the sources and quantities of iron in the wetland (Lamers *et al.*, 2002). For example, a *Typha-Carex* wetland at Tienhoven near Utrecht, Netherlands, had relatively high iron, and a continuing supply from groundwater. Addition of sulfate caused oxidation of iron in response to sulfate reduction, and considerable quantities of phosphorus were released. However, a second wetland, dominated by *Nuphar* surrounded by *Phragmites*, in Weerribben National Park, Netherlands, did not have a groundwater supply of iron. Sulfate additions to that wetland caused large quantities of sulfide formation, but no phosphorus was released.

## 11.5 TRACE METALS: GENERAL CONSIDERATIONS

### Toxic Effects in Water and Sediments

A number of trace metals are essential micronutrients at low concentrations, but some trace metals may occur in municipal wastewaters at concentrations that are toxic to sensitive organisms. The probability of exceeding sensitive toxicity levels in mine water, leachates and some industrial waters is much higher. For most trace elements, biochemical transformations and chemical characteristics can lead to *biomagnification*, a phenomenon in which increasing concentrations occur in consumers along a food chain. Although most trace metals are more concentrated in biological tissues and soils than they are in surface water, hazardous situations do not always occur.

Metals in wastewater must be removed prior to final discharge to protect the environment from toxic effects, but the use of wetlands to accomplish this goal must be examined cautiously. Surface flow wetland treatment systems are open to biota that may be exposed to potentially dangerous levels of metals, primarily in the wetland sediments. The removal of metals can result in storage in sediments that is inimical to the subset of wetland organisms that live or feed in those sediments. To prevent this problem from occurring, wetland treatment system designers and regulators should consider pretreatment to reduce influent metal concentrations. Deep water systems with floating plants send sediments to depths that are out of reach of top feeders. A second alternative is to minimize the opportunity for ingestion of metals. Subsurface flow wetlands accomplish this purpose.

In the United States, most treatment wetlands are not considered waters of the United States, and would therefore not be required to meet water quality guidelines for the waters they are designed to protect. The levels of metals that may be tolerated by sensitive organisms have been promulgated in the form of guidelines for the protection of receiving waters and associated sediments. Examples of such guidelines are presented in Tables 11.12 and 11.13. These may or may not be considered applicable to treatment wetlands, which are not necessarily themselves protected by U.S. water quality guidelines.

### Abiotic Metal Partitioning

Depositing sediments are capable of adsorbing significant quantities of trace metals directly or indirectly through the accumulation of coatings such as organic matter, iron, and manganese oxyhydroxides, which will in turn act as trace element collectors. Organic matter, which may exist as a surface coating or as a particulate, may play an important role in metal speciation and bioavailability. Microbial decomposition of organic matter typically results in sediments that are anoxic under a thin oxic surface layer. Under these anoxic conditions, divalent cationic metals such as cadmium, copper, lead, nickel, and zinc are readily form metal sulfides. As long as amorphous sulfide concentrations are in excess of the total trace-metal concentration (on a molar basis), these metals will occur predominantly as insoluble metal sulfides. In anoxic sediments, metals in excess of sulfide may complex with organic matter. This buffers organisms against metal toxicity (Doig and Liber, 2006).

An appreciation of transformation rates of divalent cationic partitioning is needed to predict behavior and risk within natural environments. Although divalent cationic metals are not redox-active species within soil or aquatic environments,

**TABLE 11.12**
**Guidelines for Metal Concentrations in Water**

| Metal | Symbol | Hardness = 100 mg/L | U.S. EPA Freshwater CCC (µg/L) | U.S. EPA Freshwater CMC (µg/L) | NOAA SQRT (µg/L) | U.S. EPA Human Health (µg/L) |
|---|---|---|---|---|---|---|
| Silver | Ag | * | 3.2 | — | 0.1 | — |
| Aluminum | Al | — | 750 | 87 | 87 | — |
| Arsenic | As | — | 340 | 150 | 150 | 0.018 |
| Cadmium | Cd | * | 2 | 0.25 | 1.1 | — |
| Chromium | Cr(III) | * | 570 | 74 | 210 | — |
| Chromium | Cr(IV) | — | 16 | 11 | — | — |
| Copper | Cu | * | 13 | 9 | 12 | 1,300 |
| Iron | Fe | — | — | 1,000 | 1,000 | 300 |
| Manganese | Mn | — | — | — | — | 50 |
| Mercury | Hg | — | 1.4 | 0.77 | 0.012 | — |
| Nickel | Ni | * | 470 | 52 | 160 | 610 |
| Lead | Pb | * | 65 | 2.5 | 3.2 | — |
| Selenium | Se | — | — | 5 | 5 | 170 |
| Zinc | Zn | * | 120 | 120 | 110 | 7,400 |

*Note:* As noted, some are hardness-dependent. The criteria maximum concentration (CMC) is an estimate of the highest concentration of a material in surface water to which an aquatic community can be exposed briefly without resulting in an unacceptable effect. The criterion continuous concentration (CCC) is an estimate of the highest concentration of a material in surface water to which an aquatic community can be exposed indefinitely without resulting in an unacceptable effect. U.S. EPA numbers are provided in U.S. EPA (2002a, 2006). Screening quick reference tables (SQRT) are provided by National Oceanic and Atmospheric Administration (1999).

**TABLE 11.13**
**Guidelines for Metal Concentrations in Sediments**

| | | Ontario | | NOAA | | Wisconsin | |
|---|---|---|---|---|---|---|---|
| Metal | Symbol | Background Level (mg/L) | Lowest Effect Level (µg/g) | Severe Effect Level (µg/g) | SQRT (µg/g) | TEC (µg/g) | PEC (µg/g) |
| Silver | Ag | — | — | — | — | 1.6 | 2.2 |
| Arsenic | As | 4.2 | 6 | 33 | 5.9 | 9.8 | 33 |
| Cadmium | Cd | 1.1 | 0.6 | 10 | 0.6 | 0.99 | 5 |
| Chromium | Cr | 31 | 26 | 110 | 37.3 | 43 | 110 |
| Copper | Cu | 25 | 16 | 110 | 35.7 | 32 | 150 |
| Iron | Fe | 31,200 | 20,000 | 40,000 | — | 20,000 | 40,000 |
| Manganese | Mn | 400 | 1,100 | — | — | 460 | 1,100 |
| Mercury | Hg | 0.1 | 0.2 | 2 | 0.17 | 0.18 | 1.1 |
| Nickel | Ni | 31 | 16 | 75 | 18 | 23 | 49 |
| Lead | Pb | 23 | — | — | — | 36 | 130 |
| Zinc | Zn | 65 | 120 | 820 | 123 | 120 | 460 |

*Note:* Wisconsin (WDNR, 2003) levels are a threshold effect concentration (TEC) and a probable effect concentration (PEC). Ontario guidelines from OMOEE (1994). Screening quick reference tables (SQRT) are provided by National Oceanic and Atmospheric Administration (1999).

oxidation and reduction reactions may nevertheless affect partitioning. Retention can be modified by changes in substrate chemistry. For instance, zinc sorbs strongly to iron and manganese oxyhydroxides in aerated systems, and reacts with hydrogen sulfide to yield zinc sulfide in anaerobic environments. Thus, changes in redox status may shift zinc partitioning. For example, reductive dissolution of iron and manganese oxyhydroxides under anaerobic conditions releases zinc into the aqueous phase; persistence of anoxic conditions may then lead to a repartitioning of zinc into sulfide or carbonate precipitates. However, slow transformation rates and fluctuation in conditions may alter these predicted phase changes. Redox is affected by wetland water depth, and depth is therefore a partial surrogate for redox. At a contaminated wetland site in Idaho, these speciation effects were observed (Table 11.14) (Bostick *et al.*, 2001). There are associated consequences for metal removal. For instance, for zinc, an increase in water depth from 0.3 to 1.0 m produced a decrease in removal from 38–18% for a fixed detention time of one day (Gillespie *et al.*, 2000).

**TABLE 11.14**
**Speciation of Sedimentary Zinc as a Function of Water Depth in a Metal-Contaminated Wetland**

| | | Hydroxide-Sorbed | | |
|---|---|---|---|---|
| Water Depth (cm) | ZnO (%) | Zn (%) | ZnS (%) | $ZnCO_3$ (%) |
| 0 | 0 | 97 | 3 | 0 |
| 36 | 0 | 38 | 62 | 0 |
| 39 | 43 | 0 | 57 | 0 |
| 89 | 0 | 0 | 85 | 15 |
| 105 | 0 | 0 | 0 | 88 |

*Note:* Values are for the top 5 cm of sediment cores.

*Source:* Adapted from Bostick *et al.* (2001) *Environmental Science and Technology* 35: 3823–3829.

## Sorption Relations

The Langmuir isotherm fits metal sorption data for a number of wetland substrates:

$$C_S = \frac{KC_L}{1 + aC_L} \tag{11.11}$$

where

$a$ = Langmuir parameter, L/mg
$C_L$ = metal concentration in the water, mg/L
$C_S$ = metal concentration in the soil, mg/g
$K$ = Langmuir parameter, L/g

The maximum capacity for metal is $C_{Smax} = K/a$, which is only achieved at high water concentrations. The half-saturation water concentration, where $C_S = 0.5C_{Smax}$, is equal to $1/a$. At low water concentrations, $C_L << 1/a$, the Langmuir relation reduces to a linear partition equation:

$$C_S = KC_L \tag{11.12}$$

Freundlich isotherms are also often used to fit partitioning data:

$$C_S = K_F C_L^n \tag{11.13}$$

where

$n$ = Freundlich parameter, dimensionless
$K_F$ = Freundlich parameter, $(mg/kg) \times (mg/L)^{-n}$

Lesage *et al.* (2006) reported that removal of Co, Ni, Cu, and Zn by gravel and straw could be well described by the Langmuir isotherm with $R^2 \geq 0.97$ for gravel and $R^2 \geq 0.93$ for straw. El-Gendy (2006) reported that sorption of heavy

metals from the landfill leachate by roots of water hyacinth (*Eichhornia crassipes*) could be described by both Langmuir and Freundlich isoterms with respective correlation coefficients $R^2$ of 0.94 and 0.93. Organic sediments contain polar functional groups such as acids and phenolics that are responsible for cation exchange capacity (CEC), and can be involved in chemical binding (Ho and McKay, 2000). The metal-sediment reaction may be represented as:

$$2HP + M^{2+} \rightleftharpoons MP_2 + 2H^+ \qquad (11.14)$$

where $P$ represents the organic material (peat).

If Equation 11.14, cation exchange, is presumed to represent data, then the amount of metal bound is represented by (Kadlec and Keoleian, 1986):

$$K_{eq} = \frac{[MP_2][H^+]^2}{[HP]^2[M^{2+}]} \qquad (11.15)$$

where the bracket notation denotes molar concentration. This suggests that the partition coefficient ($[MP_2]/[M^{2+}] = C_S/C_L = K$) should go down markedly with decreasing pH, which is in fact the observation of many investigators, as shown in Kadlec and Keoleian (1986).

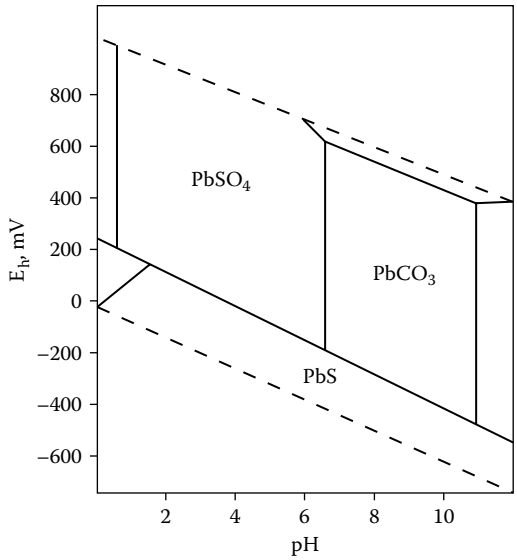



**FIGURE 11.7** Solid phases of lead in the presence of sulfate, sulfide, and carbonate. (From DeVolder *et al*. (2003) *Journal of Environmental Quality,* 32(3): 851–864. Reprinted with permission.)

## Equilibrium Metal Chemistry Calculations

There are several computer codes that have been developed to compute the theoretical thermodynamic equilibria in water solutions. For instance, MINTEQA2 is an equilibrium speciation model that can be used to calculate the equilibrium composition of dilute aqueous solutions in the laboratory or in natural aqueous systems (U.S. EPA, 1991). The model calculates the equilibrium mass distribution among dissolved species, adsorbed species, and multiple solid phases under a variety of conditions. A comprehensive database is included that is adequate for solving a broad range of problems without need for additional user-supplied equilibrium constants. The model employs a predefined set of components that includes free ions and neutral and charged complexes. The database of reactions is written in terms of these components as reactants. An ancillary program, PRODEFA2, serves as an interactive preprocessor to help produce the required MINTEQA2 input files. Code to achieve similar results, PHREEQC, is available from the U.S. Geological Survey (Parkhurst *et al*., 1980). These equilibrium calculations may be used to prepare solubility diagrams for metals in the presence of a variety of anions (see, e.g., Figure 11.7).

There are, however, substantive discrepancies that occur between predictions and observed wetland water chemistry. Solubility calculations for a mesocosm of homogenized sediment indicated supersaturation with respect to the sulfides of iron, copper, nickel, and zinc, yet measurements demonstrated a substantial supply of both trace metals and sulfide from the solid phase to the pore waters (Naylor *et al*., 2005). Ratios of metals measured in pore waters were consistent with their release from iron and manganese oxides, indicating that supply, as much as removal processes, determines the pseudo-steady state concentrations in the pore waters. The Naylor *et al*. (2005) observations suggest that trace metals are not immediately bound in an insoluble, inert form when they are in contact with sulfide. As a result of the complex wetland situation, forecasts from computer programs such as MINTEQA2 are not accurate representations of wetland situations. For example, Frandsen and Gammons (2000) found predictions of zinc remaining in solution to be underestimated by several factors of ten at low water concentrations, and to be strongly dependent upon the assumed form of the solid sulfide (Figure 11.8).

The reasons for such large discrepancies are presumably related to the complexities of the treatment wetland environment, compared to abiotic laboratory experiments with controlled water chemistry. The calculations are for thermodynamic equilibrium conditions, which may not be satisfied for the dynamic conditions of flow through wetlands. Unfortunately, it would appear that equilibrium calculations are of little value in predicting treatment wetland performance for metal removal.

## Design Equations for Metal Removal

The literature does not currently contain a clear indication of what first-level design calculation should be used for treatment wetland sizing for metal removal. The two simplest choices are a fixed load removal or a first-order areal calculation. The former is advocated in the mine water treatment literature (Hedin *et al*., 1994; Younger, 2000; Younger *et al*., 2002), and is termed the area-adjusted contaminant removal rate. This is essentially a zero-order model, which fixes the removal rate per unit area of the wetland:

$$R_A = \frac{Q}{A}(C_i - C_o) \qquad (11.16)$$

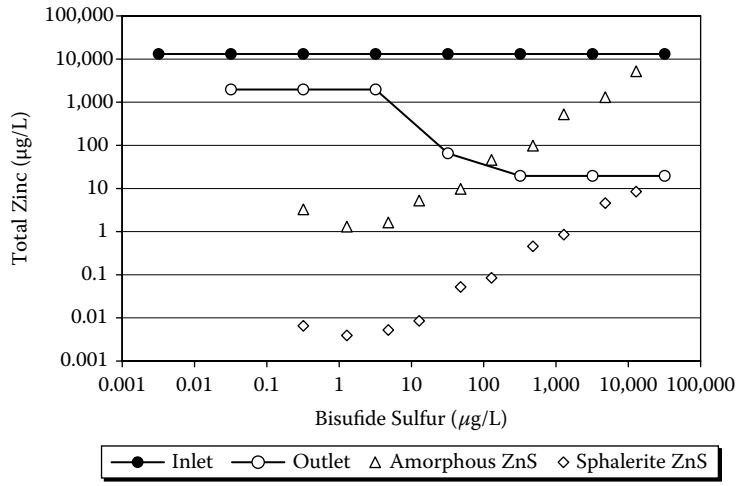



**FIGURE 11.8** The reduction in zinc in an anaerobic treatment wetland in Butte, Montana, compared to MINTEQA2 forecasts for amorphous and mineral ZnS. (Adapted from Frandsen and Gammons (2000) In *Wetlands & Remediation: An International Conference*. Battelle Press, Columbus, Ohio, pp. 423–430.)

where

$A$ = wetland area, $m^2$
$C_i$ = inlet concentration, mg/L
$C_o$ = outlet concentration, mg/L
$Q$ = flow rate, $m^3/d$
$R_A$ = area-adjusted contaminant removal rate, $g/m^2 \cdot d$

Tarutis *et al.* (1999) determined $R_A$ values for 35 wetlands for iron and manganese, and found median and mean values of 3.5 and 6.5 $g/m^2 \cdot d$ for iron and 0.24 and 0.73 $g/m^2 \cdot d$ for manganese, respectively. Younger *et al.* (2002) found $R_A$ median and mean values of 10 and 11.4 $g/m^2 \cdot d$ for iron (N = 20 wetlands) and 0.10 and 0.28 $g/m^2 \cdot d$ for manganese (N = 17 wetlands), respectively. However, in both studies, the coefficients of variation were unacceptably large, in the range of 100–200%. This zero-order uptake model would correspond to a fixed rate of supply of a precipitating reactant, such as sulfide. The subsequent sections of this chapter show that metal removal is not constant for most wetlands, but strongly correlated to the metal loading to the wetland.

The second choice, a first-order model, presumes that more detention time (lower hydraulic loading) will result in greater metal removal, and that higher inlet concentrations will result in more removal. It is consistent with the concept of mass transfer of the metal to the reactive sediment layer, which is driven by the concentration difference between water and sediment pore-water metals. Younger states that "However, as the design of passive systems advances, it is likely that volume-based or retention-time based measures of performance will prove to be more appropriate in many circumstances." Tarutis *et al.* (1999) concluded that evidence showed that the first-order model was a better choice, and advocated the plug-flow form:

$$\frac{C_o}{C_i} = \exp\left(-\frac{kA}{Q}\right) \tag{11.17}$$

where

$k$ = areal rate constant, m/d

Data from 35 wetlands gave a mean value $k$ = 0.29 m/d (106 m/yr) for iron, and $k$ = 0.057 m/d (21 m/yr) for manganese (Tarutis *et al.*, 1999). Younger *et al.* (2002) reported $k$ = 0.105 m/d (38 m/yr) for iron, and $k$ = 0.061 m/d (22 m/yr) for manganese, at the Quaking House system.

Goulet *et al.* (2001) examined the efficacy of the first-order model for iron, manganese, copper, and zinc in a stormwater wetland in Kanata, Ontario, over a two-year period. These authors found that it was acceptable for some metals (e.g., zinc) but not for others (e.g., iron and manganese). They observed that such a model could not account for releases, and might also be affected by the low concentrations that existed in the incoming water. Crites *et al.* (1997) found exponentially decreasing profiles for zinc in water along the flow direction at the Sacramento wetlands (Figure 11.9), as indicated by Equation 11.17.

One of the premier uses of mesocosm experimentation is to control external factors, in an effort to elucidate processes. The efforts of Manyin *et al.* (1997) assist in the understanding of which model may be more realistic. Side-by-side mesocosms were fed iron solutions of varying concentrations and at varying flow rates. Data fit to Equation 11.17, across three inlet concentrations and four hydraulic loadings, produced a mean $k$ = 53 m/yr, with $R^2$ = 0.83. Thus, when there are not artifacts of variable pH, temperature, flow, or ancillary chemistry, the first-order model is quite effective in describing data. As a corollary, the use of an area-adjusted contaminant removal rate was entirely inappropriate for these controlled conditions.

Another approach to investigation of removal models relies upon the accumulation of metals in new wetland sediments. If for the sake of simplicity, the plug flow model is used as a basis for interpretation, it may be shown that exponential reductions in waterborne metals along the flow path result in the appearance of exponentially distributed

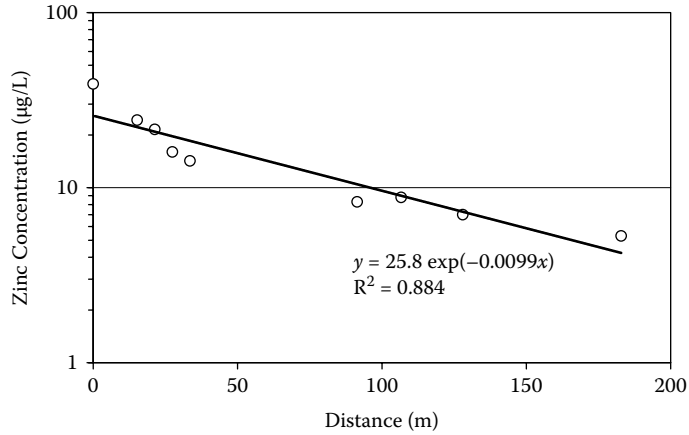



**FIGURE 11.9** Profile of water-phase zinc along the flow direction in Sacramento, California, Cell 7B in May 1995. The corresponding $k$ = 45 m/yr. (Data from Crites *et al.* (1995) *Removal of metals in constructed wetlands.* Proceedings of the 68th Annual WEFTEC Conference. Water Environment Federation, Alexandria, Virginia; and Crites *et al.* (1997) *Water Environment Research* 69(2): 132–135.)

top sediment accretions of those metals. The buildup of those new top sediments may be considered the result of the wetland carbon cycle, which produces a stable annual accretion for a fully developed wetland. When combined with Equation 11.17, the concentration in the new sediments can be computed at each location. The rate of deposition at some fractional distance $y$ along the wetland flow path is:

$$J_{\text{metal}} = kC(y) = kC_{\text{i}} \exp\left(-\frac{kAy}{Q}\right) \tag{11.18}$$

Then a mass balance on the top layer of deposited material determines the concentration in the surficial solid layer:

$$C_{\text{S}} = \frac{J_{\text{metal}}}{J_{\text{sed}}} = \frac{kC_{\text{i}}}{J_{\text{sed}}} \exp\left(-\frac{kAy}{Q}\right) \tag{11.19}$$

where

$C_{\text{S}}$ = top layer solid-phase metal concentration, mg/g
$C_{\text{i}}$ = inlet concentration, µg/L = mg/m$^3$
$J_{\text{metal}}$ = metal accumulation rate, mg/m$^2$·yr
$J_{\text{sed}}$ = sediment accumulation rate, g/m$^2$·yr
$y$ = fractional distance along flow path, dimensionless

A benefit of this analysis is that the temporal variability of water concentrations is averaged by the sampling of many months or years of accretion. The Sacramento, California, project sampled longitudinal profiles after 3.5 years of operation (Nolte and Associates, 1998b), and designated the new top sediments as the *A layer.* Interpretation of the Sacramento data was via the first-order areal model (Dombeck *et al.*, 1998). Results indicated that exponential declines explained a considerable amount of the variability in *A layer* solids concentrations (Figure 11.10). Rate constants fall in the range 22–71 m/yr. Thus it appears that the first-order model is applicable to field situations on a long-term average basis.

More detailed models have been proposed, but these generally lack calibration to multiple systems. A significant effort at detailed process modeling was the development and calibration of the Constructed Wetland Fate and Aquatic Transport Evaluation (CWFATE) model for the Sacramento, California project (Jones & Stokes Associates, 1993; Nolte and Associates, 1998a, b). CWFATE attempts to describe water budgets, lead biomass cycles, and partitioning, but not chemical precipitation. Flanagan *et al.* (1994) proposed a detailed model, and calibrated it for iron and aluminum at the Lick Run, Pennsylvania, wetland, but this model has not gained general acceptance. The PIRAMID project developed a proprietary metals model for mine water applications (PIRAMID Consortium, 2003a).

In this chapter, system data analyses are presented for percent concentration reduction, areal removal rates, and first-order rate constants.

## Storage in Plants

Plants are a secondary location for metal storage, compared to sediments. Furthermore, most of the metal found in plants is located in the roots and rhizomes (Table 11.15). Consequently, harvest of aboveground plant parts is not an effective means of removing metals from the wetland.

## Sediment Storage Concentrations

### A Well-Mixed Surficial Zone

There are two ways to consider the buildup of stored metals in a treatment wetland. The first presumes that metal storage occurs throughout the root zone of the FWS wetland, with transfers to sediments and roots driven by processes such as sorption and transpiration flows.

Additionally, wetland sediments contain a variety of organisms that ingest sediments over a wide range of depths, and redeposit their gut contents primarily at the sediment surface (Robbins, 1986). By such means, metals may reach the entire root zone layer, and continue to build up in concentration. As for other contaminants in wetlands, the major storage will be in the roots, soils, and sediments, rather than in aboveground tissues. Thus, this approach considers the root zone to be a well-mixed region, with ever increasing

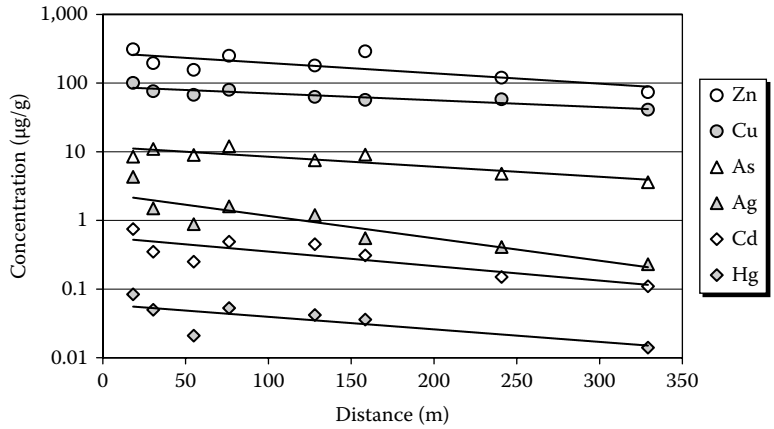



**FIGURE 11.10** Profiles of metals in sediments along the flow direction for Sacramento, California, Cell 7 in 1997. (Data from Nolte and Associates (1998a) *Sacramento Regional Wastewater Treatment Plant Demonstration Wetlands Project*. 1997 Annual Report to Sacramento Regional County Sanitation District, Nolte and Associates, Sacramento, California.) The corresponding areal *k*-values are:

| Metal | $R^2$ | *k* (m/yr) |
|---|---|---|
| Zn | 0.582 | 33 |
| Cu | 0.755 | 22 |
| As | 0.795 | 32 |
| Ag | 0.808 | 71 |
| Cd | 0.709 | 47 |
| Hg | 0.565 | 40 |

solid concentrations. The metal load removed contributes to the increase, in simple form, as:

$$\rho_b H \frac{dC_S}{dt} = J_{metal} \tag{11.20}$$

$$C_S = C_S^\circ + \frac{J_{metal} t}{\rho_b H} \tag{11.21}$$

where

$C_S^\circ$ = initial solid-phase metal concentration, mg/g
$H$ = root zone thickness, m
$\rho_b$ = solids bulk density, g/m$^3$
$t$ = time, yr

This method was applied by Nolte and Associates (1998a), to calculate the time required to reach toxicity sediment standards. For example, in the Sacramento, California, FWS wetland, copper was 27 mg/kg in 1996, and the Threshold Effects Limit (TEL) is 35.7 mg/kg (National Oceanic & Atmospheric Administration, 1999). The measured accretion of copper was 132 mg/m$^2$·yr. The top 25 cm of the wetland contained 50 kg/m$^2$ of dry solids (at $\rho_b$ = 200,000 g/m$^3$), and hence the rate of concentration increase was 2.65 mg/m$^2$·yr. Therefore, the increase from 27 to 35.7 mg/kg would take 3.3 years. Storage lifetimes for other metals ranged from zero (TEL exceeded for baseline condition) to 120 years for lead.

This calculation presumes vertical uniformity in the root zone, which is not typically observed. Further, it does not include the effects of new sediment deposition, which is very common in FWS treatment wetlands. Accretion, precipitation of metals, and downward diffusion with sorption all tend to create layering in the upper soil sediment horizon.

**TABLE 11.15**
**Fraction of Removed Metal Load Found in Plants after Five Years of Operation**

| Metal | Aboveground (%) | Belowground (%) |
|---|---|---|
| Ag | 0.0 | 2.0 |
| As | 0.6 | 10.1 |
| Cd | 0.0 | 13.3 |
| Cr | 2.2 | 16.8 |
| Cu | 0.6 | 5.5 |
| Hg | 0.0 | 6.7 |
| Ni | 0.3 | 4.7 |
| Pb | 2.0 | 11.8 |
| Zn | 0.4 | 6.1 |

*Source:* Adapted from Nolte and Associates (1998b) *Sacramento Regional Wastewater Treatment Plant Demonstration Wetlands Project: Five Year Summary Report 1994–1998*. Report to Sacramento Regional County Sanitation District, http://www.srcsd.com/cw.html, Nolte and Associates, Sacramento, California.

## Managed Peat Systems

When metal concentrations are high, accumulation on sediments is also high for many metals. Under high loadings, the

wetland ecosystem cannot generate enough sorption capacity via its carbon cycle. In some instances, the initial amount of substrate may provide for a long removal lifetime. For instance, Eger and Lapakko (1989) estimated lifetimes for four wetlands associated with nickel removal at the Dunka Mine site in Minnesota. At hydraulic loadings of 0.12–0.60 cm/d, and nickel concentrations of 0.9–15.4 mg/L, they estimated sorption capacity would last 20–780 years. In other cases, there may be a need to replenish the sorbent on a more frequent basis. In the wetland environment, that means digging out the substrate and finding appropriate means of disposal. Operational and maintenance costs so incurred must then be considered in the evaluation of a project. The alternative of using the peat as a sorbent in a controlled, mechanical apparatus should then be considered (Coupal and Lalancette, 1976; Sharma and Forster, 1993; Brown *et al.*, 2001; Fine *et al.*, 2005).

## The "Layer Cake" Assumption

The second way to interpret metal storage in FWS wetlands is to presume that all the current storage is contained in a newly formed top sediment layer, which does not mix with previous layers (Kadlec, 1998). This is a limiting concept, because there is very likely to be some sediment metal mixing as a result of bioturbation and vertical chromatographic flows.

Vertical stratification of FWS treatment wetland sediments/soils usually includes a top-most floc layer, which has been described as "a slurry of dark, decomposing, loosely structured detrital material that pours out when the sampler is tipped" and termed the *A layer* (Nolte and Associates, 1997). This layer has been observed in a range of treatment wetland types, ranging from those receiving secondary effluent in Sacramento, California, to those receiving agricultural runoff in South Florida (SFWMD, 2006). The thicknesses are about 5–30 cm; for instance, 11.3 ± 2.7 cm along the flow direction at Sacramento Cell 7 in 1996. Below this floc, there may be a second layer dominated by litter and sediment, termed the *B layer* by Nolte and Associates (1997). This *B layer* averaged 4.6 ± 1.0 cm for Sacramento Cell 7 in 1996. Below this, there existed a *C layer* of gleyed clay soil at Sacramento, in turn atop the base clays of the wetland basins. The metals content of these various layers were found to be markedly different, with 2–13-fold higher concentrations in the *A layer* than the *C layer* (Figure 11.11). It is probable that the layering pattern in concentrations is driven by processes of precipitation and sorption.

Metal storage in the top layer creates sustained sediment concentrations that reflect the dilution of new metal deposits by the accretion of new wetland solids. Such accretions result from microbial, algal, and macrophyte detritus production, and lead to replacement of floc layers, which are depleted by consolidation. In a sense, this concept amounts to plug flow of metals and biomass downward into the top sediments, or "conveyor-belt" deposition. Under this assumption, a mass balance on the top layer of deposited material determines the concentration in the surficial solid layer:

$$C_S = \frac{J_{metal}}{J_{sed}} \tag{11.22}$$

where

$C_S$ = top layer solid concentration, mg/g
$J_{metal}$ = cadmium accumulation rate, mg/m$^2$·yr
$J_{sed}$ = sediment accumulation rate, g/m$^2$·yr

For example, consider $J_{sed}$ = 200 dry g/m$^2$·yr of new sediments, originating from 1,000 dry g/m$^2$·yr of biomass production. Suppose the metal removal rate is $J_{metal}$ = 10 mg/m$^2$·yr. When this metal stream is diluted by the new sediment stream, a new, top sediment concentration of 50 µg/g (0.05 mg/g) is forecast. This may then be compared to the appropriate sediment quality criterion for the metal in question. This example may be continued to estimate the corresponding water concentration:

$$C = \frac{J_{metal}}{k} \tag{11.23}$$

For the example under consideration, if the value of $k$ = 50 m/yr, then $C$ = 0.2 mg/m$^3$ (µg/L). From the magnitudes of

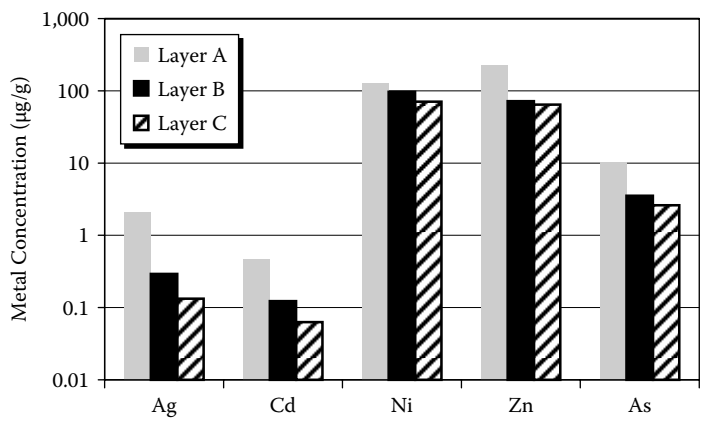



**FIGURE 11.11** The vertical layering effect on average sediment metal concentrations in the first 100 m of Sacramento, California, Cell 7. (Data from Nolte and Associates (1998a) *Sacramento Regional Wastewater Treatment Plant Demonstration Wetlands Project*. 1997 Annual Report to Sacramento Regional County Sanitation District, Nolte and Associates, Sacramento, California.)

the numbers in this hypothetical example, it is easily seen that if the wetland is effective, and there is elevated metal in the incoming water, it will be difficult to meet stringent sediment quality standards (Kadlec, 1998).

## 11.6 THE OXIDE FORMERS

### Iron

Iron is a metal that may occur at trace to high concentrations in wetland surface waters and sediments. It is required by plants and animals at significant concentrations. In plants, iron is an essential element in chlorophyll synthesis, cytochromes, and in the enzyme nitrogenase. In animals, iron is important in oxidative metabolism and is a key component in hemoglobin.

Iron at low to moderate concentrations is not generally regarded as a threat to human health or aquatic life. The U.S. EPA has recommended a continuous concentration criterion of 1,000 µg/L for protection of freshwater aquatic environments, and a drinking water human health criterion of 300 µg/L (U.S. EPA, 2002b). The province of Ontario, Canada, has a lower standard for protection of aquatic life, also 300 µg/L. Perhaps the greater concern is for the blanketing effect of thick deposits of iron precipitates in wetlands designed to treat high iron concentrations (Kelly-Hooper, 1999).

High concentrations of soluble iron in surface water and wetland systems may result from natural or artificial iron sources, typically as seeps of ferrous iron and iron sulfides (pyrites) from anaerobic groundwaters. Iron bacteria that produce ocher, such as *Leptothrix ochracea* and *Spirophyllum ferrugineum*, derive their energy needs from the oxidation of reduced iron. These bacteria typically occur in wetland areas where anoxic waters meet aerated surface conditions, such as upwelling springs or other venting groundwaters. At such locations, reddish brown flocculent deposits form.

Iron in the wetland waters may be dissolved or particulate. Most reported wetland studies do not specify which forms were determined. This may be a critical unresolved issue, because there may be very large amounts of suspended iron in wetland waters. Gammons *et al.* (2000b) report that the iron concentrations between filtered and unfiltered samples can differ by a factor of up to 100. Their data may be approximated as unfiltered iron equal to the square root of filtered iron, over the range 10–5,000 µg/L.

### Wetland Storage and Processing of Iron

In an oxygenated environment, ferric iron is present as insoluble oxyhydroxides, denoted as FeOOH. If there is not sufficient alkalinity in the water, the reaction produces acidity:

$$Fe^{2+} + \frac{1}{4}O_2 + H^+ \rightarrow Fe^{3+} + \frac{1}{2}H_2O \qquad (11.24)$$

$$Fe^{3+} + 2H_2O \rightarrow FeOOH_{(s)} + 3H^+ \qquad (11.25)$$

However, if there is sufficient alkalinity, removal of iron to precipitates is not accompanied by a decrease in pH:

$$Fe^{2+} + \frac{1}{4}O_2 + 2HCO_3^- \rightarrow FeOOH + 2CO_2 + \frac{1}{2}H_2O \qquad (11.26)$$

The water is deemed acidic for iron removal if the ratio of iron to $CaCO_3$ alkalinity is greater than 1.1 (Younger *et al.*, 2002).

Oxidation and reduction of iron occurs relatively easily depending on redox potential and pH (Faulkner and Richardson, 1989). $Fe^{3+}$, or ferric iron, is the dominant form under oxidized conditions ($E_h > 0$ at pH ≥ 6.5). $Fe^{2+}$, or ferrous iron, is the dominant form under reduced conditions in wetlands and other aquatic environments. $Fe^{3+}$ forms stable complexes with a variety of ligands. It joins with the hydroxide ion in surface waters to form reddish-brown ferric hydroxide ($Fe(OH)_3$), which is also known as ocher. Ocher is insoluble and either settles to the bottom sediments or remains in suspension, adsorbed to living and dead organic matter (see Figure 7.10). Other important compounds formed by ferric iron include ferric phosphate ($FePO_4$), iron-humate complexes, and ferric hydroxide-phosphate complexes.

Ferric iron is reduced to the ferrous form under anaerobic conditions. The ferrous iron is more soluble, resulting in the release of dissolved iron and associated anions such as phosphate from anaerobic sediments in wetlands. The formation of this soluble ferrous iron may be controlled somewhat by sulfide, which forms the relatively insoluble ferrous sulfide (FeS). Sulfide formation is written as:

$$Fe^{2+} + HS^- \rightarrow FeS + H^+ \qquad (11.27)$$

The required $HS^-$ is microbially generated, and occurs preferentially in organic environments by the reduction of sulfate (see Equations 11.1, 11.2, and 11.3).

The role of sulfate-reducing bacteria (SRB) in the cycling of iron and sulfur was studied in a young constructed wetland located in Kanata, Ontario, Canada (Fortin *et al.*, 2000). Sediments and water samples were collected over the course of one year within each of three FWS cells. SRB populations were largest during the cold winter months, when the temperature of the water was 1°C. The presence of high-SRB populations also corresponded to highly anoxic conditions within the sediments and to a decrease of sulfate concentrations, suggesting that cold temperature did not affect the activity of SRB. The results indicated that iron and sulfur cycling in the constructed wetland was active throughout the year, especially in the cold winter months. This suggests that iron removal in wetlands can be effective in temperate climates, even though the temperature of the water decreases drastically during the winter.

#### *Soils and Sediments*

Wetland soils can contain large amounts of iron, especially when exposed to metalliferous waters (Table 11.16). On a dry basis, ferric oxide contains 70% iron by weight (700,000 mg/kg), and this represents an upper limit to sedimentary iron concentrations in oxic wetland waters. Iron sulfides contain 53% ($FeS_2$) and 66% (FeS) iron. Such mineral precipitates are diluted by newly formed organic materials in the wetland

**TABLE 11.16**
**Iron Content of Top Sediments in a Variety of Wetlands**

| Location | Notes | Water Source | Iron (mg/kg) | Reference |
|---|---|---|---|---|
| Michigan | Fen | Natural | 4,924 | Faulkner and Richardson (1989) |
| North Carolina | Pocosin | Natural | 2,370 | Faulkner and Richardson (1989) |
| Maryland | Bog | Natural | 5,710 | Faulkner and Richardson (1989) |
| Maryland | Swamp | Natural | 5,410 | Faulkner and Richardson (1989) |
| North Carolina | Swamp | Natural | 1,301 | Faulkner and Richardson (1989) |
| Ireland | Salt marsh | Natural | 6,000–38,000 | Doyle and Otte (1997) |
| Germany | Lake Schöhsee | Natural | 58,000 | Wetzel (1975) |
| Panel, Ontario | Cattail marsh | Urban stormwater | 12,000 | Goulet and Pick (2001) |
| Monahan, Ontario | Cattail marsh | Urban stormwater | 2,500 | Goulet and Pick (2001) |
| Falconbridge, Ontario | Cattail marsh | Acid mine drainage | 1,500 | Goulet and Pick (2001) |
| Riverwalk, Ontario | Cattail marsh | Tailings leachate | 1,500 | Goulet and Pick (2001) |
| West Page Swamp, Idaho | Cattail, Arrowhead | Tailings leachate | 115,500 | DeVolder *et al.* (2003) |
| Widows Creek, Alabama | Cattail, Juncus | Tailings leachate | 60,000–85,000 | Ye *et al.* (2001a,b) |
| Show Low, Arizona | Pintail marsh | Municipal | 30,575 | NADB database (1998) |
| Tres Rios, Arizona | Four wetlands | Municipal | 18,857 | Wass, Gerke, and Associates (2002) |
| Sacramento, California | Seven wetlands | Municipal | 17,096 | Nolte and Associates (1998) |
| Champion Paper, Florida | Pilot | Pulpand paper | 9,400 | NADB database (1998) |
| Monroe Co., New York | Pilot | Leachate | 2,560–2,720 | Eckhardt *et al.* (1997) |
| Poland | 11 lakes | Brown coal pits | 115–21,500 | Samecka-Cymerman and Kempers (2001) |

environment, and lesser concentrations are observed. For instance, Doyle and Otte (1997) measured 6,000–40,000 mg/kg, and higher values in the rhizosphere and near worm burrow walls.

Freeze-coring and analysis of the wetland substrates indicated that total sulfur was present in three forms, in the following proportions (Younger, 2000): FeS: 35%; $FeS_2$: 31%; S°: 34%.

On the basis of these observations, it was postulated that pH rise was due to the consumption of protons via reactions involving reduction of ferric hydroxide and precipitation of elemental sulfur. The removal of iron from solution and formation of significant quantities of S° is consistent with the following coupled reactions:

$$5H^+ + 2\,FeOOH + HS^- \rightarrow 2\,Fe^{2+} + S^\circ + 4H_2O \qquad (11.28)$$

$$Fe^{2+} + H_2S + 2HCO_3^- \rightarrow FeS + 2H_2O + 2CO_2 \qquad (11.29)$$

Goulet and Pick (2001) found that the presence of cattails had little effect on the partitioning of iron in shallow wetland sediments in FWS wetlands. Studies at four Ontario treatment wetlands showed total iron in the sediments of 2,000–12,000 mg/kg, with sediment organic content of 8–20%. About half of the sediment iron was in reactive forms, oxides, monosulfides, or sorbed on organic matter. The balance was dominated by forms associated with either the pyrite (one wetland) or the silicate fraction of the sediment (three wetlands).

### *Wetland Plants*

Metals reach plants via their fine root structure, and most are intercepted there. Some small amounts may find their way to stems, leaves, and rhizomes. Upon root death, some fraction of the metal content may be permanently buried, but there are no data on metal release during root decomposition. However, wetland plants bring oxygen to their root zone to maintain respiration, and some fraction is lost by radial diffusion away from the roots. This creates small aerobic zones near the roots, in which iron precipitates may form. These are termed *iron plaque*.

Nonetheless, some iron is taken up into aboveground tissues. Iron occurs in wetland plants at concentrations ranging from about 200–2,000 mg/kg dry mass (Vymazal, 1995). Plant roots contain a much higher concentration of iron than stems or leaves (Table 11.17). Uptake by plants and algae may be for purposes of growth enhancement, or at higher metal concentrations for protective purposes. Biomagnification of iron does not occur.

A common concept of wetland treatment is the perceived risk of seasonal release of contaminants during winter, when wetland macrophytes die back. This theoretical risk was investigated experimentally in mesocosm experiments on plant litter collected from long-established mine water treatment wetlands in the United Kingdom (Batty and Younger, 2002). Metals were not released from the plant litter; and iron concentrations in the litter increased after 6 months of decomposition, which was attributed to adsorption. Field studies undertaken within the PIRAMID project (PIRAMID Consortium, 2003a) found that wetlands were net sinks for iron in all seasons.

## Performance of Wetlands for Iron Removal

Wetlands interact strongly with iron in a number of ways, and thus are capable of significant metal removal. Three major mechanisms are operative:

**TABLE 11.17**
**Iron Content of Above- and Belowground Plant Parts in a Variety of Wetlands**

| Location | Wetland | Plant | Above (μg/g) | Below (μg/g) | Reference |
|---|---|---|---|---|---|
| TVA Mussel Shoals, Alabama | FWS CW | *Typha* spp. | 45–142 | 1,011–7,437 | NADB database (1998) |
| New York | FWS CW | *Typha glauca* | 292 | 10,745 | Bernard (1997) |
| New York | FWS CW | *Typha glauca* | 67 | 18,006 | Bernard (1997) |
| Widows Creek, Alabama | FWS CW | *Typha latifolia* | 1,217 | 68,469 | Ye *et al.* (2001a,b) |
| Coeur d'Alene, Idaho | FWS Nat | *Typha latifolia* | 200 | — | DeVolder *et al.* (2003) |
| Wisconsin | FWS Nat | *Typha latifolia* | 138 | 10,55 | Vymazal (1995) |
| Germany | SSF CW | *Typha angustiolia* | 1,100 | — | Vymazal (1995) |
| 11 lakes, Poland | FWS Nat | *Typha angustifolia* | 350 | — | Samecka-Cymerman and Kempers (2001) |
| TVA Mussel Shoals, Alabama | FWS CW | *Phragmites australis* | 112–161 | 2,533–4,547 | NADB database (1998) |
| Nucice, Czech Republic | SSF CW | *Phragmites australis* | 139 | — | Vymazal and Krása (2005) |
| New York | FWS CW | *Phragmites australis* | 618–799 | 7,060–9,280 | Eckhardt *et al.* (1997) |
| Břehov, Czech Republic | SSF CW | *Phragmites australis* | 74 | 3,677 | Vymazal *et al.* (2006) |
| 11 lakes, Poland | FWS Nat | *Phragmites australis* | 1,053 | — | Samecka-Cymerman and Kempers (2001) |
| TVA Mussel Shoals, Alabama | FWS CW | *Phalaris arundinacea* | 89–309 | 2,445–8,352 | NADB database (1998) |
| Nucice, Czech Republic | SSF CW | *Phalaris arundinacea* | 323 | — | Vymazal and Krása (2005) |
| Břehov, Czech Republic | SSF CW | *Phalaris arundinacea* | 70 | 3,383 | Vymazal *et al.* (2006) |
| 11 lakes, Poland | FWS Nat | *Phalaris arundinacea* | 1,202 | — | Samecka-Cymerman and Kempers (2001) |
| TVA Mussel Shoals, Alabama | FWS CW | *Scirpus acutus* | 47–107 | 1,820–2,754 | NADB database (1998) |
| TVA Mussel Shoals, Alabama | FWS CW | *Scirpus cyperinus* | 83–723 | 1,185–2,228 | NADB database (1998) |
| Czech Republic | FWS Nat | *Scirpus lacustris* | 129 | — | Vymazal (1995) |
| Germany | SSF CW | *Scirpus lacustris* | 780 | — | Vymazal (1995) |
| 11 lakes, Poland | FWS Nat | *Scirpus lacustris* | 430 | — | Samecka-Cymerman and Kempers (2001) |
| Widows Creek, Alabama | FWS CW | *Juncus effusus* | 320 | 41,318 | Ye *et al.* (2001a,b) |
| Coeur d'Alene, Idaho | FWS Nat | *Sagittaria latifolia* | 220 | — | DeVolder *et al.* (2003) |
| Ireland | FWS CW | Algae | 39 | — | O'Sullivan *et al.* (2000) |
| Ireland | FWS CW | Algae | 108 | — | O'Sullivan *et al.* (2000) |

*Note:* FWS = free water surface; SSF = subsurface flow; CW = constructed wetland; Nat = natural wetland

1. Binding to soils, sediments, particulates, and soluble organics
2. Precipitation as insoluble salts, principally sulfides and oxyhydroxides
3. Uptake by plants, including algae (however, after the plant senescence or algal die back most iron is leached out)

Information on the effects of wetlands on iron concentrations has been reported at a low level of detail, with emphasis on removal percentages. Three principal categories of influent waters are mine drainage, landfill leachates, and municipal wastewater. Mine drainage control dominates the applications, with well over a hundred systems in place and reporting data by 1990 (Weider, 1989). Many landfill leachates contain enough iron to induce monitoring of wetland systems built for leachate improvement. Iron is not typically monitored in municipal wastewater wetlands, and there are but few data sets that exist.

Example performances are shown in Table 11.18. Inflow and outflow concentrations vary over 1,000-fold. Removal percentages are high for mine waters and leachates, but lower or negative for waters with low iron content. SSF systems, due to prevailing anoxic/anaerobic character of the filtration bed, usually do not perform well in terms of iron removal. The filtration bed becomes more anaerobic over the period of operation and majority of iron is reduced to more soluble ferrous compounds which are washed out of the system. If hydrogen sulfide is present as a consequence of sulfate reduction (Equations 11.1 and 11.2), iron may form insoluble ferrous sulfides (Equation 11.27) and is deposited in the bed.

### *Coal Mine Drainage Wetlands*

Information on iron removal in wetlands is available primarily from acid mine drainage (AMD) wetland treatment systems in the United States (Girts *et al.*, 1987; Kleinmann and Hedin, 1989; Hedin, 1989). However, treatment wetlands also became widely used in the United Kingdom in the 1990s. Younger (2000) lists 24 wetland systems, which fall into two principal categories:

- *Aerobic FWS wetlands,* usually vegetated by *Phragmites australis*. These are typically used for iron reduction in net-alkaline waters.
- *FWS with an anoxic compost substrate,* typically vegetated by *Phragmites australis.* Because of the organic substrate, these are called anaerobic FWS wetlands. These are typically used for iron reduction in more acidic waters.

Of the 137 AMD wetland treatment sites reviewed by Wieder (1989), 66% had influent iron concentrations less than 50 mg/L. An average total iron concentration of 60.6 mg/L was reduced to an average outflow concentration of 15.4 mg/L,

**TABLE 11.18**
**Removal of Iron in Constructed Wetlands**

| System | Location | Water Source | Inlet (mg/L) | Outlet (mg/L) | Reduction (%) | Removal ($g/m^2 \cdot yr$) | Reference |
|---|---|---|---|---|---|---|---|
| **FWS** | | | | | | | |
| Eleven Systems | United Kingdom | Coal mine water | 19.6 | 2.53 | 87 | — | Younger (2000) |
| Ten Systems | Tennessee Valley | Coal acid mine | 58.3 | 9.36 | 82 | 1,028 | Brodie (1990) |
| Albright | Pennsylvania | Coal mine water | 2.45 | 0.33 | 87 | 56 | Hoover *et al.* (1998) |
| Springdale | Pennsylvania | Coal mine water | 12.46 | 0.27 | 98 | 436 | Hoover *et al.* (1998) |
| Musselwhite | Ontario | Metal mine water | 0.155 | 0.038 | 75 | 22 | Bishay and Kadlec (2005) |
| Kanata Monahan | Ontario | Metal mine water | 0.36 | 0.27 | 25 | 11 | Goulet and Pick (2001) |
| Elliot Lake Panel | Ontario | Metal mine water | 15 | 0.3 | 98 | 141 | Goulet and Pick (2001) |
| Bahco | Argentina | Metal processing | 13.7 | 0.38 | 97 | 243 | Maine *et al.* (2004a) |
| Esval | Norway | Leachate | 45.0 | 9.0 | 80 | 394 | Mæhlum (1997) |
| Bolstad | Norway | Leachate | 32.00 | 1.28 | 96 | 561 | Mæhlum (1997) |
| Escambia County | Florida | Leachate | 3.500 | 0.361 | 90 | 128 | DeBusk (1997) |
| Monroe County | New York | Leachate | 51 | 15 | 71 | 125 | Eckhardt *et al.* (1997) |
| New Hanover County | North Carolina | Leachate | 1.746 | 0.134 | 92 | 1.3 | Unpublished data |
| Isanti-Chisago | Minnesota | Leachate | 0.790 | 0.340 | 57 | 16 | Kadlec (2003c) |
| Estevan | Saskatchewan | Municipal lagoon | 0.296 | 0.197 | 33 | 1 | Unpublished data |
| Brookhaven | New York | Domestic | 7.40 | 2.10 | 72 | 24 | Hendrey *et al.* (1979) |
| Twelve Systems | Arcata, California | Municipal lagoon | 0.60 | 0.55 | 8.3 | 2 | NADB database (1998) |
| Blue Pond | Tennessee | Groundwater | 3.07 | 0.61 | 80 | 31 | Gessner *et al.* (2005) |
| Hidden River | Florida | Urban runoff | 0.28 | 0.34 | –21 | 0 | Carr and Rushton (1995) |
| Six Systems | Champion, Florida | Pulp and paper | 0.330 | 0.530 | –61 | –3 | NADB database (1998) |
| Orange Eastern | Orlando, Florida | Municipal 3° | 0.017 | 0.067 | –294 | –0.34 | NADB database (1998) |
| Tres Rios | Phoenix, Arizona | Municipal 3° | 0.200 | 0.233 | –17 | –2.9 | Wass, Gerke, and Associates (2002) |
| **Median FWS** | | | | | **78** | **24** | |
| **HSSF** | | | | | | | |
| Mořina | Czech Republic | Municipal | 0.40 | 0.18 | 55 | 1.8 | Unpublished data |
| Břehov | Czech Republic | Municipal | 1.5 | 1.64 | –11 | –3.7 | Unpublished data |
| Slavošovice | Czech Republic | Municipal | 0.95 | 2.45 | –158 | –14.6 | Unpublished data |
| **Median HSSF** | | | | | **–11** | **–3.7** | |

for an average iron removal efficiency of 58.2% and a median value of 80.9% (Wieder, 1989). This median is clearly also close to that experienced at the U.K. sites (Table 11.18). This treatment efficiency was found to weakly correlate directly with wetland area and inversely with wetland depth. An average iron removal rate of 3,650 kg/ha·yr at pH 6 in AMD treatment wetlands was given by Kleinmann and Hedin (1989). Because the iron removal rate is correlated with iron loading rate (Hedin and Nairn, 1990), lower removal rates are expected at low influent concentrations.

### *Landfill Leachate Wetlands*

Scrap iron items discarded to landfill create an underground source of iron, which then appears as a constituent of leachate. Raw leachate concentrations may be as high as 500 mg/L, mostly in soluble form because of the anaerobic condition of the water in the pile (McBean and Rovers, 1999). At such high concentrations, iron precipitates pose a serious clogging threat even for FWS wetlands. Consequently, aeration and precipitation are often included as pretreatment steps (Hoover *et al.*, 1998; Loer *et al.*, 1999). Oxidation/precipitation basins are open water impoundments designed to provide aeration for precipitation of aqueous iron, detention time to settle precipitates, and storage volume for accumulating precipitate sludge. These basins are a key component in passive systems where iron is present. A detention time of at least 24 hours is recommended to produce a clear water discharge (Hoover *et al.*, 1998). The leachates in Table 11.18 all had lower iron than that in raw leachates due to aeration basin pretreatment or dilution.

FWS wetlands produce considerable further reduction in iron, with a median removal of 85% for the six systems in Table 11.18. Subsurface systems may also be used. For instance, Surface *et al.* (1993) measured an iron removal efficiency of 78.6% in a HSSF wetland planted with common reed. Bulc *et al.* (1997) reported an iron removal of 80% in a landfill leachate HSSF system in Slovenia with outflowing concentration of 10 mg/L.

### *Other Water Sources*

A variety of other wastewaters have been subjected to wetland treatment, and in some few cases the iron content of inflows and outflows has been measured. This is usually ancillary monitoring, which does not target regulatory requirements. Byekwaso *et al.* (2002) found the iron reduction of 27% in a FWS-HSSF system in Uganda treating cobalt extracting wastes. The iron

content of pretreated municipal wastewater is likely to be less than 1 mg/L in many regions, although higher in regions with high occurrence of iron minerals (Table 11.18). For example, Vymazal (2003) reported reduction of Fe from 1.1 to 0.72 mg/L in a HSSF system treating municipal sewage near Prague in the Czech Republic. However, other studies from the Czech Republic (Table 11.18) showed rather negative removal effects for HSSF systems due to conditions discussed above. When iron concentrations are very low, the use percent removal is no longer an appropriate measure, because very small excursions in concentration can greatly affect the percentage. Nonetheless, removals may or may not occur for very low iron content waters (see Table 11.18).

## Examples and Models for Iron Removal in Treatment Wetlands

It is likely that wetland removal performance in FWS systems is area-specific (Younger *et al.*, 2002), rather than volume-specific. Three simple predictive approaches have been suggested:

- *Constant percent removal.* Specification of a supposedly constant percentage removal.
- *Zero-order removal.* Specification of a supposedly constant areal removal rate, in g Fe/m$^2$·d. This corresponds to zero-order removal kinetics, for which removal is independent of the iron concentration.
- *First-order removal.* The reduction rate is proportional to the iron concentration (Tarutis *et al.*, 1999). Some of the contributing processes are first-order, such as oxidation of ferrous iron and bacterial sulfate reduction (Younger *et al.*, 2002), as is the process of particulate settling of precipitates. However, global removal does not necessarily follow such a model.

Data from 35 FWS natural wetlands receiving inputs of ferruginous acid mine drainage in western Pennsylvania served as a basis for evaluating the merits of these three approaches. Stark and Williams (1995) found that percentage removal was a better index of treatment than areal removal. Nevertheless, a fixed areal removal rate of 10 g Fe/m$^2$·d remains the accepted design guideline for high levels of removal, while 20 g Fe/m$^2$·d is allowable to cause considerable improvement (Hedin *et al.*, 1994; PIRAMID Consortium, 2003a).

Tarutis *et al.* (1999) further analyzed the 35-wetland data set, and concluded that percentage removal should not be used, and that areal removal did not separate the effects of inlet concentration and flow rate. They recommended the first-order model (Equation 11.17), and found a median rate constant of 0.18 m/d (66 m/yr) (N = 35) for calibrations of a plug flow version of the model.

Manyin *et al.* (1997) performed a factorial experiment varying flow rate and inlet concentrations to wetland mesocosms. They found that outlet iron concentrations increased with increasing inlet iron loadings. Analysis of their data produces a rate constant of 42 m/yr according to Equation 11.17.

More complex models have been proposed, but these have not yet reached a point of usefulness in wetland design. For instance, the proposed model of Flanagan *et al.* (1994) presupposes that removal is to sulfides, and contains no sedimentation step for ferric oxyhydroxides. Modeling software has been developed allowing simulation wetland systems (NOAH2D) (PIRAMID Consortium, 2003b). NOAH2D simulates overland flow and solute transport through reed beds, allowing for frictional resistance to flow offered by wetland vegetation, and also allows for exchange with (and hyporheic flow within) permeable wetland bed sediments. This code is a research tool, rather than software available for design. The complexity of the NOAH2D code requires very long run times, and full calibration is acknowledged as unlikely by the authors (Younger *et al.*, 2002).

The Manyin *et al.* (1997) findings suggest the use of a graphical representation of wetland effluent iron versus inlet iron loading. These variables are completely independent, and avoid the artifact of spurious correlations caused by the inclusion of a common factor in both ordinate and abscissa. Information from 46 constructed wetlands was used to form such a

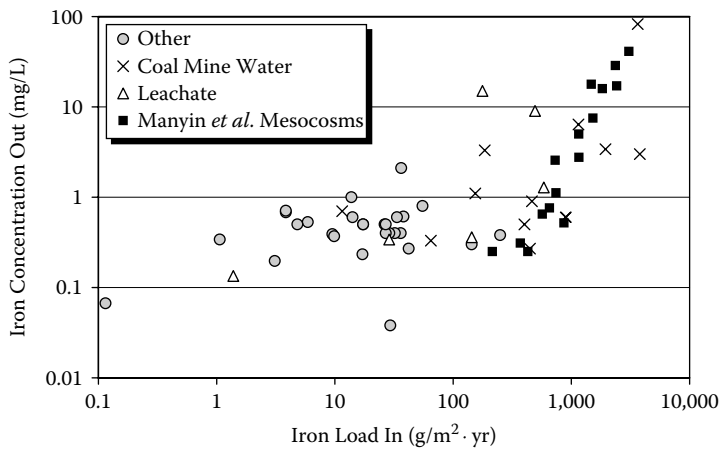



**FIGURE 11.12** Outlet iron concentrations from FWS wetlands at various loading rates.

display (Figure 11.12), together with the Manyin *et al.* (1997) mesocosm data. There are two zones apparent on this graph: above about 100 g/m$^2$·yr loading, there is an increase in outlet iron with increased iron loading; below about 100 g/m$^2$·yr loading, there is a much more gradual increase in outlet iron with increased iron loading. The 100 g/m$^2$·yr loading also represents the maximum that may be imposed without wetland effluent concentrations exceeding 1 mg/L. The *k*-values previously cited were not determined or verified for the low loading region.

In the low loading region, behavior probably reflects localized recycling of iron to and from precipitates and adsorbed forms, in response to localized variations in pH and mediated by diffusion to and from the water column. Background levels may also result from suspended particulate forms of iron. Uptake into aboveground macrophyte plant parts is low, but larger amounts of iron are found in roots, so that overall plant cycling can be of importance. Batty and Younger (2002) found that where dissolved iron concentrations in wetland waters were at or below 1 mg/L, direct uptake of iron by plants could account for 100% of iron removal. This finding explained why aerobic reed-beds removed dissolved iron at far greater rates than would be anticipated on the basis of the first-order kinetics of $Fe^{2+}$ oxidation. Batty and Younger (2002) also found 1 mg/L iron was also an optimum for healthy growth of *Phragmites australis:* at greater concentrations, the plants were not as productive, while at lower concentrations were healthy.

### *Variability of Iron Removal in Treatment Wetlands*

Intersystem variability in the Stark and Williams (1995) data sets was relatively high, with removal of 64% ± 28% (mean ± SD). It was less in the Manyin *et al.* (1997) mesocosms, with removal of 89% ± 13% (mean ± SD). The mesocosms were operated in the laboratory, and had no variability in depth, aspect ratio, weather conditions, or substrate; and were all operated at circumneutral pH.

Intrasystem variability has not been reported for continuous flow wetlands, but it is high for event-driven wetlands. For example, the coefficient of variation of outlet iron from the Hidden River wetland in Florida was 4.7 (ASCE, 2003).

## Summary

Iron is effectively removed by treatment wetlands over the high end of the concentration range, which is typical of acid mine drainage and landfill leachates. Rates are rapid, and significant loadings may be removed. At low concentrations, and loadings below 100 g/m$^2$·yr, iron cycling creates low concentrations in the water column. Removal is to precipitates, either oxyhydroxides or sulfides. Accumulation of these materials is a factor in the ecological status of the wetland.

# Aluminum

Aluminum occurs naturally in surface waters, to a small extent in the hydrated ionic forms, and to a greater extent complexed with silicates in a colloidal form. Aluminum solubility varies with pH. It is least soluble at a pH of 7 and increases in solubility as $Al^{3+}$, $Al(OH)^{2+}$, and $Al(OH)_2^+$ ions at $4 < pH < 5$; and as aluminate ion $Al(OH)_4^-$ at $pH > 9$ (Gensemer and Playle, 1999). Aluminum precipitates as amorphous $Al(OH)_3$, which may slowly form the mineral Gibbsite (Berkowitz *et al.*, 2005). Precipitation is rare in natural waters, but is of interest in treatment processes that rely upon addition of aluminum chlorides or sulfates for purposes of phosphorus removal.

## Wetland Processing and Storage of Aluminum

Aquatic systems typically contain low concentrations of total aluminum. In the Adirondacks region of the eastern United States, 203 lakes had a mean pH = 6.3, and a mean total Al = 138 µg/L. In Florida, 168 lakes had mean pH = 6.3 and a mean total Al = 89 µg/L. Lakes are typically net sinks for aluminum, with 10–50% retention for low pH, and 70% for circumneutral conditions (Gensemer and Playle, 1999). Organic complexation occurs in natural waters, as binding to humic substances. Trivalent cations such as $Al^{3+}$ are more susceptible to binding to organic ligands than divalent cations. Peat based wetlands often provide the conditions of low pH that foster $Al^{3+}$, and hence dissolved organic matter, or dissolved organic carbon, is an important factor in wetland aluminum water chemistry.

Aluminum is toxic to many species of algae, with effects being observed over a range of a few hundred to a few thousand µg/L total aluminum. However, aquatic invertebrates are much less affected, and do not biomagnify aluminum. Wetland macrophytes are tolerant of high aluminum concentrations, in the thousands of µg/L total aluminum. Salmonid fish are susceptible to a variety of effects for concentrations of a few hundred to a few thousand µg/L. U.S. EPA (1988a) ambient water quality standards are 87 µg/L (chronic) and 750 µg/L (acute), but research is currently in progress to improve these numbers.

Aluminum uptake by emergent and floating plants (*Typha* and *Lemna*) was found mainly into the roots and rhizomes (Goulet *et al.*, 2005). In contrast, there was no clear pattern for submerged plants (*Utricularia* and *Potamogeton*). Roots also had the highest aluminum concentrations at the Tres Rios, Arizona, treatment wetlands (Table 11.19; Wass, Gerke, and Associates, 2002). Sediments generated in treatment wetlands are often high in aluminum, with values in treatment wetlands ranging from 1.4% (Tres Rios, Arizona) to 4% (Sacramento, California) (Nolte and Associates, 1998b).

The process of phosphorus adsorption onto aluminum hydroxides has seen extensive application to water treatment and lake eutrophication management as a means of controlling excess dissolved phosphorus. Alum or polyaluminum chloride additions are designed to form a floc of insoluble $Al(OH)_3$, which in turn adsorbs phosphorus. In wetlands, without coagulation, this floc settles slowly or not at all, leaving particulate aluminum and phosphorus in suspension (Bachand and Richardson, 1999). However, alum addition is a pretreatment step for waters sent to wetlands for further polishing. Wetlands are also the recipients of water treatment backwash and sludge (Kaggwa *et al.*, 2001). Consequently,

**TABLE 11.19**
**Aluminum Concentrations in the Tissues of Wetland Plants in a Municipal Wastewater Treatment Wetland**

| Plant | Scientific Name | Shoots (mg/kg) | Roots (mg/kg) | Reference |
|---|---|---|---|---|
| Three square bulrush | *Scirpus olneyi* | 12 | 214 | Wass, Gerke, and Associates (2002) |
| Cattails | *Typha* spp. | 23 | 262 | Wass, Gerke, and Associates (2002) |
| Pennywort | *Hydrocotyle* spp. | 28 | 254 | Wass, Gerke, and Associates (2002) |
| Soft stem bulrush | *Scirpus validus* | 11 | 168 | Wass, Gerke, and Associates (2002) |
| Common reed | *Phragmites australis* | 39 | 2,177 | Vymazal *et al.* (2006) |
| Common reed | *Phragmites australis* | 103 | — | Samecka-Cymerman and Kempers (2001) |
| Reed canarygrass | *Phalaris arundinacea* | 1,578 | — | Samecka-Cymerman and Kempers (2001) |
| Reed canarygrass | *Phalaris arundinacea* | 43 | 2,584 | Vymazal *et al.* (2006) |
| Narrow-leaved cattail | *Typha angustifolia* | 54 | — | Samecka-Cymerman and Kempers (2001) |
| Giant bulrush | *Scirpus lacustris* | 82 | — | Samecka-Cymerman and Kempers (2001) |
| Soft rush | *Juncus effusus* | 131 | — | Samecka-Cymerman and Kempers (2001) |

aluminum is sometimes a contaminant in treatment wetland influents, either by intentional or accidental discharges.

Aluminum is a strong determinant of the phosphorus adsorption capacity in wetland soils (Reddy and D'Angelo, 1994). The phosphorus adsorption process in treatment wetlands is a temporary mechanism, which is exhausted when all antecedent soil sorption sites are used. However, the amount of phosphorus that may be bound to wetland soils forms an important part of the wetland start-up capacity for phosphorus removal. A phosphorus removal project has been implemented using aluminum sludge from a water treatment plant as a soil amendment (SAIC, 2005).

There are two main groups of projects that have measured aluminum removal in wetlands:

- FWS wetlands, which typically receive low concentrations and loads of aluminum
- SSF wetlands, which, for aluminum, typically treat acid mine waters with quite high concentrations and loads

### *Aluminum Removal Processes*

Mine waters with pH less than 4 commonly contain high concentrations of aluminum (>10 mg/L) (Younger *et al.*, 2002). Aluminum is present as trivalent Al, and the currently accepted removal reaction is:

$$Al^{3+} + 3H_2O \rightarrow Al(OH)_3 + 3H^+ \qquad (11.30)$$

The precipitate is amorphous, white, and of low density. It may later crystallize to gibbsite. These solids are common components of many natural soils, and pose no problem as new wetland sediments and soils (Younger *et al.*, 2002). In the presence of dissolved sulfate, hydroxysulfate precipitates may form as well.

## Performance of Wetlands for Aluminum Removal

Examples of wetland reductions in concentration and mass removals are shown in Table 11.20. Performance is variable across systems, with a median concentration reduction of 50%. Wieder (1989) examined 20 acid mine wetlands for reductions, and found a median of 47.7%, and a 75th percentile of 78.9%. Wieder (1989) also found one quarter of the acid mine wetlands had zero reduction. Vymazal and Krása (2003) reported decrease of aluminum concentration from 451 µg/L to less than 40 µg/L in a 62-m long HSSF constructed wetland in the Czech Republic. The major decrease (from 451 µg/L to 65 µg/L) occurred within the first 15 m of the bed. Other results from HSSF wetlands treating municipal wastewater (Table 11.20) exhibited good removal of aluminum (53–90%).

Removal is to accretion in sediments, which form a source of potential return fluxes of aluminum, most likely as particulates. In a mesocosm study, Wieder *et al.* (1990) found that although aluminum was initially removed in a wetland, this process decreased with time, and the wetland began to export aluminum to downstream waters. Wieder *et al.* (1990) found aluminum concentrations of 10 mg/L were found to be toxic to cattails, leading to their mortality and release of sorbed aluminum. Wieder *et al.* (1988) found the total aluminum content of the peat increased from 2,375 to 13,634 mg/kg dry mass, with the majority bound as organic and oxide compounds.

Flanagan *et al.* (1994) proposed a model for wetland treatment of iron and aluminum in acid mine drainage, but its utility has apparently not been tested for aluminum (Mitsch and Wise, 1998). Regarding aluminum, Younger *et al.* (2002) conclude that much remains to be investigated in detail.

## Example Treatment Wetlands for Aluminum Removal

There are but few data sets on the removal of aluminum from types of incoming water other than acid mine drainage (Table 11.20). Bachand and Richardson (1999) conducted field mesocosm studies on aluminum dosing to foster phosphorus removal in treatment wetlands in south Florida. The "pin flocs" generated did not settle effectively. Another FWS wetland study tested the concept of phosphorus removal via aluminum dosing of agricultural runoff, but with coagulation before sending the

## TABLE 11.20
## Examples of Aluminum Removals in Treatment Wetlands

| System | Water Treated | Concentration | | | Mass Rates | | | | Reference |
|---|---|---|---|---|---|---|---|---|---|
| | | In (µg/L) | Out (µg/L) | Reduction (%) | In (kg/ha·yr) | Out (kg/ha·yr) | Removal (kg/ha·yr) | Removal (%) | |
| **Free-Water Surface** | | | | | | | | | |
| South Florida STC Control | Agricultural runoff | 12 | 13 | –8 | 4 | 4 | 0 | –8 | CH2M Hill (2001) |
| South Florida NTC Control | Agricultural runoff | 18 | 14 | 22 | 6 | 5 | 1 | 22 | CH2M Hill (2001) |
| Musselwhite, Ontario | Metal mine water | 34 | 4 | 89 | 50 | 6 | 44 | 89 | Bishay and Kadlec (2005) |
| Sanford, Florida | Urban stormwater | 176 | — | — | 22 | 8 | 14 | 63 | Harper *et al.* (1986) |
| Albright, Pennsylvania | Coal combustion leachate | 628 | 89 | 86 | 166 | 24 | 142 | 86 | Hoover *et al.* (1998) |
| Springdale, Pennsylvania | Coal combustion leachate | 891 | 26 | 71 | 319 | 93 | 226 | 71 | Rightnour and Hoover (1998) |
| Arvida, Quebec | Aluminum plant | 1,050 | 480 | 54 | 183 | 75 | 109 | 59 | Goulet *et al.* (2005) |
| Russellville, Kentucky | Aluminum plant | 1,500 | 100 | 93 | 62 | 3 | 59 | 96 | Rowe and Abdel-Magid (1995) |
| Kampala, Uganda | Water treatment backwash | 1,540 | 170 | 89 | 64 | 2 | 62 | 97 | Kaggwa *et al.* (2001) |
| Laterriere, Quebec | Aluminum plant | 1,660 | 550 | 67 | 292 | 85 | 207 | 71 | Goulet *et al.* (2005) |
| South Florida STC Dosed | Al-Dosed agricultural runoff | 18,834 | 19 | 100 | 6,187 | 6 | 6,181 | 100 | CH2M Hill (2001) |
| South Florida NTC Dosed | Al-Dosed agricultural runoff | 31,536 | 51 | 100 | 10,360 | 17 | 10,343 | 100 | CH2M Hill (2001) |
| Friendship Hill, Pennsylvania | Acid mine water | 50,000 | 35,000 | 30 | — | — | — | — | Flanagan *et al.* (1994) |
| Lick Run, Ohio | Acid mine water | 80,000 | 48,700 | 39 | 13,802 | 7,344 | 6,459 | 47 | Mitsch and Wise (1998) |
| Australia | Power station | 11,200 | 11,900 | –6 | 2,725 | 2,986 | 0 | 0 | Jensen *et al.* (2006) |
| **Subsurface Flow** | | | | | | | | | |
| Nucice, Czech Republic | Municipal wastewater | 451 | <40 | 91 | 44 | 4 | 41 | 91 | Vymazal and Krása (2005) |
| Mořina, Czech Republic | Municipal wastewater | 470 | 45 | 90 | 44 | 4 | 40 | 90 | Unpublished data |
| Břehov, Czech Republic | Municipal wastewater | 1,290 | 605 | 53 | 840 | 394 | 446 | 53 | Unpublished data |
| Slavošovice, Czech Republic | Municipal wastewater | 398 | 113 | 72 | 39 | 11 | 28 | 72 | Unpublished data |
| Greenville, Kentucky | Acid mine water | 26,500 | 22,100 | 17 | 915 | 763 | 152 | 17 | Wieder (1992) |
| Greenville, Kentucky | Acid mine water | 26,500 | 30,800 | –16 | 915 | 1,063 | –148 | –16 | Wieder (1992) |
| Greenville, Kentucky | Acid mine water | 26,500 | 19,300 | 27 | 915 | 666 | 249 | 27 | Wieder (1992) |
| Greenville, Kentucky | Acid mine water | 26,500 | 10,600 | 60 | 915 | 366 | 549 | 60 | Wieder (1992) |
| Greenville, Kentucky | Acid mine water | 26,500 | 11,100 | 58 | 915 | 383 | 532 | 58 | Wieder (1992) |
| Keyser's Ridge, Maryland | Acid mine water | 49,600 | 9,020 | 82 | — | — | — | — | Wieder *et al.* (1988) |

water to FWS wetlands (CH2M Hill, 2001a). Aluminum was passed to the wetlands in varying amounts, and accumulated in the sediments. Good removal from the water was observed for the year-plus duration of the study, but a moving sediment front appeared to have developed. Undosed control wetlands had very low levels of aluminum both entering and leaving. Water treatment alum sludge entered a wetland in Uganda, and was effectively removed to the sediments (Kaggwa *et al.*, 2001). Sediment accretion was selectively higher in the inlet region of the wetland.

The aluminum industry employs FWS treatment wetlands in Kentucky to treat a mixed effluent from a metal processing plant (Rowe and Abdel-Magid, 1995). Over 90% of the aluminum is removed, from an entering level of 1.5 mg/L. Metal mine water was stripped of small quantities of aluminum in a FWS wetland in Ontario (Bishay and Kadlec, 2005), with reduction from 34 to 4 μg/L. About two thirds of the aluminum was removed from urban stormwater by a FWS in Florida (Harper *et al.*, 1986). Over 90% of the aluminum in a municipal wastewater was removed in a HSSF wetland in the Czech Republic (Vymazal and Krása, 2005).

Another coal industry wastewater is the leachate generated from combustion products waste piles. These waters have much less aluminum than acid mine drainage (0.6–0.9 mg/L), and only slightly acidic pH. Studies supported by the Electric Power Research Institute (EPRI) in Pennsylvania found that 70–90% of aluminum was removed in FWS wetlands (Hoover *et al.*, 1998; Rightnour and Hoover, 1998).

Based on this limited data, it appears that aluminum typically is removed from surface water passing through wetlands.

### Manganese

Manganese is an essential element that is chemically similar to iron in its behavior in surface waters. Manganese is vital to plant photosynthesis and is used as an enzyme cofactor for respiration and nitrogen metabolism by plants and animals. Although manganese is toxic to some organisms at elevated concentrations, this situation occurs infrequently, typically with mining wastes. Manganese concentrations greater than 2 mg/L were found to be toxic to algae in laboratory experiments (Goldman and Horne, 1983). Manganese is not observed to bioconcentrate in the wetland food chain.

Manganese may exist in oxidation states ranging from −3 to +7, but the manganous (+2) and manganic (+4) forms are the most important in aqueous systems (Wetzel, 1983). At low redox potentials and low pH the predominant form is Mn(II) (Figure 11.13). The mineral forms are pyrolusite ($MnO_2$), hausmanite ($Mn_3O_4$), rhodochrosite ($MnCO_3$), and manganite (MnOOH). Manganous manganese can form soluble complexes with bicarbonate, sulfate, and organic compounds. Under reducing conditions, manganese forms insoluble complexes with carbonate, sulfide, and hydroxide.

#### Wetland Processing and Storage of Manganese

In an oxygenated environment, manganese is primarily removed from solution by oxidation and hydrolysis (Sikora *et al.*, 2000; Younger *et al.*, 2002):

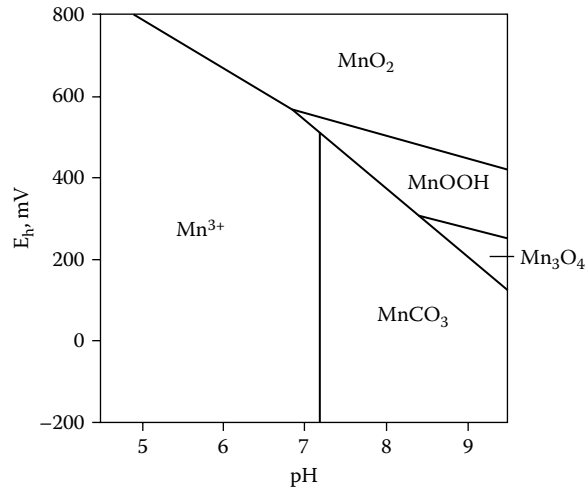



**FIGURE 11.13** Approximate distribution of manganese species according to $E_h$ and pH. (Adapted from Wetzel (1983) *Limnology*, Second Edition, Academic Press, Philadelphia; and Sikora *et al.* (2000) *Water Environment Research* 72(5): 536–544. Reprinted with permission.)

$$Mn^{2+} + H_2O + \frac{1}{2}O_2 \rightarrow MnOOH_{(s)} + 2H^+ \quad (11.31)$$

$$Mn^{2+} + \frac{1}{2}O_2 + 2H^+ \rightarrow Mn^{4+} + H_2O \quad (11.32)$$

$$Mn^{4+} + 2H_2O \rightarrow MnO_{2(s)} + 4H^+ \quad (11.33)$$

Abiotic oxidation of Mn(II) is very slow, and the process is generally considered to be biologically mediated, by bacteria, fungi, and algae. Carbonate may be an intermediate, but likely as substituted dolomite $(Ca,Mg,Mn)CO_3$ rather than the mineral rhodocrosite (Younger *et al.*, 2002). Because manganese is soluble at acidic pH, there is not a possibility for manganese precipitation in acidic waters. As a result, most of the removed manganese is extractable by weak acid (e.g., 85%; Wildeman *et al.*, 1993b).

The oxidation of manganese may be inhibited in the presence of large amounts of iron, because iron exerts a preferential claim on available oxygen (Hedin and Nairn, 1993):

$$MnO_2 + 2Fe^{2+} + 2H_2O \rightarrow 2FeOOH + Mn^{2+} + 2H^+ \quad (11.34)$$

However, dissolved Mn(II) can be removed by adsorption to the iron hydroxide floc.

Under anaerobic conditions, manganese is typically quite soluble. Manganese sulfide (mineral form alabandite) is stable only at very high pH and high concentrations of Mn(II). However, the carbonate has been postulated for anaerobic removal in some wetland systems ($MnCO_3$), but pH must be above neutral for this process (Figure 11.13) (Wildeman *et al.*, 1993b).

**TABLE 11.21**
**Manganese Content of Top Sediments in a Variety of Wetlands**

| Location | Notes | Water Source | Manganese (μg/g) | Reference |
|---|---|---|---|---|
| Germany | Lake Schöhsee | Natural | 1,600 | Wetzel (1975) |
| Panel, Ontario | Cattail marsh | Urban stormwater | 35 | Goulet and Pick (2001) |
| Monahan, Ontario | Cattail marsh | Urban stormwater | 50 | Goulet and Pick (2001) |
| Falconbridge, Ontario | Cattail marsh | Acid mine drainage | 15 | Goulet and Pick (2001) |
| Riverwalk, Ontario | Cattail marsh | Tailings leachate | 55 | Goulet and Pick (2001) |
| West Page Swamp, Idaho | Cattail, Arrowhead | Tailings leachate | 12,500 | DeVolder *et al.* (2003) |
| Widows Creek, Alabama | Cattail, Juncus | Tailings leachate | 200–400 | Ye *et al.* (2001a,b) |
| Show Low, Arizona | Pintail and telephone | Municipal | 521 | NADB database (1998) |
| Tres Rios, Arizona | Four wetlands | Municipal | 314 | NADB database (1998) |
| Champion Paper, Florida | Pilot | Pulp and paper | 285 | NADB database (1998) |
| Sacramento, California | Pilot | Secondary | 206–415 | Nolte and Associates (1998b) |
| Leadville, Colorado | *Salix* and *Carex* | Mine water | 9,500–1,790 | August *et al.* (2002) |
| Cataldo, Idaho | *Typha* and *Scirpus* | Mine water | 1,600 | Hansel *et al.* (2002) |
| Monroe County, New York | *Phragmites* FWS | Landfill leachate | 113 | Eckhardt *et al.* (1999) |
| Monroe County, New York | *Phragmites* FWS | Landfill leachate | 303 | Eckhardt *et al.* (1999) |
| Keyser's Ridge, Maryland | Peat bed | Highway runoff | 236 | Wieder *et al.* (1988) |
| Lick Run, Ohio | Mushroom compost | Acid mine | 395 | Mitsch and Wise (1998) |
| Poland | 11 lakes | Brown coal pits | 3–158 | Samecka-Cymerman and Kempers (2001) |

A Freundlich isotherm was fit to sorption data for river gravel and limestone substrates (Sikora *et al.*, 2000). They reported $n = 0.43$ and $K_F = 22.6$ for river gravel, and $n = 0.43$ and $K_F = 6.6$ (see Equation 11.13).

Manganese is found in wetland plants, algae, and sediments. Concentrations in sediments exposed to mine waters may be very high, up to 10,000 μg/g. For municipal wastewater treatment wetlands, sediment manganese is typically 200–500 μg/g (Table 11.21). Manganese concentrations in plant tissues are of the same order of magnitude, and aboveground values do not differ much from belowground values (Table 11.22).

The general conception is that in the aerobic, surface waters of a wetland, oxidized forms are abundant, Mn(IV), while in the anaerobic soils and sediments, reduced forms prevail, Mn(II) (La Force *et al.*, 2002). As a consequence, both manganese forms in sediments and manganese cycling are driven by redox patterns. For example, at the Cataldo, Idaho, mine drainage wetlands, winter represented aerobic conditions, while spring had deeper waters and anaerobic conditions in the sediments (La Force *et al.*, 2002). As a result, oxides were 34% of the total manganese in winter, but only 3% in spring. In general, the ratio of oxidized to reduced manganese species was 1:1 in spring and 1:8 in winter.

The storage of manganese in wetlands entails little or no risk for waters other than acid mine drainage, because sediment concentration standards are typically quite high (see Table 11.21). It is possible to estimate the sediment concentrations created by sustainable removals of manganese, in terms of the dilution of the manganese storage, by the accretion of new wetland solids. For example, consider 200 dry g/m$^2$·yr of new sediments, originating from 1,000 dry g/m$^2$·yr of biomass production. If the sediment concentration is to be kept below some limiting level, such as the 40,000 μg/g severe effects level of the Ontario guideline, for that Ontario guideline, deposition of 8 g Mn/m$^2$·yr could be sustainably tolerated. Meeting sediment standards on a sustainable basis is easily achieved for all concentrations.

## Performance of Wetlands for Manganese Removal

Manganese is typically removed in FWS wetlands (Table 11.23). For numerous systems, with inlet manganese ranging from 0.1–38,000 μg/L, the median concentration reduction is 54%. There is an increasing wetland exit concentration in response to increases in wetland manganese loading (Figure 11.14). Outlet concentrations are below 1.0 mg/L for wetlands receiving leachates and other waters, but coal mine drainage wetlands are much more heavily loaded, and can have exit concentrations up to 20 mg/L.

Horizontal subsurface constructed wetlands, where filtration beds are usually anoxic or anaerobic, may release manganese over the period of operation. This is due to dissolution of manganese oxyhydroxides precipitates as a consequence of low redox potential. If redox conditions become very low and sulfate is present, manganous ions may precipitate with hydrogen sulfide from sulfate reduction to form insoluble sulfides. Vymazal and Krása (2003) reported a substantial reduction of Mn in a HSSF constructed wetland in the Czech Republic; average inflow concentration of 278 μg/L was reduced to 53 μg/L. However, results from three other HSSF systems in the Czech Republic exhibit substantial Mn release (Table 11.23). Vertical flow constructed wetlands due to higher oxygenation of the bed exhibit good removal of manganese (Table 11.23).

There are two popular methods of interpreting performance data for metal removal in wetlands: the areal load removal and the first-order removal model. These are both explored in detail for iron and manganese in a 35-wetland data set by Tarutis

**TABLE 11.22**
**Manganese Content of Above- and Belowground Plant Parts in a Variety of Wetlands**

| Location | Wetland | Plant | Water (µg/L) | Above (µg/g) | Below (µg/g) | Reference |
|---|---|---|---|---|---|---|
| TVA Mussel Shoals, Alabama | SSF CW | *Scirpus acutus* | 100 | 19 | 23 | Behrends *et al.* (1996) |
| TVA Mussel Shoals, Alabama | SSF CW | *Scirpus atovirens* | 100 | 10 | 17 | Behrends *et al.* (1996) |
| TVA Mussel Shoals, Alabama | SSF CW | *Scirpus cyperinus* | 100 | 11 | 14 | Behrends *et al.* (1996) |
| TVA Mussel Shoals, Alabama | SSF CW | *Phalaris arundinacea* | 100 | 20 | 48 | Behrends *et al.* (1996) |
| TVA Mussel Shoals, Alabama | SSF CW | *Phragmites australis* | 100 | 28 | 62 | Behrends *et al.* (1996) |
| TVA Mussel Shoals, Alabama | SSF CW | *Typha* spp. | 100 | 12 | 38 | Behrends *et al.* (1996) |
| Widows Creek, Alabama | FWS CW | *Typha latifolia* | 7,600 | 1,097 | 200 | Ye *et al.* (2001a, b) |
| Coeur d'Alene, Idaho | FWS Nat | *Typha latifolia* | — | 1,300 | — | DeVolder *et al.* (2003) |
| Wisconsin | FWS Nat | *Typha latifolia* | — | 243 | 70 | Vymazal (1995) |
| Czech Republic | SSF CW | *Typha angustiolia* | — | 642 | — | Vymazal (1995) |
| Poland | FWS Nat | *Typha angustifolia* | 10 | 59 | — | Samecka-Cymerman and Kempers (2001) |
| Nucice, Czech Republic | SSF CW | *Phragmites australis* | 278 | 450 | — | Vymazal and Krása (2005) |
| Poland | FWS Nat | *Phragmites australis* | 502 | 122 | — | Samecka-Cymerman and Kempers (2001) |
| Nucice, Czech Republic | SSF CW | *Phalaris arundinacaea* | 278 | 41 | — | Vymazal and Krása (2005) |
| Poland | FWS Nat | *Phalaris arundinacea* | 898 | 619 | — | Samecka-Cymerman and Kempers (2001) |
| Germany | SSF CW | *Glyceria maxima* | — | 586 | — | Vymazal (1995) |
| Germany | SSF CW | *Scirpus lacustris* | — | 1,200 | — | Vymazal (1995) |
| Poland | FWS Nat | *Scirpus lacustris* | 10 | 67 | — | Samecka-Cymerman and Kempers (2001) |
| Widows Creek, Alabama | FWS CW | *Juncus effusus* | 7,600 | 312 | 117 | Ye *et al.* (2001a, b) |
| Poland | FWS Nat | *Juncus effusus* | 832 | 667 | — | Samecka-Cymerman and Kempers (2001) |
| Coeur d'Alene, Idaho | FWS Nat | *Sagittaria latifolia* | — | 830 | — | DeVolder *et al.* (2003) |
| Bielkowo, Poland | SSF CW | *Glyceria maxima* | — | 241 | 545 | Obarska-Pempkowiak *et al.* (2005) |
| Bielkowo, Poland | SSF CW | *Typha latifolia* | — | 540 | — | Obarska-Pempkowiak *et al.* (2005) |
| Bielkowo, Poland | SSF CW | *Phragmites australis* | — | 158 | — | Obarska-Pempkowiak *et al.* (2005) |
| Chaco-Pampa, Argentina | FWS FAP | *Pistia stratiotes* | 1,000 | 1,319 | — | Miretsky *et al.* (2004) |
| Chaco-Pampa, Argentina | FWS FAP | *Spirodela intermedia* | 1,000 | 5,038 | — | Miretsky *et al.* (2004) |
| Cataldo, Idaho | FWS Nat | *Typha latifolia* | 135 | 3,400 | 790 | Hansel *et al.* (2002) |
| Cataldo, Idaho | FWS Nat | *Phalaris arundinacea* | 135 | 1,200 | 1,500 | Hansel *et al.* (2002) |
| Cataldo, Idaho | FWS Nat | *Scirpus microcarpus* | 135 | 1,800 | 1,900 | Hansel *et al.* (2002) |
| Cataldo, Idaho | FWS Nat | *Equisetum arvense* | 135 | 1,500 | 1,000 | Hansel *et al.* (2002) |
| Monroe County, New York | FWS CW | *Phragmites australis* | — | 265 | 567 | Eckhardt *et al.* (1999) |

*Note:* FWS = free water surface; SSF = subsurface flow; CW = constructed wetland; Nat = natural wetland; FAP = floating aquatic plants.

*et al.* (1999). The former presumes a fixed removal per square meter of wetland, regardless of inlet concentrations and flows, and is basically a zero-order model. The latter separately includes the effects inlet concentration and flow rate.

A problem of the zero-order model is that it cannot be transferred from high loadings to low loadings. The recommended design removal rates are low: 0.5–1.0 g/m$^2$·d (Hedin *et al.*, 1994); 0.5 g/m$^2$·d (PIRAMID Consortium, 2003a). However, this removal far exceeds the inlet loading to nonmine water wetlands, which is only half that value at a maximum (Figure 11.14), and removal is not complete for those lightly loaded systems.

Miretsky *et al.* (2004) determined first-order volumetric rate constants for batch floating plant mesocosms. For initial concentrations of 1–4 mg/L, 87–98% of the manganese was removed in less than one day for *Pistia stratiotes*. Resulting values of $k_V$ were 5.8 h$^{-1}$ for *Pistia stratiotes* and 28.9 h$^{-1}$ for *Spirodela intermedia*. Sikora *et al.* (2000) also found very high plug flow rate constants for SSF systems, with 10–60 m/yr in river gravel wetlands and 100–600 m/yr in limestone wetlands. Tarutis *et al.* (1999) took data at 35 FWS coal mine drainage wetlands over a one-year period. Plug flow rate constants averaged 21 m/yr.

Plants are a minor repository for removed manganese. The Ye *et al.* (2001a) Widows Creek, Alabama, study determined that *Typha latifolia* and *Juncus effusus* contained just 0.95% of the annual amount of manganese entering the system. Vymazal and Krása (2005) found 78% of the added manganese in sediments and belowground plant parts in a HSSF wetland treating domestic wastewater, and 1.73% in aboveground plant parts. The majority of removed manganese is therefore associated with wetland sediments, in sorbed or chemically precipitated forms. Long-term sustainable removal requires continuing maintenance of oxidizing conditions.

## Example Treatment Wetlands for Manganese Removal

### *Quaking Houses, County Durham, United Kingdom* (*Younger et al., 2002*)

Leachate from an abandoned colliery, containing up to 15 mg/L manganese and 30 mg/L of iron, was discharging into

**TABLE 11.23**
**Removal of Manganese in Constructed Wetlands**

| System | Location | Water Source | Inlet (mg/L) | Outlet (mg/L) | Reduction (%) | Removal ($g/m^2 \cdot yr$) | Reference |
|---|---|---|---|---|---|---|---|
| **FWS** | | | | | | | |
| 2 systems | United Kingdom | Coal mine water | 19.0 | 16.0 | 15 | — | Younger (2000) |
| 35 systems | Western Pennsylvania | Coal mine water | 10.0 | 8.4 | 16 | 266 | Tarutis *et al.* (1999) |
| 10 systems | TVA (Eastern United States) | Coal acid mine | 9.6 | 5.0 | 33 | 153 | Brodie (1990) |
| 124 systems | Eastern United States | Coal acid mine | 37.7 | 24.0 | 34 | — | Wieder (1989) |
| Albright | Pennsylvania | Coal mine water | 8.5 | 1.1 | 87 | 197 | Hoover *et al.* (1998) |
| Springdale | Pennsylvania | Coal mine water | 2.7 | 0.2 | 92 | 90 | Hoover *et al.* (1998) |
| Keyser's Ridge | Maryland | Highway runoff | 29.9 | 9.3 | 69 | — | Wieder *et al.* (1988) |
| Clear Creek | Colorado | Metal mine water | 29.8 | 24.9 | 16 | — | Morea *et al.* (1989) |
| Musselwhite | Ontario | Metal mine water | 0.103 | 0.04 | 61 | 11.96 | Bishay and Kadlec (2005) |
| 6 Systems | Champion, Florida | Pulp and paper | 0.600 | 0.318 | 47 | 5 | NADB database (1998) |
| Escambia County | Florida | Leachate | 0.360 | 0.037 | 90 | 1.46 | DeBusk (1997) |
| New Hanover County | North Carolina | Leachate | 0.208 | 0.026 | 88 | 0.1 | Unpublished data |
| Isanti-Chisago | Minnesota | Leachate | 0.262 | 0.257 | 2 | 0.2 | Kadlec (2003) |
| Norco | Louisiana | Refinery | 1.208 | 0.100 | 92 | 121 | Hawkins *et al.* (1997) |
| Estevan | Saskatchewan | Municipal lagoon | 0.113 | 0.175 | 55 | -0.66 | Unpublished data |
| Brookhaven | New York | Domestic | 0.210 | 0.120 | 43 | 24.3 | Hendrey *et al.* (1979) |
| Columbia | Missouri | Municipal | 0.435 | 0.549 | –26 | 24.3 | Unpublished data |
| **Median** | | | | | **47** | **12** | |
| **HSSF** | | | | | | | |
| Nučice | Czech Republic | Municipal | 0.278 | 0.053 | 81 | 2.2 | Vymazal and Krása (2003) |
| Mořina | Czech Republic | Municipal | 0.082 | 0.215 | –162 | –1.23 | Unpublished data |
| Břehov | Czech Republic | Municipal | 0.143 | 0.290 | –103 | –3.6 | Unpublished data |
| Slavošovice | Czech Republic | Municipal | 0.147 | 0.041 | 72 | 1.0 | Unpublished data |
| **VF** | | | | | | | |
| Wapserveen 1 | The Netherlands | Municipal + dairy | 0.22 | 0.02 | 91 | 0.11 | Unpublished data |
| Wapserveen 2 | The Netherlands | Municipal | 0.40 | 0.25 | 38 | 0.14 | Unpublished data |

a stream (Stanley Burn), where it killed all aquatic life in its path. A 45-$m^2$ pilot wetland was found to be effective, comprised of a shallow pond with a substrate of stable waste (manure and straw). Subsequently, a full-scale, four-cell FWS series wetland was implemented in 1997. The first two cells, totaling 440 $m^2$, were the heart of the system, with two follow-on cells regarded as partly cosmetic. The two working cells contained baffles and islands to prevent short-circuiting. The full-scale system substrate was a mixture of municipal compost and manure. The average hydraulic loading was about 20 cm/d. The reduction in manganese was 26.4% over the first 27 months of operation, from 4.4 to 3.2 mg/L. The areal removal rate was 0.26 $g/m^2 \cdot d$ (950 kg/ha·yr), and the first-order areal removal rate constant was $k$ = 22 m/yr (0.061 m/d).

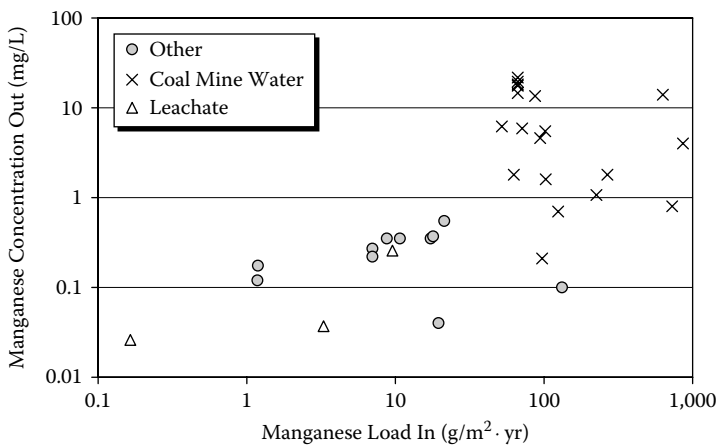



**FIGURE 11.14** Outlet manganese concentrations from treatment wetlands at various loading rates.

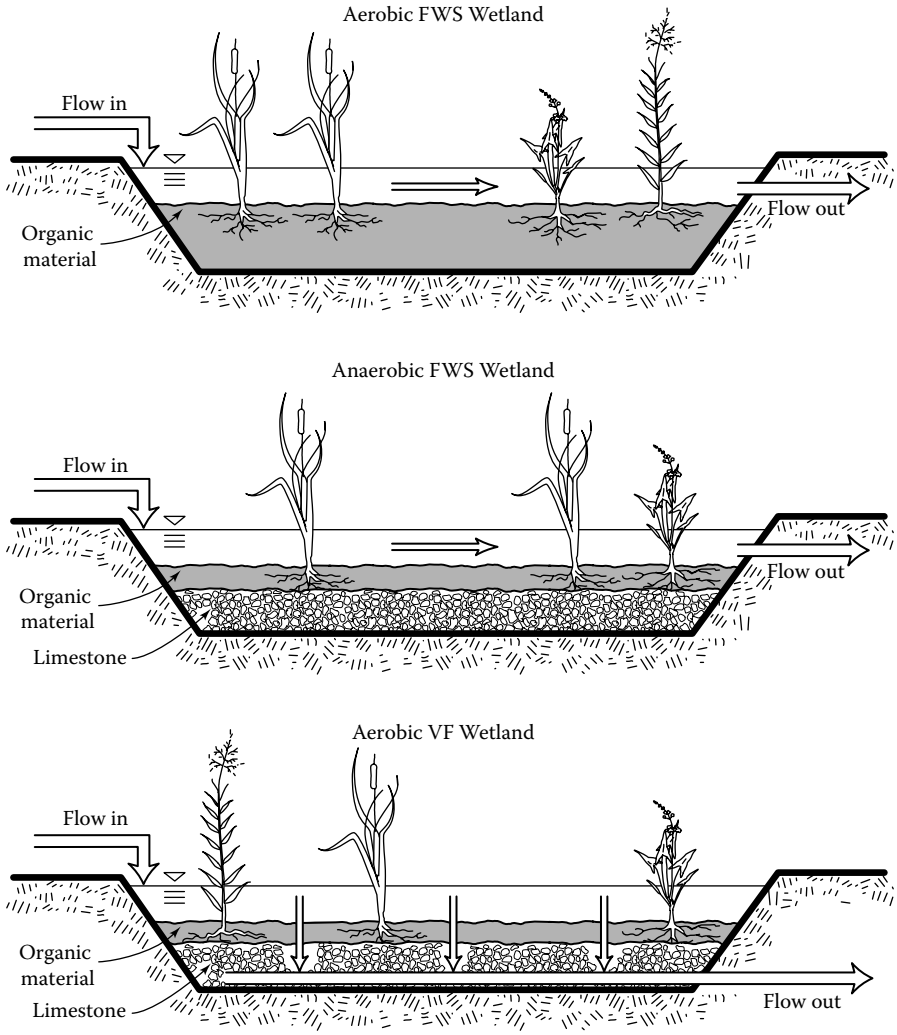



**FIGURE 11.15** Three types of constructed wetlands used in acid mine drainage treatment.

*#40 Gowen, Gaines Creek Watershed, Oklahoma* (*Nairn, 2003*)

Acid mine drainage, in the form of an artesian flow, was adding unacceptable quantities of acidity, iron, and manganese to Gaines Creek. In 1998, the University of Oklahoma constructed a four-cell treatment wetland system to demonstrate the ability of a passive system to treat this discharge. The first and third cells were vertical flow, with a layer of compost above a limestone drainage layer (see Figure 11.15). The second and fourth cells were horizontal aerobic units, and all were planted with *Typha latifolia*. A flow of 20 L/min (29 $m^3$/d) was directed to the wetlands, each of which had an area of 185 $m^2$, resulting in a hydraulic loading of about 4 cm/d. The incoming water had pH = 3.4, Fe = 250 mg/L, Al = 36 mg/L, and Mn = 14 mg/L. At the system outlet, over the 8-month study period, the pH was increased to 7.7, and Fe = 0.77 mg/L, Al = 0.05 mg/L, and Mn = 5.8 mg/L. The manganese load removed was 3,796 kg/ha·yr (1.04 g/$m^2$·d). Observations during water quality sampling events indicated considerable wildlife use of the treatment wetlands. Several species of amphibians (e.g., bullfrogs, leopard frogs, and salamanders), reptiles (e.g., snapping turtles, garter snakes), birds (e.g., red winged blackbirds, killdeer, great blue herons), and mammals (e.g., moles, voles, coyotes) used the site. Biological assessments in the summer of 2000 indicated healthy populations of fish and macroinvertebrates in three of the four cells. Macroinvertebrate community structure indicates a trend from tolerant to less tolerant species with flow through the wetland system. In the final FWS cell, 314 bluegill fingerlings and 7 adults were seined (Nairn, 2003).

## 11.7 HEAVY METALS

### Copper

Copper is an essential micronutrient for plants and animals because it is used for protein synthesis and in blood pigments. Plants and animals require minimal amounts of this element, and deficiencies in nature are rare (Wetzel, 1983). The average copper concentration in the world's lakes and rivers is about 10 µg/L.

Typically, copper is present in surface waters as chelated compounds of Cu(II). The ratio of free ionic to total dissolved

copper is often less than 1% and decreases with increasing organic loads and pH above neutrality (Morel and Hering, 1993). When chelated with organic compounds, copper may remain relatively soluble. Depending upon the nature of the complexing ligand, the soluble complex may not be bioavailable.

U.S. EPA (2002a) sets criteria for freshwater concentrations of copper according to hardness. At hardness of 100 mg/L, the maximum concentration allowable is 13 µg/L, and the continuous concentration allowable is 9 µg/L, as dissolved copper. U.S. EPA (2002a) provides formulae for the limits at other hardnesses, with soft waters having lower criteria. The maximum level for human ingestion is 0.61 mg/L.

Sediment standards for copper have been set by the Province of Ontario, and have been adopted in some of the states in the United States. These set a lowest effect level of 16 µg/g, and a severe effects level of 110 µg/g (Persaud *et al.*, 1993; OMOEE, 1994). OMOEE (1994) has also identified the level for presettlement deposits as 25 µg/g in Great Lakes sediments. Wetzel (1983) reports sediments in Lake Schöhsee, Germany, were 95 µg/g, at a water concentration of 1.0 µg/L.

Copper is a biocide that is commonly used to control algae and other organisms, such as schistosomes (swimmer's itch), which require snails and waterfowl as intermediate hosts. Copper has relatively low toxicity to benthic macroinvertebrates and to fish, which may be unaffected to concentrations as high as 500 µg/L. The sensitivity of algae and the tolerance of fish and benthos to copper have resulted in the widespread use of copper sulfate as an algaecide and molluscicide in lakes.

## Wetland Processing and Storage of Copper

The storage of copper in wetlands entails a risk of creating potentially impaired sediments. It is possible to estimate the sediment concentrations created by sustainable removals of copper, in terms of the dilution of the copper storage by the accretion of new wetland solids. This is a mass balance on the top layer of deposited material (Equation 11.22).

This analysis may be extended to estimate sediment concentrations. For example, consider 200 dry g/m$^2$·yr of new sediments, originating from 1,000 dry g/m$^2$·yr of biomass production. If the sediment concentration is to be kept below some limiting level, such as the 110-µg/g severe effects level of the Ontario guideline, Equation 11.22 can be solved for the allowable removal flux. For that Ontario guideline, deposition of 22 mg Cu/m$^2$·yr could be sustainably tolerated. Meeting rigorous sediment standards on a sustainable basis is not easily achieved for any but trace concentrations, but has been accomplished for systems like Savannah River, South Carolina. But for high concentrations, like those entering White Cedar Bog, Minnesota (620 µg/L), a plume of high sediment concentration (100–3,000 µg/g) developed (Eger *et al.*, 1980).

Copper is found in wetland plants, algae, and sediments. Concentrations in algal tissues are typically higher than in plants and sediments, with concentrations of 100–1,000 µg/g dry weight for a number of species (Vymazal, 1995). In contrast, copper concentrations in plants are much lower, approximately 1–20 µg/g dry mass (Table 11.24). However, if the water concentration is increased to high levels, more copper is accumulated (Qian *et al.*, 1999; Manios *et al.*, 2003). Aboveground plant parts are of lowest concentration, and belowground organs are somewhat higher. Wetland sediments have the highest levels, and are therefore the major repository of stored copper.

### *Copper Removal Processes*

Constructed wetlands have the potential to trap and remove metals contained in wastewater. Long-term removal is expected to occur by accumulation and burial in the plant detritus in a manner similar to the removal of phosphorus.

### *Chemical Precipitation*

Copper forms very insoluble compounds with sulfur, including both cupric ($Cu^{2+}$) and cuprous ($Cu^{+}$) sulfides (Sobolewski, 1999):

$$S^{2-} + Cu^{2+} \rightarrow CuS \tag{11.35}$$

$$S^{2-} + 2Cu^{+} \rightarrow Cu_2S \tag{11.36}$$

Sulfide and bisulfide are formed by SRB in the anaerobic zones of treatment wetlands. If a source of sulfate is present in the incoming water, the wetland can be configured to provide a sustainable supply of sulfide (Sobolewski, 1996). Copper also forms insoluble hydroxides and carbonates (Morel and Hering, 1993). However, they are not of importance in the presence of the more insoluble sulfide.

The removal of copper can also take place through coprecipitation with iron and manganese oxides. However, this mechanism is contingent on considerable supplies of iron, which are typically present only in a fraction of cases. Although iron and manganese oxides are generally excellent scavengers for other metals, they are unstable under anoxic conditions (Knox *et al.*, 2004).

### *Copper Sorption*

Peat contains polar functional groups such as acids and phenolics that are responsible for its CEC, and can be involved in chemical binding (Ho and McKay, 2000). Thus the peat–copper reaction may be represented in two ways:

$$2P^{-} + Cu^{+2} \rightarrow CuP_2 \tag{11.37}$$

$$2HP + Cu^{+2} \rightarrow CuP_2 + 2H^{+} \tag{11.38}$$

where $P$ represents the organic material (peat).

If Equation 11.38, cation exchange, is presumed to represent system performance, then the amount of copper bound is represented by (Kadlec and Keoleian, 1986)

$$K_{eq} = \frac{[CuP_2][H^{+}]^2}{[HP]^2[Cu^{+2}]} \tag{11.39}$$

where the bracket notation denotes molar concentration. This suggests that the partition coefficient ($[CuP_2]/[Cu^{2+}]$ =

## TABLE 11.24
## Copper Content of Wetland Plants and Sediments

| Wetland Plant and Part | Reference | Water (µg/L) | Solid (µg/g) |
|---|---|---|---|
| **Plant Shoots** | | | |
| *Scirpus acutus* | Behrends *et al.* (1996) | 90 | 2.0 |
| *Scirpus acutus* | Nolte and Associates (1998) | 8.6 | 1.3 |
| *Scirpus atovirens* | Behrends *et al.* (1996) | 90 | 2.0 |
| *Scirpus cyperinus* | Behrends *et al.* (1996) | 90 | 1.0 |
| *Scirpus lacustris* | Samecka-Cymerman and Kempers (2001) | 3 | 6.6 |
| *Scirpus* spp. | Karpiscak *et al.* (2001a) | <30 | 2.9 |
| *Juncus effusus* | Samecka-Cymerman and Kempers (2001) | 68 | 15.4 |
| *Juncus* spp. | Chague-Goff (2005) | 11.5 | 2.5 |
| *Phalaris arundinacea* | Behrends *et al.* (1996) | 90 | 1.0 |
| *Phalaris arundinacea* | Vymazal and Krása (2005) | 11 | 1.9 |
| *Phalaris arundinacea* | Vymazal *et al.* (2006) | Sewage | 1.5 |
| *Phalaris arundinacea* | Samecka-Cymerman and Kempers (2001) | 104 | 3.3 |
| *Phragmites australis* | Behrends *et al.* (1996) | 90 | 2.0 |
| *Phragmites australis* | Vymazal and Krása (2005) | 11 | 1.9 |
| *Phragmites australis* | Vymazal *et al.* (2006) | Sewage | 0.74 |
| *Phragmites australis* | Samecka-Cymerman and Kempers (2001) | 67 | 6.8 |
| *Typha* spp. | Behrends *et al.* (1996) | 90 | 3.0 |
| *Typha latifolia* | Obarska-Pempkowiak *et al.* (2005) | Sewage | 1.6 |
| *Typha latifolia* | Maddison *et al.* (2005) | Sewage | 3.2 |
| *Typha domingensis* | Nolte and Associates (1998) | 8.6 | 2.4 |
| *Typha domingensis* | Karpiscak *et al.* (2001a) | <30 | 3.5 |
| *Typha angustifolia* | Samecka-Cymerman and Kempers (2001) | 3 | 6.1 |
| *Glyceria maxima* | Obarska-Pempkowiak *et al.* (2005) | Sewage | 1.5 |
| *Anemopsis californicus* | Karpiscak *et al.* (2001a) | <30 | 4.3 |
| *Lythrum hyssopifolia* | Chague-Goff (2005) | 11.5 | 8.0 |
| *Typha latifolia* | Manios *et al.* (2003) | Sludge | 9–15 |
| *Typha latifolia* | Lan *et al.* (1990) | 53.4 | 5.8 |
| **Plant Roots** | | | |
| *Scirpus acutus* | Behrends *et al.* (1996) | 90 | 5.0 |
| *Scirpus atovirens* | Behrends *et al.* (1996) | 90 | 5.0 |
| *Scirpus cyperinus* | Behrends *et al.* (1996) | 90 | 4.0 |
| *Scirpus* spp. | Karpiscak *et al.* (2001a) | <30 | 16.1 |
| *Juncus* spp. | Chague-Goff (2005) | 11.5 | 4.0 |
| *Phalaris arundinacea* | Behrends *et al.* (1996) | 90 | 5.0 |
| *Phalaris arundinacea* | Vymazal *et al.* (2006) | Sewage | 12.9 |
| *Phragmites australis* | Vymazal *et al.* (2006) | Sewage | 17.3 |
| *Phragmites australis* | Behrends *et al.* (1996) | 90 | 9.0 |
| *Typha* spp. | Behrends *et al.* (1996) | 90 | 8.0 |
| *Typha latifolia* | Obarska-Pempkowiak *et al.* (2005) | Sewage | 5.0 |
| *Typha latifolia*[a] | Maddison *et al.* (2005) | Sewage | 22.3 |
| *Typha domingensis* | Karpiscak *et al.* (2001a) | <30 | 13.8 |
| *Glyceria maxima* | Obarska-Pempkowiak *et al.* (2005) | Sewage | 4.0 |
| *Anemopsis californicus* | Karpiscak *et al.* (2001a) | <30 | 3.7 |
| *Lythrum hyssopifolia* | Chague-Goff (2005) | 11.5 | 10.0 |
| *Typha latifolia* | Manios *et al.* (2003) | Sludge | 40–93 |
| *Typha latifolia* | Lan *et al.* (1990) | 53.4 | 13.9 |
| **Plant Rhizomes** | | | |
| *Scirpus acutus* | Nolte and Associates (1998) | 8.6 | 9.8 |
| *Phragmites australis* | Behrends *et al.* (1996) | 90 | 2.0 |
| *Phragmites australis* | Vymazal *et al.* (2006) | Sewage | 4.5 |
| *Phalaris arundinacea* | Vymazal *et al.* (2006) | Sewage | 4.4 |
| *Typha* spp. | Behrends *et al.* (1996) | 90 | 3.0 |
| *Typha domingensis* | Nolte and Associates (1998) | 8.6 | 8.9 |
| *Typha latifolia* | Zhang *et al.* (1990) | 53.4 | 22.0 |
| **Sediments** | | | |
| System C | Pontier *et al.* (2001) | Urban | 31 |
| System H | Pontier *et al.* (2001) | Urban | 31 |
| Sacramento | Nolte and Associates (1998) | Secondary | 40 |
| 11 lakes, Poland | Samecka-Cymerman and Kempers (2001) | Brown coal pits | 0.2–18.6 |
| Christchurch, New Zealand | Chague-Goff (2005) | Lagoons | 19 |
| Bielkowo, Poland | Obarska-Pempkowiak *et al.* (2005) | Sewage | 56 |
| *Typha latifolia* | Lan *et al.* (1990) | 53.4 | 54 |

*Note:* The corresponding water concentrations are approximate.
[a] Roots and rhizomes.

$C_S/C_L$) should go down markedly with decreasing pH, which is in fact the observation of many investigators, as shown in Kadlec and Keoelian (1986). Because of the exponent of two on hydrogen ion concentration in Equation 11.39, halving the pH decreases the partition coefficient by a factor of 4. The concentration of available exchange sites, [H*P*], is related to the CEC of the peat. Brown *et al.* (2001) found that the copper sorption potential of various peats was linearly correlated with their CEC. Uptake ranged from 10–60 mg/g to 100–200 meq/100 g ($R^2 = 0.90$).

Equation 11.39 postulates that there are a finite number of exchange sites, and hence the uptake of copper should reach a maximum. The Langmuir isotherm provides for such saturation, and fits peat sorption data (Chen *et al.*, 2001; see Equation 11.11).

The maximum capacity for copper is $C_{Smax} = K/a$. Brown *et al.* (2001) found capacities from 10–60 mg/g for 11 Irish peats, whereas Chen *et al.* (2001) found 13 mg/g for a New Zealand peat. Freundlich isotherms are also often used, and fit nearly as well (Kadlec and Rathbun, 1984) (Equation 11.13).

Kadlec and Rathbun (1984) reported $n = 0.4$ and $K_F = 3.4$ at pH = 6.5, and 2.0 at pH = 5.2. Chen *et al.* (2001) reported $n = 0.145$ and $K_F = 5.6$ at $3.1 < pH < 3.6$. Whichever models are used, peats have the ability to store very large quantities of copper. By extrapolation, organic substrates in general have similarly large capacities. According to Equation 11.13, peats can store 5–10 mg/g for water concentrations of 1–10 mg Cu/L and circumneutral pH.

## Performance of Wetlands for Copper Removal

Copper is effectively removed in FWS wetlands (Table 11.25). For 26 systems, with inlet copper ranging from 1 to 7,300 µg/L, the median concentration reduction is 66%. There is a possibility that first-order models of copper removal would be appropriate (Tarutis *et al.*, 1999), but there are no calibrations for existing systems. It is probable that for overland flow wetlands, the rate-limiting step is the transfer of copper from the water to sorption and precipitation sites in the sediments. Traditional mass transfer modeling would utilize a first-order model for that process (see Chapter 6). It is also likely that percentage reductions in Table 11.25 would be greater if the hydraulic efficiencies were higher, e.g., closer to plug flow. The Sinicrope *et al.* (1992) data correspond to plug flow *k*-values from 18 to 47 m/yr for a SSF system. Vymazal and Krása (2003) reported the *k*-values of 17 m/yr and 15.5 m/yr for HSSF systems in Nučice and Mořina, the Czech Republic. However, in the HSSF system at Břehov, copper was washed out of the system. This corresponds well with increase concentrations of iron and manganese at the outflow from this system. The *k*-value for this system was –5.8 m/yr. Kadlec and Srinivasan (1995) report copper uptake in FWS cattail microcosms of $25 < k_{PF} < 120$ m/yr. The Savannah River, South Carolina, results give a first-order areal plug flow removal rate constant $k_{PF} \approx 75$ m/yr. These are consistent with estimates of mass transfer coefficients (Kadlec and Knight, 1996).

Quite complicated models have been written for copper removal. The effort of Lung and Light (1996) was exercised, but not calibrated or verified. The most significant effort was the development and calibration of the CWFATE model for the Sacramento, California, project (Jones & Stokes Associates, 1993; Nolte and Associates, 1998a,b). CWFATE attempts to describe water budgets, copper biomass cycles, and partitioning, but not chemical precipitation.

Plants are a minor repository for removed copper. The Murray-Gulde *et al.* (2005) Savannah River, South Carolina, study determined that *Scirpus californicus* took up 6.76% of copper entering the system. In the Sinicrope *et al.* (1992) study, the removed copper was found mostly in the soil (91%), with smaller amounts in fine roots (8%), and the balance in coarse roots, rhizomes, and shoots. Vymazal and Krása (2005) found 76% of the added copper in sediments and belowground plant parts in an HSSF wetland treating domestic wastewater, and 6.3% in aboveground plant parts.

The majority of removed copper is therefore associated with wetland sediments, in sorbed or chemically precipitated forms. These may be categorized according to sequential extraction schemes (Morea *et al.*, 1989; Sobolewski, 1996). The principal forms are associated with iron and manganese oxides, sulfides, and exchange sites on organics (Figures 11.16 and 11.17). In each case, long-term sustainable removal requires a continuing supply of secondary materials.

### *Anaerobic Wetlands*

Many wetland systems have been constructed using compost or other organic waste to generate an anaerobic environment and provide a source of organic carbon (Skousen, 1997; Younger *et al.*, 2002; Eger and Wagner, 2003) (see Figure 11.15). These wetlands include various flow patterns, including fully flooded VF downflow and shallow FWS overland flow.

Reduction of organic matter over sulfate generates hydrogen sulfide (Equations 11.1 and 11.2), which may react with copper to form insoluble copper sulfides (Equations 11.35 and 11.36). Lifetime estimates based on the total amount of carbon in these systems suggested that the substrate would last for several decades, but data indicate that these predictions were overly optimistic. Eger and Wagner (2003) suggest that the amount of available substrate carbon is about 10% of the theoretical. Decaying wetland plants are a potential carbon source. Bioavailable carbon production in wetlands is about 60 g C/m$^2$·yr (see the denitrification section of Chapter 9). According to Equation 11.2, this would support sulfide production of 80 g S/m$^2$·yr, a factor of 50 less than the typical design rate recommended by Wildeman *et al.* (1993b). The corresponding copper removal rate is 160 g Cu/m$^2$·yr, which is also far less than the design value of 3,650 g Cu/m$^2$·yr tentatively advanced by the PIRAMID Project of the European Union (PIRAMID Consortium, 2003a) for anaerobic wetlands. Results from HSSF wetlands (Table 11.25) indicate generally high removal of copper (81–84%), but in one instance copper was released from the system.

## TABLE 11.25
## Copper Reduction in Treatment Wetlands

| System | Notes | Concentration (µg/L) In | Concentration (µg/L) Out | Concentration (µg/L) Reduction (%) | Mass Rates (kg/ha·yr) In | Mass Rates (kg/ha·yr) Out | Mass Rates (kg/ha·yr) Removal | Mass Rates (kg/ha·yr) Removal (%) | Reference |
|---|---|---|---|---|---|---|---|---|---|
| **FWS** | | | | | | | | | |
| **Mine Water** | | | | | | | | | |
| Albright, Pennsylvania | Coal combustion leachate | 1 | 2 | −100 | 0.26 | 0.53 | −0.26 | −100 | Hoover *et al.* (1998) |
| Woodcutters, NT, Australia | Metal mine water | 1.4 | 0.9 | 36 | — | — | — | — | Noller *et al.* (1994) |
| Springdale, Pennsylvania | Coal combustion leachate | 12 | 13 | –8 | 4.30 | 4.65 | −0.36 | −8 | Hoover *et al.* (1998) |
| Tom's Gully, NT, Australia | Metal mine water | 13 | 1 | 96 | — | — | — | — | Noller *et al.* (1994) |
| Musselwhite, Ontario | Metal mine water | 14 | 3 | 79 | 26.57 | 5.69 | 20.88 | 79 | Bishay and Kadlec (2005) |
| Dunka Mine, Minnesota | Metal mine water | 145 | 15 | 90 | — | — | — | — | Eger *et al.* (1993) |
| Orcopampa, Peru | Metal mine water | 205 | 30 | 85 | — | — | — | — | Unpublished data |
| Wheal Jane, United Kingdom | Tin/zinc mine water | 400 | 200 | 50 | — | — | — | — | Younger (2000) |
| Dunka Mine, Minnesota | Metal mine water | 7,300 | 70 | 99 | — | — | — | — | Eger (1994) |
| **Landfill Leachate** | | | | | | | | | |
| New Hanover County, NC | Landfill leachate | 5 | 3 | 40 | — | — | — | — | Unpublished data |
| Isanti-Chisago, Minnesota | Landfill leachate | 10 | 10 | 0 | — | — | — | — | Kadlec (2003c) |
| Monroe County, New York | Landfill leachate | 30 | 10 | 67 | — | — | — | — | Eckhardt *et al.* (1999) |
| **Urban Stormwater** | | | | | | | | | |
| Hidden River, Florida | Urban stormwater | 4 | 3 | 25 | 0.38 | 0.28 | 0.094 | 25 | Carr and Rushton (1995) |
| Greenwood, Florida | Urban stormwater | 14 | 6 | 57 | 0.246 | 0.101 | 0.145 | 59 | McCann and Olson (1994) |
| Sanford, Florida | Urban stormwater | 20 | — | — | 1.38 | 0.83 | 0.55 | 40 | Harper *et al.* (1986) |
| Norco, Louisiana | Refinery stormwater | 22 | 15 | 33 | — | — | — | — | Hawkins *et al.* (1997) |
| Aiken, South Carolina | Lab stormwater | 30 | 5 | 83 | — | — | — | 82 | Nelson *et al.* (2004) |
| Crestwood, Virginia | Urban stormwater | 33 | 12 | 64 | 0.035 | 0.012 | 0.023 | 66 | Godrej *et al.* (1999) |
| Kildare, Ireland | Highway runoff | 19–95 | 8–39 | 68 | — | — | — | — | Higgins *et al.* (2006b) |
| **Municipal Wastewater** | | | | | | | | | |
| Estevan, Saskatchewan | FWS lagoon | 5 | 3 | 40 | — | — | — | — | Unpublished data |
| Sacramento, California | FWS | 8.62 | 3.57 | 59 | — | — | — | — | Unpublished data |
| Christchurch, New Zealand | FWS | 11.5 | 2.2 | 81 | — | — | — | — | Chague-Goff (2005) |
| Central Illinois | FAP | 57 | 8 | 86 | — | — | — | — | Jin *et al.* (2003) |
| Central Illinois | SSF | 57 | 9 | 84 | — | — | — | — | Jin *et al.* (2003) |
| Central Illinois | FWS | 57 | 11 | 81 | — | — | — | — | Jin *et al.* (2003) |
| San Diego, California | FWS/SSF | 60 | 22 | 63 | 11 | 4 | 6.8 | 63 | Sinicrope *et al.* (1992) |
| San Diego, California | FWS/SSF | 1,200 | 144 | 88 | 268 | 32 | 236 | 88 | Sinicrope *et al.* (1992) |
| **HSSF** | | | | | | | | | |
| Central Illinois | — | 57 | 9 | 84 | — | — | — | — | Jin *et al.* (2003) |
| Nučice, Czech Republic | Municipal | 11.3 | 2.0 | 82 | 1.2 | 0.2 | 1.0 | 82 | Vymazal and Krása (2003) |
| Mořina, Czech Republic | Municipal | 17.3 | 3.3 | 81 | 1.6 | 0.3 | 1.3 | 81 | Unpublished data |
| Břehov, Czech Republic | Municipal | 28.5 | 36.1 | –27 | 7.0 | 8.9 | −1.9 | −27 | Unpublished data |
| **VSSF** | | | | | | | | | |
| Wepserveen, The Netherlands | Municipal + dairy | 37 | 14 | 62 | 0.2 | 0.08 | 0.12 | 62 | Unpublished data |
| Wepserveen, The Netherlands | Municipal | 27 | 13 | 52 | 2.5 | 1.2 | 1.3 | 52 | Unpublished data |
| Nijkerk, The Netherlands | Municipal | 99 | 25 | 75 | 19.9 | 5 | 14.9 | 75 | Unpublished data |
| Bokrijk, Belgium | Municipal | 17 | 6 | 65 | 1.75 | 0.6 | 1.15 | 65 | Unpublished data |
| Zemst-Larbeek, Belgium | Municipal | 26 | 10 | 62 | — | — | — | — | Unpublished data |
| Sint-Niklaas-Heimolen, Belgium | Municipal | 28 | 15 | 46 | — | — | — | — | Unpublished data |

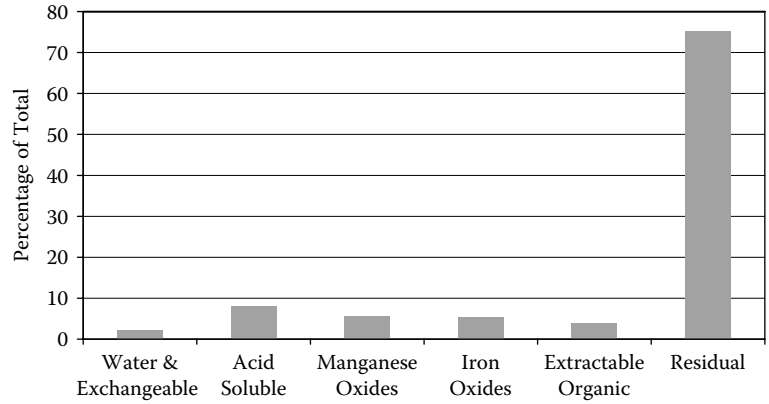



**FIGURE 11.16** Distribution of copper species in the sediments from the Savannah River, South Carolina, FWS treatment wetland site. The residual fraction includes sulfides. Water was at about 15 µg/L. (Data from Knox *et al*. (2004) *Metals retention in constructed wetland sediments*. Report WSRC-MS-2004-00444, U.S. Department of Energy, Oak Ridge, Tennessee.)

### *Aerobic Wetlands*

These systems are FWS wetlands intended to generate an aerobic environment overlying an anaerobic substrate (Skousen, 1997; Younger *et al*., 2002; Eger and Wagner, 2003) (see Figure 11.15). Metal removal in aerobic systems occurs by sorption, complexation, and ion exchange on the substrate. These processes function best at circumneutral pH.

The ability of these reactions to remove metals is limited by the number of reaction sites, which is a function of the volume of substrate that the flow can contact. Preexisting sites will be used up over time, and sustainable removal requires the generation of new sites on newly accreting wetland sediments. This forms a basis for estimating sustainable removal capacity (Eger and Wagner, 2003).

Sites will fill to the capacity dictated by incoming water concentration before substantial breakthrough occurs, because of the nature of the sorption isotherms (Freundlich $n < 1.0$). FWS wetlands typically produce 200 dry g/m$^2$·yr of new sediments. Sustainable sorption would contribute about 66 mg Cu/m$^2$·yr for water at 50 µg/L, and 2,000 mg Cu/m$^2$·yr for water at 10 mg/L, based on Chen *et al*. (2001). These copper removal rates are also far less than the design value of 18 g Cu/ m$^2$·yr tentatively advanced by the PIRAMID Project of the European Union (PIRAMID Consortium, 2003a) for aerobic wetlands.

Due to the high oxygenation of the bed, VF wetlands exhibit good removal of copper, presumably due to coprecipitation with Mn- and Fe-hydroxides and oxyhydroxides. The removal efficiency in 6 VSSF systems in the Netherlands and Belgium (Table 11.25) varied between 46–75%.

## Example Treatment Wetlands for Copper Removal

### *Stormwater and Process Water in Outfall A-01 from the Savannah River*

Discharges from the Technical Center of the U.S. Department of Energy contain unacceptably high levels of copper, typically in the range of 30–50 µg/L. Conventional treatment systems for metal removal (e.g., ion exchange, chemical precipitation, etc.) were estimated to be very expensive for the volume of water that needed to be treated (3,700 m$^3$/d average, 10,500 m$^3$/d peak; and the extremely low concentrations that

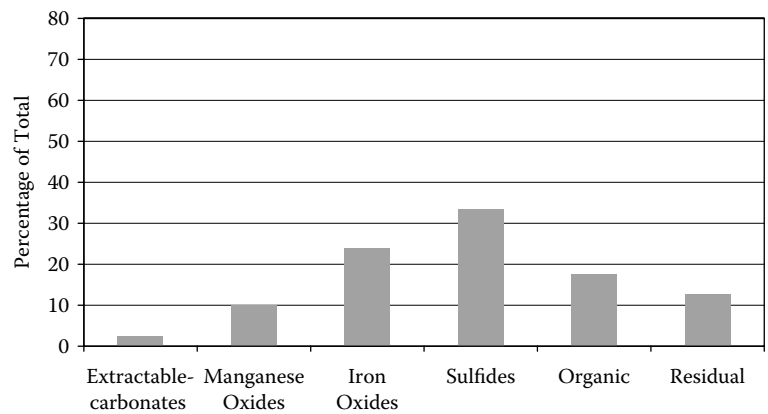



**FIGURE 11.17** Distribution of copper species in the sediments from the Central City, Colorado, FWS treatment wetland site. Water was at about 900 µg/L. (Data from Morea *et al*. (1989) *Assessment of a passive treatment system for acid mine drainage at a Colorado Superfund Site*. Proceedings of the 62nd Annual Conference of the Water Pollution Control Federation, 15–19 October 1989, San Francisco, California.)

must be achieved in the water before it was released to the stream (22 μg/L) (Nelson *et al.*, 2002). Mesocosm testing indicated that peat-based wetlands could easily achieve the desired goals (Gladden *et al.*, 2001). Construction of a 3.24-ha, four-cell FWS facility was carried out during 2000 (Nelson *et al.*, 2002). Removals were acceptable (82%, outlet copper 3–8 μg/L) over the first four years of operation (Nelson *et al.*, 2004).

Murray-Gulde *et al.* (2005) determined that shoots and roots of *Scirpus californicus* took up 0.88% and 5.88%, respectively, of copper entering the Savannah River, South Carolina, system. The balance of the removal (82% [total removal] – 7% [plants] = 75% of the input) can be attributed to storage in the wetland sediments. Knox *et al.* (2004) determined that the majority (80%) of the stored copper in sediments was not readily remobilized (see Figure 11.16).

### *Experimental Peat-Based Wetlands at the Former Bell Copper Mine*

These wetlands at Smithers, British Columbia, Canada (Noranda Minerals, Inc.), removed copper from low strength (0.3–1.0 mg/L, pH 6–8) and high strength (35–50 mg/L, pH 3) copper-contaminated mine drainage (Sobolewski, 1996). Copper species retained in wetland sediments were quantified as a means to assess the long-term potential of these wetlands for treating mine drainage. Previous microbial and chemical analyses had indicated that sulfide was biologically generated in these wetlands, and mineralogical analysis confirmed that copper sulfides were present in wetland sediments. However, sequential leaching of these sediments showed that copper was predominantly in organically bound and oxide-bound phases, with a much smaller proportion present as sulfides. Dosing with high strength water (50 mg Cu/L) indicated a relatively short removal lifetime of about 4 months.

## Nickel

The oxidation state for nickel in surface waters is divalent Ni(II). Under aerobic conditions, divalent nickel may be complexed with hydroxides, sulfates, or carbonates. Under anaerobic conditions, nickel forms sulfide precipitates. Much of the nickel in surface waters is associated with soluble organic ligands (Xue *et al.*, 2001). Depending upon the nature of the complexing ligand, the soluble complex may not be bioavailable. Hutchinson (1975) indicated that the average nickel concentration in surface waters is 5 μg/L. Xue *et al.* (2001) reported total nickel concentration in lakes and rivers in Switzerland as 0.2–1.8 μg/L.

Nickel is sometimes an essential micronutrient for growth, but it also is toxic at modest concentrations. Nickel is not biomagnified in the food chain. U.S. EPA (2002a) sets criteria for freshwater concentrations of nickel according to hardness. At a hardness concentration of 100 mg/L as $CaCO_3$, the maximum allowable nickel concentration is 470 μg/L, and the continuous concentration allowable is 52 μg/L, as dissolved nickel. U.S. EPA (2002a) provides formulae for the limits at other hardnesses, with soft waters having lower criteria. The maximum level for human ingestion is 0.61 mg/L. Sediment standards for nickel have been set by the Province of Ontario, and adopted in some of the states in the United States. These set a lowest effect level of 16 μg/g, and a severe effects level of 75 μg/L (Persaud *et al.*, 1993; OMOEE, 1994). OMOEE (1994) has also identified the level for pre-settlement deposits as 31 μg/g in Great Lakes sediments.

### Wetland Processing and Storage of Nickel

Using Equation 11.22, the analysis may be extended to estimate sediment concentrations. For example, consider 200 dry g/m$^2$·yr of new sediments, originating from 1,000 dry g/m$^2$·yr of biomass production. If the sediment concentration is to be kept below some limiting level, such as the 75 μg/g severe effects level of the Ontario guideline, Equation 11.22 can be solved for the allowable removal flux. For that Ontario guideline, deposition of 22 mg Ni/m$^2$·yr could be sustainably tolerated. Meeting rigorous sediment standards on a sustainable basis is not easily achieved for any but trace concentrations. For high concentrations, like those entering White Cedar Bog, Minnesota (24,500 μg/L), a plume of high sediment concentration (100–3,000 μg/g) developed (Eger *et al.*, 1980).

Wetland plants have trace amounts of nickel in their tissues, typically with higher values for roots and rhizomes than for aboveground plant parts (Table 11.26). Concentrations can become elevated in wetland plant tissues in response to exposure to high water concentrations of nickel (Manios *et al.*, 2003). At nonmine water sites in Minnesota, the average nickel concentration in cattails ranged from 4.9 mg/kg in the leaves to 18.0 mg/kg in the roots (P. Eger, Minnesota DNR, unpublished data). Following exposure to mine drainage, average nickel concentrations in these plants increased to about 14.8–25 mg/kg dry mass in the cattail leaves, and 195–246.6 mg/kg dry mass in the cattail roots. Wetland sediments have the potential to contain large amounts of nickel, in response to chemical precipitate formation and cation exchange with nickel in the water. In most situations, that capacity is not realized, because of the low concentrations in the wetland waters. For instance, background waters at White Cedar Bog in Minnesota were at 3–8 μg/L, and the associated peats had 26–39 μg/g (Eger *et al.*, 1980). Mine leachate entered the bog at 18 mg/L, and peats at that location had up to 3,750 μg/g. Clearly, the new sediments contained nickel that originated from precipitation and partitioning, rather than from the cycling of biomass.

### *Nickel Removal Processes*

Nickel forms very insoluble sulfides (Sobolewski, 1999):

$$S^{2-} + Ni^{2+} \rightarrow NiS \qquad (11.40)$$

Sulfide and bisulfide are formed by SRB in the anaerobic zones of treatment wetlands. If a source of sulfate is present in the incoming water, the wetland can be configured to provide a sustainable supply of sulfide (Sobolewski, 1996).

Peat contains polar functional groups such as acids and phenolics that are responsible for its CEC, and can be involved in chemical binding (Ho and McKay, 2000). There are a

**TABLE 11.26**
**Nickel Content of Treatment Wetland Plants and Sediments**

| Wetland Plant and Part | Reference | Water (µg/L) | Solid (µg/g) |
|---|---|---|---|
| **Plant shoots** | | | |
| *Scirpus acutus* | Nolte and Associates (1998) | 8.6 | 0.7 |
| *Scirpus lacustris* | Vymazal (1995) | W | 1.7 |
| *Scirpus lacustris* | Samecka-Cymerman and Kempers (2001) | 5 | 2.8 |
| *Phalaris arundinacea* | Vymazal and Krása (2005) | 9.5 | 9.9 |
| *Phalaris arundinacea* | Vymazal *et al.* (2006) | W | 3.6 |
| *Phalaris arundinacea* | Samecka-Cymerman and Kempers (2001) | 5 | 4.4 |
| *Phragmites australis* | Samecka-Cymerman and Kempers (2001) | 5 | 3.8 |
| *Phragmites australis* | Vymazal *et al.* (2006) | W | 1.8 |
| *Phragmites australis* | Vymazal and Krása (2005) | 9.5 | 8.4 |
| *Typha domingensis* | Nolte and Associates (1998) | 8.6 | 1.8 |
| *Pontederia cordata* | Mudroch and Capobianco (1978) | <1 | 1.6 |
| *Lythrum salicaria* | Mudroch and Capobianco (1978) | <1 | 3.0 |
| *Typha angustifolia* | Vymazal (1995) | W | 1.9 |
| *Typha angustifolia* | Samecka-Cymerman and Kempers (2001) | 5 | 1.1 |
| *Typha latifolia* | Mudroch and Capobianco (1978) | <1 | 2.1 |
| *Typha latifolia* | Manios *et al.* (2003) | Sludge | 17–28 |
| *Bidens laevis* | Garcia *et al.* (2006) | 50,000 | 845 |
| *Juncus effusus* | Garcia *et al.* (2006) | 50,000 | 19–97 |
| *Juncus effusus* | Samecka-Cymerman and Kempers (2001) | 8.8 | 4.2 |
| *Polygonum hydropiperoides* | Garcia *et al.* (2006) | 50,000 | 220 |
| **Plant Roots** | | | |
| *Pontederia cordata* | Mudroch and Capobianco (1978) | <1 | 9.0 |
| *Lythrum salicaria* | Mudroch and Capobianco (1978) | <1 | 4.0 |
| *Typha latifolia* | Mudroch and Capobianco (1978) | <1 | 12.0 |
| *Typha latifolia* | Manios *et al.* (2003) | Sludge | 30–55 |
| *Phalaris arundinacea* | Vymazal *et al.* (2006) | W | 9.2 |
| *Phragmites australis* | Vymazal *et al.* (2006) | W | 9.0 |
| *Bidens laevis* | Garcia *et al.* (2006) | 50,000 | 1,143 |
| *Juncus effusus* | Garcia *et al.* (2006) | 50,000 | 352 |
| *Polygonum hydropiperoides* | Garcia *et al.* (2006) | 50,000 | 433 |
| **Plant Rhizomes** | | | |
| *Scirpus acutus* | Nolte and Associates (1998) | 8.6 | 7.1 |
| *Typha domingensis* | Nolte and Associates (1998) | 8.6 | 4.6 |
| *Phalaris arundinacea* | Vymazal *et al.* (2006) | W | 0.8 |
| *Phragmites australis* | Vymazal *et al.* (2006) | W | 0.6 |
| **Sediments** | | | |
| Sacramento | Nolte and Associates (1998) | Secondary | 49 |
| St. Clair Marshes | Mudroch and Capobianco (1978) | <1 | 16.4 |
| 11 lakes, Poland | Samecka-Cymerman and Kempers (2001) | 0–55 | 0.2–31 |

*Note:* The corresponding water concentrations are approximate; W = wastewater.

finite number of exchange sites, and hence the uptake of nickel should reach a maximum. The Langmuir isotherm provides for such saturation, and fits peat sorption data (Chen *et al.*, 2001) (see Equation 11.11). The maximum capacity for nickel is $C_{Smax} = K/a$. Chen *et al.* (2001) found 28 mg/g for a New Zealand peat. The corresponding $K$ = 1.3 L/g, and $a$ = 0.046 L/mg.

Freundlich isotherms fit nearly as well (see Equation 11.13). Chen *et al.* (2001) reported $n = 0.268$ and $K_F = 5.3$ at $3.1 <$ pH $< 3.6$. Doig and Liber (2006) measured $n$ = 0.56–0.63, and $K_F$ = 4.7, 8.1, and 13.8 at pH = 6, 7, and 8, respectively.

Whichever models are used, peats have the ability to store very large quantities of nickel. By extrapolation, organic substrates in general have similarly large capacities. According to Equation 11.11, peats can store 1–9 mg/g for water concentrations of 1–10 mg Ni/L and circumneutral pH. This compares well to the suggestion of 10 mg/g as a capacity value by Eger and Lapakko (1989).

## Performance of Wetlands for Nickel Removal

Nickel is removed in wetlands when the incoming amounts are greater than background (Table 11.27). For 23 systems, with inlet nickel ranging from 2–24,000 µg/L, the median concentration reduction is 34%.

There is a possibility that first-order models of nickel removal would be appropriate (Tarutis *et al.*, 1999), but there are no calibrations for existing systems. It is probable that for

## TABLE 11.27
## Nickel Reduction in Treatment Wetlands

| System | Notes | Concentration (µg/L) | | | Mass Rates (kg/ha·yr) | | | | Reference |
|---|---|---|---|---|---|---|---|---|---|
| | | In | Out | Reduction (%) | In | Out | Removal | Removal (%) | |
| **Mine Waters** | | | | | | | | | |
| Musselwhite, Ontario | Metal mine water | 7 | 1 | 82 | 13.3 | 2.4 | 10.9 | 82 | Bishay and Kadlec (2005) |
| Albright, Pennsylvania | Coal ash leachate | 88 | 23 | 74 | 23.3 | 6.1 | 17.2 | 74 | Hoover *et al.* (1998) |
| Springdale, Pennsylvania | Coal ash leachate | 55 | 31 | 43 | 97 | 37 | 60 | 62 | Ye *et al.* (2001a,b) |
| Dunka Mine, Minnesota | Metal mine water | 5,195 | 335 | 94 | 188 | 88 | 100 | 53 | Eger *et al.* (1993) |
| Dunka Mine, Minnesota | Metal mine water | 24,500 | 118 | 100 | — | — | — | — | Eger (1994) |
| Tom's Gully, NT, Australia | Metal mine water | 52 | 5 | 90 | — | — | — | — | Noller *et al.* (1994) |
| **Landfill Leachate** | | | | | | | | | |
| New Hanover County, NC | Landfill leachate | 19 | 4 | 79 | — | — | — | — | Unpublished data |
| Isanti-Chisago, Minnesota | Landfill leachate | 28 | 28 | 0 | — | — | — | — | Kadlec (2003c) |
| Monroe County, New York | Landfill leachate | 65 | 60 | 8 | — | — | — | — | Eckhardt *et al.* (1999) |
| **Urban Stormwater** | | | | | | | | | |
| DUST, California | | — | — | 20 | — | — | — | — | Meiorin (1989) |
| Orlando, Florida | Cattail mesocosm | 32 | 21 | 34 | — | — | 2.7 | — | Kent *et al.* (1997) |
| Orlando, Florida | Pickerelweed mesocosm | 32 | 18.5 | 44 | — | — | 4.5 | — | Kent *et al.* (1997) |
| Orlando, Florida | Urban | 4 | 3 | 25 | — | — | — | — | Schiffer (1989) |
| Sanford, Florida | Urban | 2.7 | — | — | 0.23 | 0.07 | 0.16 | 70 | Harper *et al.* (1986) |
| **Industrial Wastewater** | | | | | | | | | |
| San Tomé, Argentina | Tool factory | 62.5 | 30.8 | 51 | 11.4 | 5.6 | 5.8 | 51 | Maine *et al.* (2006) |
| **Municipal Wastewater** | | | | | | | | | |
| Hayward, California | Secondary | 7.4 | 16.0 | −116 | — | — | — | — | NADB database (1998) |
| Lakeland, Florida | Secondary | 14.8 | 12.0 | 19 | — | — | — | — | NADB database (1998) |
| Central Illinois | FAP | 11.0 | 4.0 | 64 | 5.07 | 1.84 | 3.23 | 64 | Jin *et al.* (2003) |
| Central Illinois | SSF | 11.0 | 16.0 | −45 | 5.07 | 7.38 | −2.31 | −45 | Jin *et al.* (2003) |
| Central Illinois | FWS | 11.0 | 10.0 | 9 | 5.07 | 4.61 | 0.46 | 9 | Jin *et al.* (2003) |
| Nučice, Czech Republic | HSSF | 9.5 | 0.7 | 92 | 0.95 | 0.07 | 0.88 | 92 | Vymazal and Krása (2003) |
| Mořina, Czech Republic | HSSF | 5.4 | 11.3 | −109 | 0.55 | 1.15 | −0.6 | −109 | Unpublished data |
| Břehov, Czech Republic | HSSF | 52 | 65 | −25 | 12.8 | 16 | −3.2 | −25 | Unpublished data |
| Slavošovice, Czech Republic | HSSF | 23 | 25 | −9 | 2.2 | 2.4 | −0.2 | −9 | Unpublished data |
| Wepserveen, The Netherlands | VSSF | 9 | 31 | −244 | 0.05 | 0.17 | −0.12 | −244 | Unpublished data |
| Wepserveen, The Netherlands | VSSF | 11 | 20 | −82 | 1.0 | 1.8 | 0.8 | −82 | Unpublished data |
| Nijkerk, The Netherlands | VSSF | 7.8 | 45 | −476 | 1.6 | 9 | −7.4 | −476 | Unpublished data |
| Bokrijk, Belgium | VSSF | 14 | 13 | 7 | 1.6 | 1.5 | 0.1 | 7 | Unpublished data |
| Zemst-Larbeek, Belgium | VSSF | 16 | 11 | 31 | — | — | — | — | Unpublished data |
| Sint-Niklaas-Heimolen, Belgium | VSSF | 11 | 14 | −27 | — | — | — | — | Unpublished data |
| San Diego, California | FWS/SSF | 300 | 111 | 63 | 53.87 | 19.93 | 33.94 | 63 | Sinicrope *et al.* (1992) |
| San Diego, California | FWS/SSF | 300 | 153 | 49 | 66.90 | 34.12 | 32.78 | 49 | Sinicrope *et al.* (1992) |
| Sacramento, California | FWS | 5.6 | 6.0 | −7 | 1.29 | 0.87 | 0.42 | 32 | Dombeck *et al.* (1998) |
| Tres Rios, Arizona | FWS C2 | 4.6 | 4.8 | −4 | 2.65 | 1.58 | 1.07 | 40 | Wass, Gerke, and Associates (2002) |
| Orlando Easterly, Florida | Tertiary | 11.9 | 10.2 | 14 | — | — | — | — | Simon (2003) |

overland flow wetlands, the rate-limiting step is the transfer of nickel from the water to sorption and precipitation sites in the sediments. Traditional mass transfer modeling would utilize a first-order model for that process (see Chapter 6). It is also likely that percentage reductions in Table 11.27 would be greater if the hydraulic efficiencies were higher, i.e., closer to plug flow. The Sinicrope *et al.* (1992) data correspond to plug flow *k*-values from 15 to 18 m/yr for an FWS–SSF combination system.

A significant effort at detailed process modeling was the development and calibration of the CWFATE model for the Sacramento project (Jones & Stokes Associates, 1993; Nolte and Associates, 1998a, b). CWFATE attempts to describe water budgets, nickel biomass cycles, and partitioning, but not chemical precipitation.

Plants are a minor repository for removed nickel. Vymazal and Krása (2005) found 71% of the added nickel in sediments and belowground plant parts in a HSSF wetland treating domestic wastewater, and 21% in aboveground plant parts. The majority of removed nickel is therefore associated with wetland sediments, in sorbed or chemically precipitated forms. The principal forms are very likely associated with sulfides and exchange sites on organics.

### *Anaerobic Wetlands*

Under anaerobic conditions nickel reacts with hydrogen sulfide to form insoluble sulfides (Equation 11.40, see also Equations 11.1 and 11.2). Decaying wetland plants are a potential carbon source. Bioavailable carbon production in wetlands is about 60 g C/m$^2$·yr (see the denitrification section of Chapter 9). According to Equations 11.2 to 11.4, this would support sulfide production of 80 g S/m$^2$·yr, a factor of 50 less than the typical design rate recommended by Wildeman *et al.* (1993b). The corresponding nickel removal rate is 147 g Ni/m$^2$·yr, which is also far less than the design value of 730 g Ni/m$^2$·yr tentatively advanced by the PIRAMID Project of the European Union (PIRAMID Consortium, 2003a) for anaerobic wetlands.

Vymazal (2003) reported a 92% removal of nickel from the municipal sewage in a HSSF wetland in the Czech Republic with an average outflow Ni concentration of 0.7 µg/L and *k*-value of 25 m/yr. Byekwaso *et al.* (2002) reported removal of 65% for Ni in a cobalt extracting waste in Uganda in a combination FWS–HSSF wetland system. However, results from three other HSSF systems in the Czech Republic (Table 11.27) exhibited increased nickel concentration at the outflow with an increase varying between 9% and 109%. This is not surprising because concentration and distribution of many elements, such as Ni (and Cu, Zn, or Cd) in sediments, is strongly influenced by adsorption or co-precipitation with Fe and Mn oxides (Goldberg, 1954; Krauskopf, 1956; Jenne, 1968; Murray and Gill, 1978; Feely *et al.*, 1983; Ferris *et al.*, 1989). Nickel, in particular, is co-precipitated in Fe oxides (Stumm and Morgan, 1970). Under anoxic/anaerobic conditions both Fe and Mn are reduced to more soluble ferrous and manganous compounds and during this process co-precipitated metals are released.

### *Aerobic Wetlands*

FWS wetlands typically produce 200 dry g/m$^2$·yr of new sediments. Sustainable sorption would contribute about 13 mg Ni/m$^2$·yr for water at 50 µg/L, and 1,780 mg Ni/m$^2$·yr for water at 10 mg/L, based on Chen *et al.* (2001). These nickel removal rates are also far less than the design value of 15 g Ni/m$^2$·yr tentatively advanced by the PIRAMID Project of the European Union (PIRAMID Consortium, 2003a) for aerobic wetlands.

Vertical flow constructed wetlands provide more oxic conditions than HSSF systems; however, the results from 6 VF systems in the Netherlands and Belgium exhibited quite low removal efficiency for nickel ranging from +31 to –476% with the median value of 55%.

## Example Treatment Wetland for Nickel Removal

Drainage from mineralized Duluth Complex stockpiles located at LTV Steel Mining Company's Dunka Mine in northeastern Minnesota contains elevated concentrations of nickel, copper, cobalt, and zinc. This drainage had increased metal concentrations in nearby receiving waters to levels as much as 500 times natural background concentrations. A feasibility analysis, conducted in 1985, concluded that although an active treatment plant could generally achieve water quality guidelines, a more cost-effective, passive approach might also be successful.

Wetland treatment has been a crucial aspect of this passive approach; although previous work had demonstrated the effectiveness of peat to remove trace metals from mine drainage (Eger *et al.*, 1980), no field data from an actual treatment system existed. In 1986, LTV Steel Mining Company and the Minnesota Department of Natural Resources conducted a cooperative program to develop data on optimal wetland treatment design and system life (Eger, 1994). The goal of the program was to collect data for the design of full-scale treatment systems for the stockpile drainages at the Dunka Mine by constructing and operating wetland treatment test plots.

That preliminary work was followed by three full-scale projects, each in the 1–2 ha range, starting in 1992, 1993, and 1997, respectively (Eger *et al.*, 1993; Frostman, 1993). Removals initially were 98–99% for copper, cobalt, nickel, and zinc, but both incoming loads and mass removals declined with time (Eger *et al.*, 2000; Eger and Wagner, 2003).

## Lead

Lead occurs in surface waters most often in the divalent, Pb(II) form, which forms insoluble precipitates with sulfides, carbonates, sulfates, and chlorides. Lead combines with organic ligands to form soluble and colloidal complexes. Lead solubility is likely to be less than about 1 µg/L at a pH above 8.5, with increasing solubility at lower pH values and lower alkalinities. Lead is largely unavailable in most natural conditions because of its formation of insoluble salts and partitioning to dissolved organics and suspended particulates. Lead forms alkyl compounds, such as tetraethyl lead, a

## TABLE 11.28
## Lead Content of Wetland Plants and Sediments

| Wetland Plant and Part | Reference | Water (µg/L) | Solid (µg/g) |
|---|---|---|---|
| **Plant Shoots** | | | |
| *Typha latifolia* | Lan *et al.* (1990) | 1,570 | 90.6 |
| *Typha latifolia* | Ye *et al.* (1997) | Roadside | 19.0 |
| *Typha latifolia* | Ye *et al.* (1997) | Pb/Zn tailings | 40.0 |
| *Typha latifolia* | Wojcik and Wojcik (2000) | Pb/Zn tailings | 30.0 |
| *Typha latifolia* | Scholes *et al.* (1999) | Urban | 119.0 |
| *Typha latifolia* | Obarska-Pempkowiak *et al.* (2005) | Sewage | 0.04 |
| *Typha latifolia* | Maddison *et al.* (2003) | Sewage | <0.1 |
| *Typha latifolia* | Zhang *et al.* (1990) | 53.4 | 19.0 |
| *Typha domingensis* | Nolte and Associates (1998a) | 1.1 | 0.5 |
| *Typha domingensis* | Karpiscak *et al.* (2001a) | <200 | 1.4 |
| *Typha angustifolia* | Samecka-Cymerman and Kempers (2001) | 1 | 6.0 |
| *Scirpus acutus* | Nolte and Associates (1998a) | 1.1 | 0.9 |
| *Scirpus lacustris* | Samecka-Cymerman and Kempers (2001) | 1 | 15.3 |
| *Scirpus* spp. | Karpiscak *et al.* (2001a) | <200 | 1.1 |
| *Phalaris arundinacea* | Vymazal and Krása (2005) | 120 | 4.4 |
| *Phalaris arundinacea* | Vymazal *et al.* (2006) | Sewage | 0.04 |
| *Phalaris arundinacea* | Samecka-Cymerman and Kempers (2001) | 85 | 12.6 |
| *Phragmites australis* | Vymazal *et al.* (2006) | Sewage | 0.06 |
| *Phragmites australis* | Wojcik and Wojcik (2000) | Pb/Zn tailings | 160.0 |
| *Phragmites australis* | Vymazal and Krása (2005) | 120 | 5.6 |
| *Phragmites australis* | Scholes *et al.* (1999) | Urban | 75.5 |
| *Phragmites australis* | Obarska-Pempkowiak *et al.* (2005) | Sewage | 0.1 |
| *Phragmites australis* | Samecka-Cymerman and Kempers (2001) | 37 | 17.3 |
| *Juncus effusus* | Samecka-Cymerman and Kempers (2001) | 10 | 12.6 |
| **Plant Roots** | | | |
| *Typha latifolia* | Lan *et al.* (1990) | 1,570 | 444.3 |
| *Typha latifolia* | Ye *et al.* (1997) | Roadside | 25.0 |
| *Typha latifolia* | Ye *et al.* (1997) | Pb/Zn tailings | 1,108.0 |
| *Typha latifolia* | Wojcik and Wojcik (2000) | Pb/Zn tailings | 250.0 |
| *Typha latifolia* | Maddison *et al.* (2003) | Sewage | 30.0 |
| *Typha latifolia*[a] | Maddison *et al.* (2005) | Sewage | 18.9 |
| *Typha latifolia* | Obarska-Pempkowiak *et al.* (2005) | Sewage | 0.2 |
| *Typha latifolia* | Zhang *et al.* (1990) | 53.4 | 95.0 |
| *Typha domingensis* | Karpiscak *et al.* (2001a) | <200 | 3.4 |
| *Typha domingensis* | Nolte and Associate (1998a) | 1.1 | 1.2 |
| *Scirpus* spp. | Karpiscak *et al.* (2001a) | <200 | 5.9 |
| *Phragmites australis* | Wojcik and Wojcik (2000) | Pb/Zn tailings | 220.0 |
| *Phragmites australis* | Vymazal *et al.* (2006) | Sewage | 6.7 |
| *Phalaris arundinacea* | Vymazal *et al.* (2006) | Sewage | 6.1 |
| **Plant Rhizomes** | | | |
| *Typha latifolia* | Ye *et al.* (1997) | Roadside | 40.0 |
| *Typha latifolia* | Ye *et al.* (1997) | Pb/Zn tailings | 354.0 |
| *Typha latifolia* | Zhang *et al.* (1990) | 53.4 | 30.0 |
| *Scirpus acutus* | Nolte and Associates (1998b) | 1.1 | 1.5 |
| *Phragmites australis* | Vymazal *et al.* (2006) | Sewage | 0.7 |
| *Phalaris arundinacea* | Vymazal *et al.* (2006) | Sewage | 0.8 |
| **Sediments** | | | |
| Guangdong, China | Lan *et al.* (1990) | Pb/Zn tailings | 7,019 |
| *Typha latifolia* | Ye *et al.* (1997) | Roadside | 26 |
| *Typha latifolia* | Ye *et al.* (1997) | Factory | 434 |
| *Typha latifolia* | Ye *et al.* (1997) | Fluorspar tailings | 18,894 |
| Sacramento | Nolte and Associates (1998a) | Secondary | 6.375 |
| *Typha latifolia* | Zhang *et al.* (1990) | 53.4 | 840 |
| Nucice, Czech Republic | Vymazal (2003) | 120 | 150 |
| 11 lakes, Poland | Samecka-Cymerman and Kempers (2001) | 1–124 | 1–64 |

*Note:* The corresponding water concentrations are approximate.

[a] Roots and rhizomes.

former antiknock additive in gasoline. Methylation can occur microbially in the environment (Odum *et al.*, 2000).

The average concentration of lead in water reported for the world's rivers is 0.2 µg/L. For example, the Sacramento River in California had 0.77 µg/L dissolved, and 1.16 µg/L total lead in a 1991–1992 study (Jones & Stokes Associates, 1993).

Lead is a physiological and neurological toxin to humans, and therefore human health standards have been set, typically at 50 µg/L. Lead poisoning of waterfowl from direct ingestion of mine tailings water has been reported at Coeur d'Alene, Idaho (DeVolder *et al.*, 2003). U.S. EPA (2002a) has determined that lead is a toxic priority pollutant, and sets water quality criteria for freshwater concentrations of lead according to hardness. At hardness of 100 mg/L, the maximum concentration allowable is 65 µg/L, and the continuous concentration allowable is 2.5 µg/L, as dissolved lead. U.S. EPA (2002a) provides formulae for the limits at other hardnesses, with soft waters having lower criteria.

Sediment standards for lead have been set by the Province of Ontario, and adopted in some of the states in the United States. These set a lowest effect level of 31 µg/g, and a severe effects level of 250 µg/g (Persaud *et al.*, 1993; OMOEE, 1994). OMOEE (1994) has also identified the level for pre-settlement deposits as 23 µg/g in Great Lakes sediments. Lead does not appreciably biomagnify in aquatic in aquatic food-chains (Eisler, 1988; Odum *et al.*, 2000). For more details on the cycling of lead in aquatic ecosystems, see Nriagu (1978).

## Wetland Processing and Storage of Lead

Lead is found in wetland plants, algae, and sediments. Concentrations in tissues are higher when the plants or algae grow in lead-contaminated waters, such as the leachates from lead mine tailings piles (Table 11.28). In uncontaminated waters, macrophyte tissue concentrations of 1–20 µg/g dry mass are common, and the same range exists for algae in clean waters (Vymazal, 1995). In contrast, lead concentrations in plants in lead-contaminated waters are much higher, about 100–500 µg/g dry mass for macrophytes (Table 11.28), and 1,000–10,000 µg/g dry weight for algae (Vymazal, 1995). Aboveground plant parts are of lowest concentration, rhizomes are somewhat higher, and roots have the highest tissue concentrations. There is a direct relation between lead in the soil and lead in the roots of cattails (Figure 11.18).

Wetland sediments are typically the major repository of stored lead in wetlands. Any of several compounds may precipitate, and there is strong partitioning to wetland organic sediments. Therefore, the lead removed by the wetland builds up in the sediments, primarily in the top layers. For instance, O'Sullivan *et al.* (2004) found lead in the top 15 cm of sediments in a treatment wetland accumulated from an initial 31 mg/kg to 3,300–3,900 mg/kg dry mass after 4.58 years of exposure to variable water-phase lead. DeBusk *et al.* (1996) found that the top 2 cm of sediments in cattail mesocosms increased from about 10 mg/kg to 295 mg/kg dry mass upon exposure to 396 µg Pb/L for 13 months. The lead levels in the next deeper 3 cm fell to 135 mg/kg dry mass.

DeBusk *et al.* (1996) hypothesized that the lead was bound as sulfides, based upon acid extractions with analyses for lead and sulfides. However, other lead precipitates were identified by DeVolder *et al.* (2003), namely carbonates (up to 66%) and sulfates (up to 20%). Lead fate in wetlands is that removed lead is found in sediments and to a lesser extent in belowground plant parts. For instance, Vymazal and Krása (2005) found over 97% of the stored lead in those locations. Aboveground plant storage is minimal in comparison.

As for other metals, the storage of lead in wetlands entails a risk of creating potentially impaired sediments. For example, consider 200 dry g/m$^2$·yr of new sediments, originating from 1,000 dry g/m$^2$·yr of biomass production. If the sediment concentration is to be kept below some limiting level, such as the 250 µg Pb/g severe effects level of the Ontario guideline, Equation 11.22 can be solved for the allowable removal flux. For that Ontario guideline, deposition of 50 mg Pb/m$^2$·yr could be sustainably tolerated. Meeting rigorous sediment standards on a sustainable basis is not easily achieved for any but trace concentrations, such as those occurring in stormwater and municipal wastewater (Table 11.29).

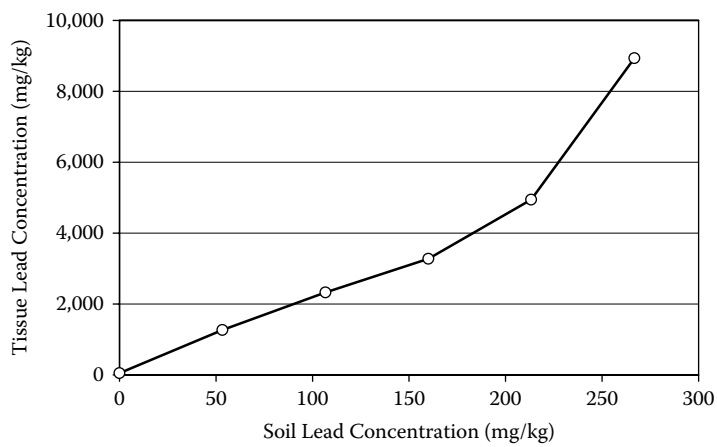



**FIGURE 11.18** Lead concentrations in cattail (*Typha angustifolia*) grown in various soil lead concentrations for 90 days. (Data from Panich-Pat *et al.* (2004) *Water, Air, and Soil Pollution* 155(1–4): 159–171.)

## TABLE 11.29
## Lead Reduction in Treatment Wetlands

| System | Notes | Concentration (µg/L) | | | Mass Rates (kg/ha·yr) | | | | Reference |
|---|---|---|---|---|---|---|---|---|---|
| | | In | Out | Reduction (%) | In | Out | Removal | Removal (%) | |
| **FWS** | | | | | | | | | |
| **Mine Water** | | | | | | | | | |
| Albright, Pennsylvania | Metal mine water | 2 | 1 | 50 | — | — | — | — | Hoover *et al.* (1998) |
| Tom's Gully, NT, Australia | Gold mine | 9 | 1 | 94 | — | — | — | — | Noller *et al.* (1994) |
| Woodcutters, NT, Australia | Ag/Pb/Zn mine | 12 | 0 | 98 | — | — | — | — | Noller *et al.* (1994) |
| Orcopampa, Peru | Metal mine water | 310 | 237 | 24 | — | — | — | — | Unpublished data |
| Palmerton, Pennsylvania | Zn mine | 1,000 | 100 | 90 | — | — | — | — | Hafner (1993) |
| Biala river, Poland | Pb/Zn mine | 1,200 | 500 | 58 | — | — | — | — | Wojcik and Wojcik (2000) |
| Guangdong, China | Pb/Zn mine | 1,570 | 87 | 94 | 1,948 | 108 | 1,840 | 94 | Lan *et al.* (1990) |
| Middlesex University, UK | Simulated mine water | 5,000 | 480 | 90 | — | — | — | — | Mungur *et al.* (1997) |
| **Landfill Leachate** | | | | | | | | | |
| Isanti-Chisago, Minnesota | Landfill leachate | 1 | 1 | 0 | — | — | — | — | Kadlec (2003c) |
| New Hanover County, NC | Landfill leachate | 3 | 3 | 0 | — | — | — | — | Unpublished data |
| Monroe County, New York | Landfill leachate | 13 | 5 | 62 | — | — | — | — | Eckhardt *et al.* (1999) |
| Orange County, Florida | Mesocosms | 396 | 196 | 51 | — | — | — | — | DeBusk *et al.* (1996) |
| **Stormwater** | | | | | | | | | |
| Hidden River, Florida | Urban | 2 | 1 | 40 | 0.04 | 0.01 | 0.033 | 83 | Carr and Rushton (1995) |
| Irondequoit Creek, New York | Urban | 7 | 5 | 29 | 3.18 | 2.27 | 0.91 | 29 | Coon *et al.* (2000) |
| Crestwood, Virginia | Urban | 12 | 3 | 74 | 0.013 | 0.003 | 0.010 | 75 | Godrej *et al.* (1999) |
| Sanford, Florida | Urban | 25 | — | — | 1.97 | 0.89 | 1.080 | 55 | Harper *et al.* (1986) |
| Greenwood, Florida | Urban | 63 | 5 | 92 | 0.246 | 0.101 | 0.145 | 59 | McCann and Olson (1994) |
| Savannah River, South Carolina | Lab water, urban | — | — | 83 | — | — | — | — | Nelson *et al.* (2004) |
| Kildare, Ireland | Highway runoff | 27–92 | 16–45 | 62 | — | — | — | — | Higgins *et al.* (2006) |
| **Municipal Wastewater** | | | | | | | | | |
| Sacramento, California | FWS | 1.1 | 0.2 | 80 | 0.22 | 0.04 | 0.17 | 80 | Nolte and Associates (1998) |
| Christchurch, New Zealand | FWS | 1.8 | 1.5 | 17 | — | — | — | — | Chague-Goff (2005) |
| Central Illinois | FAP | 2.6 | 1 | 58 | — | — | — | — | Jin *et al.* (2003) |
| Central Illinois | FWS | 2.6 | 1 | 58 | — | — | — | — | Jin *et al.* (2003) |
| Lakeland, Florida | FWS | 8.2 | 6.7 | 18 | — | — | — | — | NADB database (1998) |
| San Diego, California | FWS/SSF | 300 | 42 | 86 | 53.9 | 7.5 | 46.3 | 86 | Sinicrope *et al.* (1992) |
| San Diego, California | FWS/SSF | 300 | 60 | 80 | 66.9 | 13.4 | 53.5 | 80 | Sinicrope *et al.* (1992) |
| **Median (All FWS)** | | | | **62** | | | **0.91** | | |

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| **HSSF** | | | | | | | | | |
| Central Illinois | SSF | 2.6 | 1 | 63 | — | — | — | — | Jin *et al.* (2003) |
| Poland | HSSF/municipal | 15.7 | 7.3 | 54 | — | — | — | — | Obarska-Pempkowiak (2003) |
| Montana | HSSF/mining | 3 | 3 | 0 | — | — | — | — | Pantano *et al.* (2000) |
| Uganda | FWS/HSSF | 43 | 27 | 37 | — | — | — | — | Byekwaso *et al.* (2002) |
| Nučice, Czech Republic | HSSF/municipal | 120 | 2.8 | 98 | 11.9 | 0.3 | 11.6 | 98 | Vymazal (2003) |
| Mořina, Czech Republic | HSSF/municipal | 19 | 22 | −16 | 1.8 | 2.1 | −0.3 | −16 | Unpublished data |
| Břehov, Czech Republic | HSSF/municipal | 3.9 | 3.4 | 13 | 1.0 | 0.8 | 0.2 | 13 | Unpublished data |
| Slavošovice, Czech Republic | HSSF/municipal | 3.5 | 11.2 | −220 | 0.34 | 1.1 | −0.76 | −220 | Unpublished data |
| **Median (All HSSF)** | | | | **25** | | | | | |
| **VSSF** | | | | | | | | | |
| Wepserveen, The Netherlands | VSSF/municipal + dairy | 5 | 3 | 40 | 0.03 | 0.02 | 0.01 | 40 | Unpublished data |
| Wepserveen, The Netherlands | VSSF/municipal | 5 | 1 | 80 | 0.49 | 0.09 | 0.40 | 80 | Unpublished data |
| Bokrijk, Belgium | VSSF/municipal | 11 | 6 | 46 | 1.96 | 1.1 | 0.86 | 46 | Unpublished data |
| Zemst-Larbeek, Belgium | VSSF/municipal | 17 | 6 | 65 | — | — | — | — | Unpublished data |
| Sint-Niklaas-Heimolen, Belgium | VSSF/municipal | 23.5 | 20 | 15 | — | — | — | — | Unpublished data |
| **Median (All VSSF)** | | | | **46** | | | | | |

### *Lead Removal Processes*

Constructed wetlands have the potential to trap and remove lead contained in wastewater. Long-term removal is expected to occur by accumulation and burial in the sediments.

### *Particulate Settling*

The majority of total lead in wetland waters is usually bound to dissolved organic matter (Koelmans and Radovanovic, 1998), but some fraction is associated with incoming particulate matter. Particulate lead is subject to removal with total suspended solids (TSS), by processes of settling and trapping.

### *Chemical Precipitation*

Lead forms very insoluble compounds with sulfide, sulfate, and carbonate, among others:

$$S^{2-} + Pb^{2+} \rightarrow PbS \quad (11.41)$$

$$SO_4^{2-} + Pb^{2+} \rightarrow PbSO_4 \quad (11.42)$$

$$CO_3^{2-} + Pb^{2+} \rightarrow PbCO_3 \quad (11.43)$$

Sulfide is formed by SRB in the anaerobic zones of treatment wetlands, while incoming sulfate persists in aerobic zones. If a source of sulfate is present in the incoming water, the wetland can be configured to provide a sustainable supply of sulfide (Sobolewski, 1996). Carbonates are ubiquitous in wetland environments. Equilibrium conditions for different lead species are summarized in Figure 11.7.

### *Lead Sorption*

Lead may partition to organic constituents in the wetland sediments, or to inorganic substrates, such as oxyhydroxides of iron and manganese (Koelmans and Radovanovic, 1998). There are a finite number of exchange sites, and hence the uptake of lead reaches a maximum. The Langmuir isotherm provides for such saturation, and fits peat sorption data (Ho and McKay, 1999; see Equation 11.11).

The maximum capacity for lead is $C_{Smax} = K/a$. Ho and McKay (1999) found 122 mg/g for an unspecified peat. It is noteworthy that their value of $a = 0.374$ L/mg means that the isotherm is essentially linear up to about 1.0 mg Pb/L. Chen *et al*. (2001) determined values of $a = 3.09$ L/mg and $K = 381$ L/g for a New Zealand peat. The corresponding $C_{Smax} = K/a = 104$ mg/g. Therefore, in most nonmine water situations, a simple linear partition coefficient suffices. The value of $K = 45.6$ L/g for peat, but was much higher for mineral sediments dominated by manganese, $K = 894$ L/g (Koelmans and Radovanovic, 1998).

The concepts of partitioning may be carried over to the absorption of lead by algae as a means of removal. Aksu and Kutsal (1991) determined Freundlich isotherms (Equation 11.13) for the partitioning of lead to *Chlorella vulgaris*, regarded as a sorbent. The parameter values for *Chlorella vulgaris* were $n = 0.43$ and $K_F = 17.3$ at pH = 5 and $T = 25°C$.

The strong propensity for partitioning to sediments means that if ionic lead is present in the water, there will be associated lead in large amounts in the sediments at equilibrium. However, in the presence of dissolved organic matter (DOM) typical of wetlands, the large majority of dissolved lead occurs as complexes with organic ligands and not as ionic lead. For example, Koelmans and Radovanovic (1998) suggest that over 97% is soluble-bound to DOM in lakes in the Netherlands.

## Performance of Wetlands for Lead Removal

Lead is removed in wetlands when the incoming amounts are greater than background (see Table 11.29). For 26 FWS systems, with inlet lead ranging from 1 to 5,000 µg Pb/L, the median concentration reduction is 62%. Removal rates are positively correlated with inlet concentrations, with increasing removals as a response to increasing inlet concentrations (log–log, $R^2 = 0.71$).

Removal of lead in HSSF systems is highly variable (see Table 11.29), with removal efficiencies between –220% and 98% for eight systems and the median removal of 25%. However, most data available from SSF constructed wetlands are from the systems treating municipal sewage where lead is not the target and the inflow concentrations are usually very low as compared to, for example, mine drainage waters (see Table 11.29). Removal of lead in vertical flow constructed wetlands (see Table 11.27) is generally good, ranging between 15 and 80% with the median removal value of 46%.

Rate constant modeling of lead removal has not been attempted, and there are no estimates of $k$ or $C^*$. A significant effort at detailed process modeling was the development and calibration of the CWFATE model for the Sacramento project (Jones & Stokes Associates, 1993; Nolte and Associates, 1998a,b). CWFATE attempts to describe water budgets, lead biomass cycles, and partitioning, but not chemical precipitation.

## Example Treatment Wetlands for Lead Removal

### *Tara Mines, Ireland (O'Sullivan et al., 2004)*

Outukumpu Zinc operates the Tara Mines, 50 km north of Dublin, which are the largest European producer of lead and zinc ores. Water used in the extraction is enriched in metals and sulfate, and is stored in a tailings pond (O'Sullivan *et al.*, 2000). Two anaerobic wetlands were built to treat seepage from the tailings pond. A substrate of fine grit (75%) was amended with mushroom compost (25%), and subjected to a hydraulic loading of about 4 cm/d. These treatment wetlands were analyzed over a five-year period. Lead, zinc, iron, and sulfate were removed from tailings water passing though the treatment systems and were significantly retained in the substrates. Lead was reduced from 17 µg/L to below detection. Algal lead content was reduced from 45 µg/g at the inlet, to 18 µg/g. Results showed that the majority of metals and sulfur were retained in immobile residual forms, principally as metal sulfides. The neutral-alkaline pH and reducing biogeochemistry of the treatment systems suggest that metals are not toxic under these conditions.

*Sacramento, California* (*Dombeck et al., 1998*)

The Sacramento Constructed Wetland Demonstration Project was conducted from January 1994 to December 1998. One of the principal goals of the project was to identify the fate of trace metals within FWS treatment wetlands (Nolte and Associates, 1998a). Four of the 11 pilot wetlands were operated as flow through marshes with water depths of about 45 cm and a nominal detention time of about ten days. Lead was one of six metals that consistently exhibited removals of over 60%. Over the five years, lead was reduced from 1.14 µg/L to 0.23 µg/L (80%). The annual mass removals averaged 77%. Only 25–30% of the lead was in dissolved form. The five-year average lead concentration in sediments was 6.4 mg/kg, but there was a linear trend increase over the project period of 1.3 mg Pb/kg·yr, from 5.2 mg Pb/kg in 1994 to 9.4 mg Pb/kg. Lead was below state and federal sediment standards. Lead in both aboveground and belowground plant parts was quite low (see Table 11.28).

## Cadmium

Cadmium is currently used primarily in the production of Ni-Cd batteries (75%), whereas it was formerly used in electroplating (75% prior to 1940). However, it is estimated that less than 2% of cadmium entering the environment is from such sources, because most comes from waste incineration and the metallurgical industry (Laws, 1993; Plachy, 2000). Cadmium concentration in wetland sediments in a tidal freshwater marsh on the Hudson River in New York was measured between 2 and 42,000 mg/kg dry weight, near the outfall from an Ni-Cd battery facility (Hazen and Kneip, 1980). The cadmium concentration in surface water at this site averaged about 32 µg/L; plant tissue concentrations ranged from 13 to 513 mg/kg dry weight, near the outfall; and invertebrates and fish had muscle tissue concentrations generally between 0.4 and 59 mg/kg dry weight.

Cadmium is a naturally occurring heavy metal with no known nutritional requirement for biota. Knight (1980) and Giesy *et al.* (1979) did observe a slight stimulation to aquatic ecosystem productivity at very low cadmium concentrations and increased populations of some periphytic algal species at cadmium concentrations as high as 10 µg/L. In surface waters, cadmium typically occurs as Cd(II) and is most soluble at low pH in waters with low hardness. In solution, cadmium is present as the ion and in a number of complexed, soluble compounds with carbonate, sulfate, chloride, hydroxides, and humates. These cadmium complexes are highly adsorbable on organic particulates.

U.S. EPA (2002a) sets criteria for freshwater concentrations of cadmium according to hardness. At hardness of 100 mg/L, the maximum concentration allowable is 2.0 µg/L, and the continuous concentration allowable is 0.25 µg/L, as dissolved cadmium. U.S. EPA (2002a) provides formulae for the limits at other hardnesses, with soft waters having lower criteria. Sediment standards for cadmium have been set by the province of Ontario, and adopted in some of the states in the United States. These set a lowest effect level of 0.6 µg/g, and a severe effects level of 10 µg/g (Persaud *et al.*, 1993; OMOEE, 1994). OMOEE (1994) has also identified the level for presettlement deposits as 1.1 µg/g in Great Lakes sediments. The state of Wisconsin has a guideline for a probable effects sediment concentration of 5 µg/g. Cadmium is toxic to benthic macroinvertebrates. At concentrations of 2.5–5.0 µg/g, there is 22% mortality of the test organism *Chironomus tentans* in 10–14 days, and 60% mortality of the test organism *Hyalella azteca* (Wisconsin Department of Natural Resources, 2003).

Freshwater biota are sensitive to elevated cadmium concentrations. Birds and mammals are comparatively less sensitive. Giesy *et al.* (1979) found that aquatic fungi were sensitive to cadmium concentrations as low as 5 µg/L, resulting in decreased nutrient cycling in shallow, wetland-like streams and lower primary productivity. Eisler (1985) estimated that adverse effects of cadmium on fish and wildlife are possible at water concentrations above about 3 µg/L in freshwater or 0.1 mg/kg wet weight in the diet. However, results demonstrate that laboratory toxicity tests may overestimate chronic toxicity responses of *Hyalella azteca* to cadmium in effluent-dominated streams (Brooks *et al.*, 2004; Stanley *et al.*, 2005). The 42-day survival Lowest-Observed-Effect Concentrations (LOECs) were 0.94, 4.53, and 22.97 µg/L for the laboratory hard water, laboratory effluent, and stream mesocosm exposures, respectively. Similar results were found for the test organisms *Ceriodaphnia dubia* and *Pimephales promales*. These authors concluded that binding to organic ligands was likely responsible for reduced toxicity.

### Wetland Processing and Storage of Cadmium

Wetland plants have trace amounts of cadmium in their tissues, typically with higher values for roots and rhizomes than for aboveground plant parts (Table 11.30). Plant shoots contain less than 4 µg/g dry weight for nonmine water situations, but concentrations can become elevated in wetland plant tissues in response to exposure to very high concentrations of cadmium (Figure 11.19; Maine *et al.*, 2001; Ait Ali *et al.*, 2004; Göthberg *et al.*, 2004). Some freshwater algae protect their environment by sequestering cadmium, even though it is toxic. *Chlorella vulgaris*, a single-celled algae, can remove over 10% of its dry weight of cadmium (Kadlec and Srinivasan, 1995; Pascucci and Kowalak, 1999).

The concentration of cadmium in the roots of water spinach (*Ipomoea aquatica*) was found to increase with increasing cadmium in water, up to 4,250 µg/g at 5 mg/L (Göthberg *et al.*, 2004) (Figure 11.19). Concentrations in leaves and stems were much lower, 10–30 µg/g in leaves and 50–130 µg/g in stems. However, root mass is a low proportion of the total; thus the mass fraction stored in roots was 50–70%.

Cadmium was transferred primarily to soils (42% of removal) and fine roots (35%) in the Sinicrope *et al.* (1992) study. Vymazal and Krása (2005) found 44% of the stored cadmium was in aboveground tissues, and 33% was in belowground roots, rhizomes, and sediments in the Nučice, Czech Republic, treatment wetland. DeBusk *et al.* (1996) found that the belowground storages were 7.0% in plants (nearly all in roots), 93.0%

## TABLE 11.30
## Examples of Cadmium Content of Plant Tissues and Sediments in Treatment Wetlands

| Wetland Plant and Part | Reference | Water (µg/L) | Solid (µg/g) |
|---|---|---|---|
| **Plant Shoots** | | | |
| *Scirpus acutus* | Nolte and Associates (1998b) | 0.08 | 0.05 |
| *Scirpus lacustris* | Samecka-Cymerman and Kempers (2001) | 0.1 | 1.5 |
| *Phalaris arundinacea* | Vymazal and Krása (2005) | 0.43 | 1.09 |
| *Phalaris arundinacea* | Vymazal (2003) | 0.43 | 2.00 |
| *Phalaris arundinacea* | Deng *et al.* (2004) | Cu | 1.00 |
| *Phalaris arundinacea* | Vymazal *et al.* (2006) | Sewage | 0.008 |
| *Phalaris arundinacea* | Samecka-Cymerman and Kempers (2001) | 2.8 | 1.4 |
| *Phragmites australis* | Samecka-Cymerman and Kempers (2001) | 1.2 | 2.3 |
| *Phragmites australis* | Vymazal *et al.* (2006) | Sewage | 0.004 |
| *Phragmites australis* | Vymazal and Krása (2005) | 0.43 | 2.25 |
| *Cyperus alternifolius* | Cheng *et al.* (2002) | 9 | 0.30 |
| *Glyceria maxima* | Obarska-Pempkowiak *et al.* (2005) | Sewage | 0.14 |
| *Typha latifolia* | Obarska-Pempkowiak *et al.* (2005) | Sewage | 0.21 |
| *Typha latifolia* | Mungur *et al.* (1995) | 6.5 | 3.40 |
| *Typha latifolia* | Eckhardt *et al.* (1999) | < 1 | 1.36 |
| *Typha latifolia* | Maddison *et al.* (2005) | Sewage | 0.47 |
| *Typha latifolia* | Lan *et al.* (1990) | 22.0 | 0.56 |
| *Typha angustifolia* | Deng *et al.* (2004) | Cu | 0.50 |
| *Typha angustifolia* | Samecka-Cymerman and Kempers (2001) | 0.1 | 0.5 |
| *Juncus effusus* | Deng *et al.* (2004) | Multi-metal | 3.00 |
| *Juncus effusus* | Samecka-Cymerman and Kempers (2001) | 1.2 | 3.1 |
| **Plant Roots** | | | |
| *Phragmites australis* | Mungur *et al.* (1995) | 6.5 | 15.00 |
| *Phragmites australis* | Vymazal *et al.* (2006) | Sewage | 0.18 |
| *Cyperus alternifolius* | Cheng *et al.* (2002) | 9 | 9.20 |
| *Phalaris arundinacea* | Vymazal (2003) | 0.43 | 7.50 |
| *Phalaris arundinacea* | Deng *et al.* (2004) | Cu | 7.00 |
| *Phalaris arundinacea* | Vymazal *et al.* (2006) | Sewage | 0.18 |
| *Iris pseudocoris* | Mungur *et al.* (1995) | 6.5 | 12.00 |
| *Typha latifolia* | Eckhardt *et al.* (1999) | < 1 | 0.59 |
| *Typha latifolia* | Obarska-Pempkowiak *et al.* (2005) | Sewage | 0.33 |
| *Typha latifolia* | Mungur *et al.* (1995) | 6.5 | 4.00 |
| *Typha latifolia* | Lan *et al.* 1990 | 22.0 | 4.03 |
| *Typha latifolia*[a] | Maddison *et al.* (2005) | Sewage | 0.39 |
| *Typha angustifolia* | Deng *et al.* (2004) | Cu | 3.00 |
| *Juncus effusus* | Deng *et al.* (2004) | Multi-metal | 10.00 |
| **Plant Rhizomes** | | | |
| *Cyperus alternifolius* | Cheng *et al.* (2002) | 9 | 1.60 |
| *Scirpus acutus* | Nolte and Associates (1998b) | 0.08 | 0.26 |
| *Typha latifolia* | Eckhardt *et al.* (1999) | < 1 | 0.09 |
| *Phalaris arundinacea* | Vymazal *et al.* (2006) | Sewage | 0.04 |
| *Phragmites australis* | Vymazal *et al.* (2006) | Sewage | 0.016 |
| **Sediments** | | | |
| Dagenham | Scholes *et al.* (1999) | Urban | 6.50 |
| Brentwood | Scholes *et al.* (1999) | Urban | 6.20 |
| Site 3 | Mungur *et al.* (1995) | 46.0 | 2.90 |
| Site 4 | Mungur *et al.* (1995) | 8.0 | 4.50 |
| *Typha latifolia* | Eckhardt *et al.* (1999) | < 1 | 1.39 |
| *Typha latifolia* | Lan *et al.* (1990) | 22.0 | 24.79 |
| *Phalaris arundinacea* | Vymazal (2003) | 0.43 | 27.50 |
| *Cyperus alternifolius* | Cheng *et al.* (2002) | 9 | 1.15 |
| Sacramento | Nolte and Associates (1998) | Secondary | 0.45 |
| Gdansk, Poland | Obarska-Pempkowiak (2001) | Sewage | 0.98 |
| Bielkowo, Poland | Obarska-Pempkowiak *et al.* (2005) | Sewage | 7.10 |
| Widows Creek, Alabama | Ye *et al.* (2001a, b) | 48 | 30–70 |
| Guangdong, China | Deng *et al.* (2004) | Pb/Zn | 17.00 |
| Hubei, China | Deng *et al.* (2004) | Cu | 17.00 |
| Hunan, China | Deng *et al.* (2004) | Multi-metal | 46.00 |
| 11 lakes, Poland | Samecka-Cymerman and Kempers (2001) | 0.1–4.9 | 0–3.3 |

[a] Roots and rhizomes.

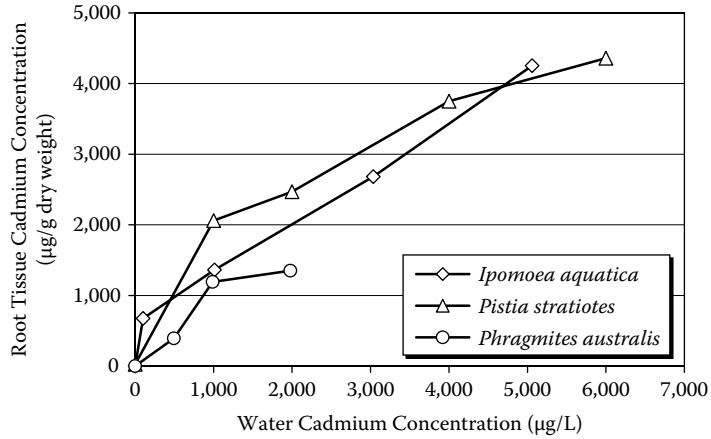



**FIGURE 11.19** Concentration of cadmium in the roots of aquatic plants. (Data for *Ipomoea aquatica* from Göthberg *et al*. (2004) *Journal of Environmental Quality* 33(4): 1247–1255. Data for *Pistia stratiotes* from Maine *et al*. (2001) *Water Research* 35(11): 2629–2634. Data for *Phragmites australis* from Ait Ali *et al*. (2004) *Aquatic Botany* 80: 163–176.)

in sediments for *Typha domingensis*; and 9% in plants, 91% in sediments for *Lemna minor,* in a Florida mesocosm study. Valiela *et al.* (1976) found 97% of the stored cadmium from wastewater in sediments and 3% in salt marsh plants.

During the start-up phase of a treatment wetland, stored cadmium will contribute to a buildup of the metal in preexisting sediments. That buildup was measured to be from an initial sediment concentration of near-zero to 150–225 µg/g dry mass in cattail (*Typha domingensis*) mesocosms over a two-year period when exposed to 105 µg/L of cadmium (DeBusk *et al*., 1996).

As for other metals, cadmium storage in wetlands eventually creates sustained sediment concentrations that reflect the dilution of new cadmium deposits by the accretion of new wetland solids. If the sediment concentration is to be kept below some limiting level, such as the 10-µg Cd/g severe effects level of the Ontario guideline, Equation 11.22 can be solved for the allowable removal flux. For that Ontario guideline, deposition of 2 mg Cd/m$^2$·yr (0.02 kg/ha·yr) could be sustainably tolerated. Meeting rigorous sediment standards on a sustainable basis is not easily achieved for any but trace concentrations, such as those occurring in stormwater and municipal wastewater (Table 11.31).

### Cadmium Removal Processes

Cadmium is removed by particulate settling and trapping, chemical precipitation and co-precipitation, partitioning to sediments, and to the least extent by plant uptake.

### Precipitation

Cadmium forms very insoluble compounds with sulfide and carbonate, among others:

$$S^{2-} + Cd^{2+} \rightarrow CdS \qquad (11.44)$$

$$CO_3^{2-} + Cd^{2+} \rightarrow CdCO_3 \qquad (11.45)$$

Precipitation of the carbonate can be microbially mediated, for example, by *Alcaligenes denitrificans* (Remacle *et al*., 1992). The removal of cadmium can also take place through co-precipitation with iron, manganese, and aluminum oxyhydroxides. Cadmium may be incorporated into the lattice of iron oxides, Cd-hematite, and Cd-goethite being formed. The mole percent cadmium which can replace iron in the iron oxides increases with increasing cadmium in the system. Although iron and manganese oxides are generally excellent scavengers for other metals, they are unstable under anoxic conditions (Knox *et al*., 2004).

### Cadmium Sorption

The Langmuir isotherm (Equation 11.11) fits sorption data. The maximum capacity for cadmium is $C_{Smax} = K/a$. Miretsky *et al*. (2006) found 36, 33, and 37 µg/g for detritus of *Spirodela intermedia, Lemna minor,* and *Pistia stratiotes,* respectively. The corresponding $K$ = 5.85, 14.58, and 2.19 L/g, and $a$ = 1.45, 3.95, and 0.53 L/mg. The authors concluded that the adsorption process for the three species studied followed first-order kinetics, and the mechanism involved ion exchange between monovalent metals as counter ions present in the macrophytic biomass and heavy metal ions and protons taken up from water. Algal necromass was also found to be an effective sorbent (Pascucci and Kowalak, 1999).

In contrast, peat was found to have capacity in excess of 200 mg/g, a good portion of which was nonexchangeable (Fine *et al*., 2005). Oxic sediments in Oregon rivers had cadmium sorption capacities of 190–920 µg/g (Fu and Allen, 1992). When water concentrations are low, sorption would fall in the linear range, and hence simple proportionality may be used, in the form of a partition coefficient. Koelmans and Radovanovic (1998) determined that most cadmium partitioned to iron and manganese oxyhydroxides in lakes in the Netherlands, and measured $K$ of about 286 L/g.

Freundlich isotherms (Equation 11.13) are also often used to fit partitioning data. Mohamed (2001) reported cadmium

## TABLE 11.31
## Cadmium Reduction in Treatment Wetlands

| System | Notes | Concentration (µg/L) | | | Mass Rates (kg/ha·yr) | | | | Reference |
|---|---|---|---|---|---|---|---|---|---|
| | | In | Out | Reduction (%) | In | Out | Removal | Removal (%) | |
| **Mine Water** | | | | | | | | | |
| Woodcutters, NT, Australia | Ag/Pb/Zn mine | 63 | 0.19 | 100 | — | — | — | — | Noller *et al.* (1994) |
| Palmerton, Pennsylvania | Zn mine | 1.06 | 0.02 | 98 | — | — | — | — | Hafner (1993) |
| Biala River, Poland | Pb/Zn mine | 0.01 | 0.00 | 60 | — | — | — | — | Wojcik and Wojcik (2000) |
| Guangdong, China | Pb/Zn mine | 22 | 2.00 | 91 | 27 | 2 | 25 | 91 | Lan *et al.* (1990) |
| Thailand | Synthetic | 1–20,000 | 20–160 | 99 | — | — | — | — | Jindahl and Samorkhom (2005) |
| Widows Creek, Alabama | Ash leachate | 48.00 | 5.40 | 99 | 4.9 | 0.05 | 4.85 | 99 | Ye *et al.* (2001a) |
| **Landfill Leachate** | | | | | | | | | |
| Isanti-Chisago, Minnesota | Landfill leachate | 0.73 | 0.88 | −20 | — | — | — | — | Kadlec (2003c) |
| Orange County, Florida | Mesocosms | 105 | 52 | 50 | — | — | — | — | DeBusk *et al.* (1996) |
| **Industrial Wastewater** | | | | | | | | | |
| Santo Tomé, Argentina | Tool factory | 8.2 | 2.9 | 65 | 1.5 | 0.6 | 0.9 | 65 | Maine *et al.* (2006) |
| **Stormwater** | | | | | | | | | |
| Hidden River, Florida | Urban | 2 | 0.00 | 100 | 0.04 | 0.01 | 0.038 | 87 | Carr and Rushton (1995) |
| Crestwood, Virginia | Urban | 3 | 2.17 | 29 | 0.003 | 0.002 | 0.001 | 31 | Godrej *et al.* (1999) |
| Sanford, Florida | Urban | 4 | — | — | 0.33 | 0.10 | 0.233 | 71 | Harper *et al.* (1986) |
| Greenwood, Florida | Urban | 0.4 | 0.20 | 50 | 0.246 | 0.101 | 0.145 | 59 | McCann and Olson (1994) |
| Kildare, Ireland | Highway runoff | ND-8 | ND-2 | 62 | — | — | — | — | Higgins *et al.* (2006) |
| **Municipal Wastewater** | | | | | | | | | |
| Sacramento, California | FWS | 0.08 | 0.03 | 63 | 0.02 | 0.01 | 0.01 | 63 | Nolte and Associates (1998) |
| San Diego, California | FWS/SSF | 70 | 18 | 75 | 12.6 | 3.2 | 9.4 | 75 | Sinicrope *et al.* (1992) |
| San Diego, California | FWS/SSF | 70 | 15 | 79 | 15.6 | 3.3 | 12.3 | 79 | Sinicrope *et al.* (1992) |
| Brookhaven, New York | FWS | 43 | 0.55 | 98.7 | 2.4 | 0.0 | 2.4 | 99 | Hendrey *et al.* (1979) |
| Nučice, Czech Republic | HSSF/sewage | 0.43 | 0.1 | 77 | 0.042 | 0.01 | 0.032 | 77 | Vymazal and Krása (2005) |
| Mořina, Czech Republic | HSSF/sewage | 4.2 | 0.4 | 91 | 4.0 | 0.4 | 3.6 | 91 | Unpublished data |
| Břehov, Czech republic | HSSF/sewage | 0.28 | 0.28 | 0 | 0.07 | 0.07 | 0 | 0 | Unpublished data |
| Slavošovice, Czech Republic | HSSF/sewage | 0.14 | 0.20 | −43 | 0.014 | 0.020 | 0.006 | −43 | Unpublished data |
| Wepserveen, The Netherlands | VSSF/sewage + dairy | 0.22 | 0.13 | 41 | 0.0012 | 0.0007 | 0.0005 | 41 | Unpublished data |
| Wepserveen, The Netherlands | VSSF/sewage | 0.27 | 0.15 | 44 | 0.13 | 0.07 | 0.06 | 44 | Unpublished data |
| Bokrijk, Belgium | VSSF/sewage | 1 | 1 | 0 | 0.12 | 0.12 | 0 | 0 | Unpublished data |
| Zemst-Larbeek, Belgium | VSSF/sewage | 0.5 | 0.5 | 0 | — | — | — | — | Unpublished data |
| Sint-Niklaas-Heimolen, Belgium | VSSF/sewage | 5 | 5 | 0 | — | — | — | — | Unpublished data |

*Note:* ND = Nondetect (e.g., below detection limit).

sorption by the nontoxic cyanobacterium *Gloeothece magna*. He determined $n = 0.5$ and $K_F = 910$ for dead cells at pH = 7.4, and $n = 0.54$ and $K_F = 2{,}400$ for living cells.

Complexing ligands may alter the adsorption behavior of heavy metals compared to a ligand free system, either by changes in solution speciation, or by interactions at the adsorbing mineral surface (Hoins *et al*., 1993).

### Performance of Wetlands for Cadmium Removal

Cadmium is removed in wetlands at all levels of inlet concentrations (Table 11.31). For 15 FWS systems, with inlet cadmium ranging from 0.4 to 20,000 µg Cd/L, the median concentration reduction is 79%. Removal rates are positively correlated with inlet concentrations, with increasing removals as a response to increasing inlet concentrations (log–log $R^2 = 0.64$). The design value of 0.02 g Cd/ $m^2$·d (73 kg/ha·yr) was tentatively advanced by the PIRAMID Project of the European Union (PIRAMID Consortium, 2003a) for compost wetlands. That recommendation was based upon the work of Ettner (1999). This is much higher performance than any system in Table 11.31.

HSSF wetlands (Table 11.31) exhibited variable removal effect between 43–91% with the median removal efficiency of 39%. However, the inflow concentrations in municipal sewage are very low, sometimes just above the detection limit. Five VF systems (Table 11.31) also showed variable treatment effect. While two systems provided removal of about 40%, three other systems did not produce any retention. The median removal of five VF systems was 0%.

Duckweed (*Lemna* spp.) can accumulate 7–14 kg Cd/ha·yr (2–4 mg Cd/$m^2$·d) (DeBusk *et al*., 1996). Accordingly, harvest could remove that amount, and such a procedure could be used if determined to be economical. The Sinicrope *et al*. (1992) data for SSF wetlands corresponds to removal rate constants in the range $25 < k < 35$ m/yr. The DeBusk *et al*. (1996) data suggest $k = 37$ m/yr. Maine *et al*. (2001) measured cadmium uptake by four floating plants: *Pistia stratiotes, Hydromistia stolonifera, Eichhornia crassipes*, and *Salvinia herzogii*. There was little difference among species, and the approximate first-order volumetric rate constant for removal in batch mesocosms was 0.5 $d^{-1}$. Removal was primarily (90%) to the plant roots. Removal was not complete, with the existence of water concentrations of 0.1–0.5 mg/L after a month of exposure. *Eichhornia crassipes* can strip cadmium from water, with an approximate volumetric rate constant of 0.1 $d^{-1}$ (Delgado *et al*., 1993).

There are not treatment wetlands that have been designed for the primary purpose of removing cadmium from contaminated waters. However, cadmium is a constituent of leachates from coal ash piles, and hence has been the co-target of treatment wetlands treating those leachates.

### Example Treatment Wetlands for Cadmium Removal

#### *Widows Creek, Alabama* (*Ye et al., 2001b*)

The Widows Creek FWS wetland was one of several constructed by the Tennessee Valley Authority (Brodie *et al*., 1988) in June 1986, for the primary purpose of treating acidic leachate from an abandoned coal ash pile. The wetland is located in Jackson County, Alabama, at the toe of the coal ash pile that creates the runoff. Three shallow marsh cells in series had an original cattail-vegetated area of 4,800 $m^2$. Cell 2 contained six finger berms that created a sinuous flow path, while Cells 1 and 3 had no internal berms. An infestation of army worms decimated the cattails (*Typha latifolia*) in 1987 (Snoddy et al., 1989), and the subsequent regrowth diversified to include soft rush (*Juncus effusus*), with trace amounts of spike rush (*Eleocharis palustris*) and woolgrass (*Scirpus cyperinus*). The wetland was investigated in its tenth year of operation, 1996–1997, to ascertain the long-term trace element uptake and assess the development of the wetland ecosystem (Ye *et al*., 2001b). The mean pH of the entering water was about 5.0, and the mean cadmium concentration was 48 µg/L. Removal of cadmium over the one-year period of study was 91%. Cattail shoots averaged 509 g/$m^2$, and soft rush 259 g/$m^2$. The aboveground storage of the annual cadmium loading was 0.05%, with tissue concentrations of 0.24–0.50 µg/g. Roots had tissue concentrations of 10–27 µg/g. Total storage in plants was only 2.5% of the annual loading. This study elucidated the high efficiency of trace element removal in the long run.

## Chromium

Chromium's principal uses and entry into the environment include electroplating, production of paints and pigments, tanning, wood preservation, chromium chemicals production, metal smelting, and pulp and paper production. In nature, trivalent chromium sorbs to various ligands and forms insoluble entities becoming unavailable for uptake by biota. Hexavalent chromium forms many soluble salts that can enter body membranes and induce a toxic response.

Chromium typically occurs in the trivalent [Cr(III)] or hexavalent [Cr(VI)] forms in surface waters. Major Cr(III) dissolved species are $Cr(OH)^{2+}$ and $Cr(OH)_3$, as well as several organic and a few inorganic complexes, whereas $HCrO_4^-$ and $CrO_4^{2-}$ are the main forms of Cr(VI) in natural waters (Richard and Bourg, 1991). The Cr(VI) form is the most toxic and is generally associated with the presence of industrial wastewaters. Cr(VI) is relatively unstable under most environmental conditions, and converts to the less toxic trivalent form in surface waters, especially when organic matter is present. Cr(VI) is reduced by dissolved organic carbons (DOCs) such as humic substances. Trivalent chromium hydroxides and chlorides are relatively insoluble and their formation may significantly reduce chromium availability to biota. Cr(III) and Cr(VI) are readily interconvertible under natural water conditions. Cr(VI) is reduced by Fe(II), dissolved sulfides, and certain organic compounds with sulfhydryl groups, while Cr(III) is oxidized by a large excess of $MnO_2$ and at a slow rate by $O_2$ under conditions approximating those in natural waters (Schroeder and Lee, 1975).

U.S. EPA (2002a) criteria for freshwater concentrations of Cr(III) are hardness dependent. At hardness of 100 mg/L the maximum concentration allowable is 570 µg/L, and the

continuous concentration allowable is 74 µg/L, as dissolved chromium. The criteria for Cr(VI) are not hardness dependent. The maximum concentration allowable for hexavalent chromium is 16 µg/L, and the continuous concentration allowable is 11 µg/L, as dissolved chromium.

Sediment standards for chromium have been set by the province of Ontario, and adopted in some of the states in the United States. These set a lowest effect level of 26 µg/g, and a severe effects level of 110 µg/g (Persaud *et al.*, 1993; OMOEE, 1994). OMOEE (1994) has also identified the level for presettlement deposits as 31 µg/g in Great Lakes sediments. The state of Wisconsin has a guideline for a probable effects sediment concentration of 110 µg/g (Wisconsin Department of Natural Resources, 2003).

Freshwater biota are sensitive to elevated chromium concentrations. Chromium (VI) is reported to be toxic to algae at concentrations between 20 and 10,000 g/L (Nriagu and Nieboer, 1988). Chromium is toxic to benthic macroinvertebrates. At concentrations of 55–110 µg/g, there is 22% mortality of the test organism *Chironomus tentans* in 10–14 days, and 60% mortality of the test organism *Hyalella azteca* (Wisconsin Department of Natural Resources, 2003). The chronic effects of hexavalent chromium on the fathead minnow (*Pimephales promelas*) were investigated by Pickering (1980). Survival was affected only at the high test concentration of 3.95 mg Cr/L. Growth of second-generation fish was not affected at concentrations of 1.0 mg/L or lower. Reproduction and hatchability of eggs were not affected at any chromium concentration tested.

## Wetland Processing and Storage of Chromium

Wetland plants have trace amounts of chromium in their tissues, typically with higher values for roots and rhizomes than for aboveground plant parts (Table 11.32). Plant shoots contain less than 12 µg/g dry weight for nonindustrial situations, but concentrations can become elevated in wetland plant tissues in response to exposure to very high concentrations of chromium (Maine *et al.*, 2001; Miretsky *et al.*, 2004). The concentrations of chromium in the roots and shoots of *Salvinia herzogii* and *Pistia stratiotes* were found to increase with increasing chromium in water, up to 6,200 µg/g at a water concentration of 6 mg/L (Maine *et al.*, 2004b) (Figure 11.20). Similar behavior and similar increased tissue concentrations were also observed in *Nymphaea spontanea* by Choo *et al.* (2006).

Chromium is transferred primarily to soils and sediments. Concentrations may be somewhat higher than in plant tissues, but the primary cause is the very high amount of sediment in an FWS wetland, compared to plant biomass. For instance, the sediment concentration at Sacramento was about 36 µg/g, compared to 1–4 µg/g for plant tissues. But the mass balance indicated 97% of the stored chromium was in the sediment (Nolte and Associates, 1998b).

During the start-up phase of a treatment wetland, stored chromium will contribute to a buildup of the metal in preexisting sediments. That buildup was measured to be from an initial sediment concentration of 78 µg/g, to 156 µg/g in the *A layer* at Sacramento, California, over a five-year period when exposed to 1.4 µg/L of chromium (Nolte and Associates, 1998b). This nominally exceeds the Ontario severe effects level of 110 µg/g.

The "layer cake" model (Equation 11.22) forecasts that chromium deposition in wetlands eventually creates sustained sediment concentrations that reflect the dilution of new chromium deposits by the accretion of new wetland solids. A dilution calculation on the top layer of deposited material determines the concentration. For example, consider 200 dry g/m$^2$·yr of new sediments, originating from 1,000 dry g/m$^2$·yr of biomass production. If the sediment concentration is to be kept below some limiting level, such as the 110 µg Cr/g severe effects level of the Ontario guideline, deposition of 22 mgCr/m$^2$·yr (0.22 kg/ha·yr) could be sustainably tolerated. Meeting such rigorous sediment standards on a sustainable basis is not easily achieved for any but trace concentrations, such as those occurring in stormwater and municipal wastewater (Table 11.33).

### *Chromium Removal Processes*

Chromium is removed by chemical co-precipitation, partitioning to sediments, and to the least extent by plant uptake.

### *Reduction of Cr(VI) to Cr (III)*

Because chromium exists in two valence states, there is an issue of the interconversions of Cr(III) and Cr(VI), with the hexavalent form being of most concern for toxicity. In general, wetlands are effective at removing Cr(VI) and converting it to Cr(III). However, the reverse reaction can also occur slowly via oxidation by $MnO_2$ (Eary and Ral, 1987). Xu and Jaffe (2006) found that Cr(III) was the dominant form of chromium in both sediments and plants, and precipitation of Cr(III) in the sediment was the major pathway responsible for the disappearance of aqueous Cr(VI) from the pore water. Lytle *et al.* (1998) showed that *Eichhornia crassipes* (water hyacinth), supplied with Cr(VI) in nutrient culture, accumulated nontoxic Cr(III) in root and shoot tissues. The reduction of Cr(VI) to Cr(III) appeared to occur in the fine lateral roots. The Cr(III) was subsequently translocated to leaf tissues. The chemical species of chromium in other plants, collected from wetlands that contained Cr(VI)-contaminated wastewater, was also found to be Cr(III). It is also possible that Cr(VI) can be reduced by Fe(II) in a wetland anoxic zone, leading to the formation of amorphous co-precipitates of oxyhydroxides of iron and chromium (Azizian and Nelson, 1992).

The mechanism of chromium removal from water by aquatic plants is not always well represented by a simple first-order process. There may be a rapid initial transfer, followed by a slower continued uptake, as observed in batch studies by Delgado *et al.* (1993) and Maine *et al.* (2004b). However, an exponential decline was observed in batch studies by Miretsky *et al.* (2004) and Gupta *et al.* (2001).

### *Chemical Precipitation*

The removal of chromium can take place through co-precipitation with iron, manganese, and aluminum oxyhydroxides.

## TABLE 11.32
## Examples of Chromium Content of Plant Tissues and Sediments in Treatment Wetlands

| Wetland Plant and Part | Reference | Water (µg/L) | Solid (µg/g) |
|---|---|---|---|
| **Plant Shoots** | | | |
| *Eichhornia crassipes* | Lytle *et al.* (1998) | 10,000 | 89 |
| *Eichhornia crassipes* | Maine *et al.* (2004a) | 22 | 9 |
| *Typha domingensis* | Maine *et al.* (2004a) | 22 | 12 |
| *Typha angustifolia* | Samecka-Cymerman and Kempers (2001) | 4.2 | 3.4 |
| *Phragmites australis* | Bragato *et al.* (2004) | 410 | 60 |
| *Phragmites australis* | Ranieri (2004) | 84 | 3.2 |
| *Phragmites australis* | Vymazal *et al.* (2006) | Sewage | 0.055 |
| *Phragmites australis* | Samecka-Cymerman and Kempers (2001) | 3.9 | 4.1 |
| *Phalaris arundinacea* | Samecka-Cymerman and Kempers (2001) | 3.3 | 4.6 |
| *Phalaris arundinacea* | Vymazal *et al.* (2006) | Sewage | 0.12 |
| *Carex* spp. | Bragato *et al.* (2004) | 410 | 9 |
| *Juncus effusus* | Samecka-Cymerman and Kempers (2001) | 4.0 | 5.1 |
| *Nymphaea spontanea* | Choo *et al.* (2006) | 1,000 | 400 |
| *Phragmites australis* | Xu and Jaffe (2006) | 500 | 5.88 |
| *Typha latifolia* | Xu and Jaffe (2006) | 500 | 2.71 |
| *Typha latifolia* | Paredes *et al.* (2006) | 1,500 | 3.5 |
| *Salvinia herzogii* | Maine *et al.* (2004b) | 2,000 | 221 |
| *Pistia stratiotes* | Maine *et al.* (2004b) | 2,000 | 123 |
| *Pistia stratiotes* | Miretsky *et al.* (2004) | 1,000 | 189 |
| *Spirodela intermedia* | Miretsky *et al.* (2004) | 1,000 | 99 |
| *Scirpus acutus* | Nolte and Associates (1998b) | 1.4 | 1.9 |
| *Scirpus lacustris* | Samecka-Cymerman and Kempers (2001) | 4.2 | 4.9 |
| *Typha domingensis* | Nolte and Associates (1998b) | 1.4 | 1.3 |
| **Plant Roots** | | | |
| *Eichhornia crassipes* | Lytle *et al.* (1998) | 10,000 | 5,265 |
| *Eichhornia crassipes* | Maine *et al.* (2004a) | 22 | 78 |
| *Typha domingensis* | Maine *et al.* (2004a) | 22 | 139 |
| *Phragmites australis* | Bragato *et al.* (2004) | 410 | 50 |
| *Phragmites australis* | Ranieri (2004) | 84 | 8.7 |
| *Phragmites australis* | Vymazal *et al.* (2006) | Sewage | 16.4 |
| *Phalaris arundinacea* | Vymazal *et al.* (2006) | Sewage | 15 |
| *Carex* spp. | Bragato *et al.* (2004) | 410 | 21 |
| *Nymphaea spontanea* | Choo *et al.* (2006) | 1,000 | 1,300 |
| *Phragmites australis* | Xu and Jaffe (2006) | 500 | 39.52 |
| *Typha latifolia* | Xu and Jaffe (2006) | 500 | 163.9 |
| *Typha latifolia* | Paredes *et al.* (2006) | 1,500 | 344 |
| *Salvinia herzogii* | Maine *et al.* (2004b) | 2,000 | 1,750 |
| *Pistia stratiotes* | Maine *et al.* (2004b) | 2,000 | 1,550 |
| **Plant Rhizomes** | | | |
| *Phragmites australis* | Bragato *et al.* (2004) | 410 | 20 |
| *Phragmites australis* | Vymazal *et al.* (2006) | Sewage | 2.4 |
| *Phalaris arundinacea* | Vymazal *et al.* (2006) | Sewage | 1.95 |
| *Carex* spp. | Bragato *et al.* (2004) | 410 | 10 |
| *Scirpus acutus* | Nolte and Associates (1998b) | 1.4 | 3.7 |
| *Typha domingensis* | Nolte and Associates (1998b) | 1.4 | 3.2 |
| **Sediments** | | | |
| Santo Tomé | Maine *et al.* (2004b) | 22 | 23 |
| *Phragmites australis* | Xu and Jaffe (2006) | 500 | 39 |
| *Typha latifolia* | Xu and Jaffe (2006) | 500 | 20 |
| Backwash wetlands | CH2M Hill (2005) | 2.5 | 12 |
| Sacramento | Nolte and Associates (1998) | Secondary | 35.7 |
| Sweden sedimentation basin | Karlsson (2006) | Urban stormwater | 20 |
| 11 lakes, Poland | Samecka-Cymerman and Kempers (2001) | 2.6–7.0 | 0–165[a] |

[a] "0" means below detection limit.

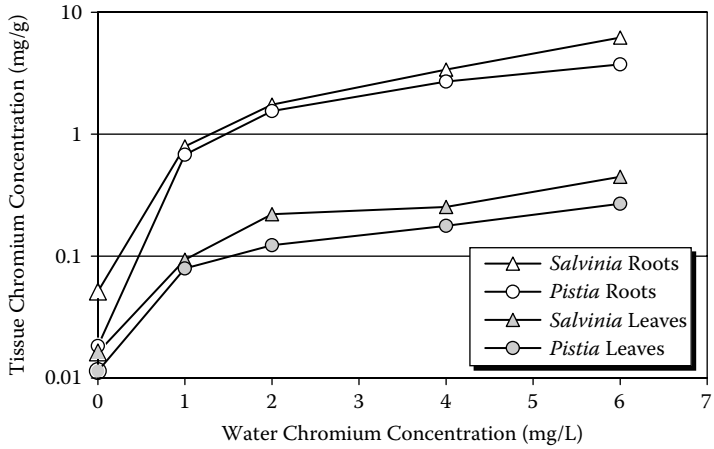



**FIGURE 11.20** Chromium content of leaves of *Salvinia herzogii* and *Pistia stratiotes* after 31 days' exposure to different water-phase concentrations. (Data from Maine *et al.* (2004b) *Water Research* 38(6): 1494–1501.)

### *Chromium Sorption*

The Langmuir isotherm (Equation 11.11) fits sorption data (Azizian and Nelson, 1992; Sharma and Forster, 1993). Azizian and Nelson (1992) found $C_{Smax} = K/a = 12.2$ µg/g for hexavalent chrome on a mineral soil of less than 0.5% carbon, and $K = 2.6$ *L*/kg. However, some Cr(VI) was also reduced to Cr(III). Sharma and Forster (1993) found $C_{Smax} = K/a = 15.3$ mg/g for hexavalent chrome on a peat soil at pH = 6, and $K = 8{,}300$ L/kg. The same data could be equally well described by the Freundlich Equation 11.13, with $K_F = 1.05$ mg/g and $n = 0.393$.

The uptake of chromium by living organisms may also be viewed as biosorption, and described by a Langmuir equation. Gupta *et al.* (2001) reported chromium sorption by green filamentous algae *Spirogyra* spp. They determined $K = 211$ L/kg and $C_{Smax} = 3{,}380$ mg/kg at pH = 5.

## Performance of Wetlands for Chromium Removal

Chromium is removed in wetlands at elevated levels of inlet concentrations (Table 11.33). For 14 FWS systems, with inlet lead ranging from 0.8–410 µg Cr/L, the median concentration reduction is 68%. Removal rates are positively correlated with inlet concentrations, with increasing removals as a response to increasing inlet concentrations (log–log $R^2 = 0.83$). Removal rates range from negative values, corresponding to release of previously stored chromium, to 16 kg/ha·yr.

The disappearance data for Cr(III) in Table 11.33 leads to estimates of first-order removal rate constants in the range 3–64 m/yr, with a median of 19 m/yr (N = 10). Experiments with dosed mesocosms yielded an areal *k*-value of 25 to 35 m/yr in a FWS/SSF system (Sinicrope *et al.*, 1992). FWS microcosm data indicate values of $20 < k < 80$ m/yr (Kadlec and Srinivasan, 1995) for Cr(VI).

Results from HSSF systems (Table 11.33) indicate that chromium is released from the systems with the increase in outflow concentrations as compared to inflow in the range from –138 to +19%. On the other hand, VF systems exhibited either positive removal or no removal (Table 11.33). These results are in a good agreement with the knowledge on chromium chemistry in wetlands. Reduction of Cr(VI) occurs at approximately the same redox levels as nitrate reduction. DeLaune *et al.* (1998) reported that in their experiments, Cr(VI) reduction was complete at 300 mV. The solubility of chromium is strongly dependent upon its oxidation state, Cr(VI) being more soluble than Cr(III). Cr(III), like other cationic metals, is rapidly adsorbed by soil Fe and Mn oxides and clay minerals (Schroeder and Lee, 1975; Bartlett and Kimble, 1976; Dreiss, 1986). Cr(VI), even though not as strongly as Cr(III), is also adsorbed by soil Mn, Al, and Fe oxides and clay and mineral colloids (Davis and Leckie, 1980).

Treatment wetlands have been implemented for the purpose of removing chromium from contaminated waters from leather tanning (Bragato *et al.*, 2004), electroplating (Ranieri, 2004), and metallurgical processing (Maine *et al.*, 2004a). However, chromium is also a trace contaminant in landfill leachates and urban runoff, and hence has been the co-target of treatment wetlands built for treatment of those waters.

## Example Treatment Wetlands for Chromium Removal

### *San Tomé, Argentina* (*Maine et al., 2004a*)

A wetland was constructed at the Bahco Argentina metallurgical plant for the purpose of treating a combination of metallurgical and domestic wastewaters. The 2,000-$m^2$ FWS system received about 100 $m^3$/d. The wetland contained primarily *Eichhornia crassipes, Pontederia cordata,* and *Typha domingensis*, and operated at depths of 0.5–0.8 m and nominal detention of 7–12 days. Water, plants, and sediments were sampled twice monthly, and analyzed for chromium, iron, nickel, and zinc. Chromium was reduced by 86% from inlet to outlet, from 22 to 3.3 µg/L over the 16-month study period. The final sediment content was 25 µg/g at the inlet and 21 µg/g at the outlet, with a slight decrease from 30 µg/g before treatment began. The leaves of both *Eichhornia crassipes* and *Typha domingensis* remained at low chromium content,

## TABLE 11.33
## Chromium Reduction in Treatment Wetlands

| System | Notes | Concentration (µg/L) | | | Mass Rates (kg/ha·yr) | | | | Reference |
|---|---|---|---|---|---|---|---|---|---|
| | | In | Out | Reduction (%) | In | Out | Removal | Removal (%) | |
| **Industrial Water** | | | | | | | | | |
| San Tomé, Argentina | Metallurgical plant | 22 | 3 | 85 | 5.46 | 0.82 | 4.64 | 85 | Maine *et al.* (2004a) |
| Chiampo Valley, Italy | Tannery, *Phragmites* | 410 | 361 | 12 | 95.78 | 84.38 | 11.40 | 12 | Bragato *et al.* (2004) |
| Chiampo Valley, Italy | Tannery, *Carex* | 410 | 356 | 13 | 95.78 | 83.23 | 12.55 | 13 | Bragato *et al.* (2004) |
| Putignano, Italy SSF | Electroplating | 84 | 24 | 72 | 23.00 | 6.44 | 16.56 | 72 | Ranieri (2004) |
| Mandan, North Dakota | Oil refinery | 100 | 12 | 88 | — | — | — | — | Litchfield (1993) |
| **Landfill Leachate** | | | | | | | | | |
| New Hanover County, NC | Landfill leachate | 36 | 5 | 86 | — | — | — | — | Unpublished data |
| Monroe County, New York | Landfill leachate | 13 | 9 | 31 | — | — | — | — | Eckhardt *et al.* (1999) |
| Isanti-Chisago, Minnesota | Landfill leachate | 0.77 | 0.73 | 5 | — | — | — | — | Kadlec (2003c) |
| **Urban Stormwater** | | | | | | | | | |
| Sanford, Florida | Urban stormwater | 3 | — | — | 0.23 | 0.06 | 0.167 | 73 | Harper *et al.* (1986) |
| **Municipal Wastewater** | | | | | | | | | |
| Sacramento, California | FWS | 1.43 | 1.11 | 22 | 0.31 | 0.24 | 0.070 | 22 | Nolte and Associates (1998b) |
| Phoenix, Arizona | Tres Rios Hayfield 1 | 5.00 | 7.00 | –40 | 2.19 | 3.07 | –0.876 | –40 | Wass, Gerke, and Associates (2002) |
| Phoenix, Arizona | Tres Rios Cobble 2 | 1.60 | 0.50 | 69 | 0.88 | 0.27 | 0.602 | 69 | Wass, Gerke, and Associates (2002) |
| Brookhaven, New York | FWS | 160 | 20.0 | 88 | 9.1 | 1.1 | 7.9 | 88 | Hendrey *et al.* (1979) |
| San Diego, California | FWS/SSF | 100 | 16 | 84 | 17.96 | 2.87 | 15.08 | 84 | Sinicrope *et al.* (1992) |
| San Diego, California | FWS/SSF | 100 | 32 | 68 | 22.30 | 7.14 | 15.17 | 68 | Sinicrope *et al.* (1992) |
| Mořina, Czech Republic | HSSF/sewage | 7.0 | 8.3 | –19 | 0.65 | 0.77 | 0.12 | –19 | Unpublished data |
| Břehov, Czech Republic | HSSF/sewage | 8.9 | 11 | –24 | 2.2 | 2.7 | 0.5 | –24 | Unpublished data |
| Slavošovice, Czech Republic | HSSF/sewage | 4.0 | 9.5 | –138 | 0.014 | 0.020 | 0.006 | –138 | Unpublished data |
| Wapserveen 1, The Netherlands | VSSF/sewage + dairy | 5.0 | 4.0 | 20 | 0.03 | 0.024 | 0.006 | 20 | Unpublished data |
| Wapserveen 2, The Netherlands | VSSF/sewage | 9.0 | 3.0 | 67 | 0.83 | 0.28 | 0.55 | 67 | Unpublished data |
| Bokrijk, Belgium | VSSF/sewage | 9.0 | 9.0 | 0 | 1.13 | 1.13 | 0 | 0 | Unpublished data |
| Zemst-Larbeek, Belgium | VSSF/sewage | 7.0 | 7.0 | 0 | — | — | — | — | Unpublished data |
| Sint-Niklaas-Heimolen, Belgium | VSSF/sewage | 5 | 5 | 0 | — | — | — | — | Unpublished data |

starting at 12 and 13 µg/g, respectively, and ending at 9 and 12 µg/g. On the other hand, roots increased markedly in chromium. *Eichhornia crassipes* increased from 16 to 78 µg/g, and *Typha domingensis* from 20 to 139 µg/g. The authors speculate that the observed shift in vegetation from *Eichhornia crassipes* to *Typha domingensis* was beneficial, which would lead to increased sediment storage.

## Zinc

Zinc is an essential element for both plants and animals. It activates some enzymatic reactions important in respiration and serves as a cofactor in plant photosynthesis and DNA synthesis. It is present in stormwaters due to runoff from galvanized roofs, and in the leachates from lead-zinc mines and tailings. However, it is typically present only at very low concentrations in municipal wastewaters. Usually, zinc is present in surface waters in particulate forms, with very little ionic Zn(II). The ratio of free ionic to total zinc is often less than 1%. For example, Westerstrand *et al.* (2006) found 98.9–99.9% particulate zinc in several Swedish stormwaters. Often, smaller particles, including colloidal materials, possess the highest concentrations, but may form the minority of the mass (Karlsson, 2006).

U.S. EPA (2002a) sets criteria for freshwater concentrations of zinc according to hardness. At hardness of 100 mg/L, the maximum concentration allowable and the continuous concentration allowable are both 120 µg/L, as dissolved zinc. U.S. EPA (2002a) provides formulae for the limits at other hardness values, with soft waters having lower criteria. The maximum level for human ingestion is 7.4 mg/L.

Sediment standards for zinc have been set by the province of Ontario, and adopted in some of the states in the United States. These set a lowest effect level of 120 µg/g, and a severe effects level of 820 µg/g (Persaud *et al.*, 1993; OMOEE, 1994). OMOEE (1994) has also identified the level for presettlement deposits as 65 µg/g in Great Lakes sediments. The state of Wisconsin has a guideline for a probable effects sediment concentration of 460 µg/g. Zinc is toxic to benthic macroinvertebrates. At concentrations of 230–460 µg/g, there is 22% mortality of the test organism *Chironomus tentans* in 10–14 days, and 60% mortality of the test organism *Hyalella azteca* (Wisconsin Department of Natural Resources, 2003).

### Wetland Processing and Storage of Zinc

Wetland plants have trace amounts of zinc in their tissues, typically with higher values for roots and rhizomes than for aboveground plant parts (Table 11.34). Plant shoots contain 10–30 µg/g dry weight for nonmine water situations, but concentrations can become elevated in wetland plant tissues in response to exposure to very high concentrations of zinc (Figure 11.21) (McCabe *et al.*, 2001; Manios *et al.*, 2003).

Wetland sediments have the potential to contain large amounts of zinc, in response to chemical precipitate formation and cation exchange with zinc in the water. In most situations, that capacity is not realized, because of the low concentrations in the wetland waters. For instance, secondary incoming waters at Sacramento, California, were at 27–49 µg/L, and the wetland sediments had 12–38 µg/g (Nolte and Associates, 1998b). In contrast, wetland water at the Guangdong, China, lead-zinc mine site were as high as 2,000 µg/L, and the corresponding sediment concentrations were 7,000 µg/g (Lan *et al.*, 1990).

Zinc was transferred primarily to soils (80%) and fine roots (17%) in the Sinicrope *et al.* (1992) study. Likewise, Vymazal and Krása (2005) found 95% of the stored zinc was belowground in the Nučice, Czech Republic, treatment wetland. Hansel *et al.* (2002) found that the belowground storages were 3.0% in roots, 97.0% in soil for *Typha latifolia*, and 8.5% in roots, 91.5% in soil for *Phalaris arundinacea,* in the Cataldo wetland in Idaho, which receives metal mine drainage.

As for other metals, the storage of zinc in wetlands entails a risk of creating potentially impaired sediments. If the sediment concentration is to be kept below some limiting level, such as the 820-µg Zn/g severe effects level of the Ontario guideline, Equation 11.22 can be solved for the allowable removal flux. For that Ontario guideline, deposition of 164 mg $Zn/m^2$·yr could be sustainably tolerated. Meeting rigorous sediment standards on a sustainable basis is not easily achieved for any but trace concentrations, such as those occurring in stormwater and municipal wastewater (Table 11.35).

#### *Zinc Removal Processes*

Zinc is removed by particulate settling and trapping, chemical precipitation and co-precipitation, partitioning to sediments, and to the least extent by plant uptake.

#### *Precipitation*

Zinc forms very insoluble compounds with sulfide and carbonate, among others:

$$S^{2-} + Zn^{2+} \rightarrow ZnS \tag{11.46}$$

$$CO_3^{2-} + Zn^{2+} \rightarrow ZnCO_3 \tag{11.47}$$

Hydrated forms of the carbonate, such as hydrozincite, also have been identified on wetland plant roots, and zinc may substitute in calcium carbonates (Hansel *et al.*, 2002; Roberts *et al.*, 2003). Sulfide and bisulfide are formed by SRB in the anaerobic zones of treatment wetlands. If a source of sulfate is present in the incoming water, the wetland can be configured to provide a sustainable supply of sulfide (Sobolewski, 1996). However, high zinc waters are not amenable to removal via sulfide formation, because ZnS does not readily precipitate at neutral pH (Younger, 2000). Carbonates may be available from atmospheric aeration, and from carbonate minerals in the wetland substrate. Therefore aerated limestone beds may be used to remove zinc (Younger, 2000).

The removal of zinc can also take place through co-precipitation with iron, manganese, and aluminum oxyhydroxides (Roberts *et al.*, 2003). However, this mechanism is contingent on considerable supplies of the secondary metals, which may not be present. Additionally, although iron and manganese

## TABLE 11.34
## Zinc Content of Wetland Plants and Sediments

| Wetland Plant and Part | Reference | Water[a] (µg/L) | Solid (µg/g) |
|---|---|---|---|
| **Plant Shoots** | | | |
| *Scirpus acutus* | Behrends *et al.* (1996) | 600 | 19 |
| *Scirpus acutus* | Nolte and Associates. (1998b) | 36 | 10.0 |
| *Scirpus atovirens* | Behrends *et al.* (1996) | 600 | 10 |
| *Scirpus cyperinus* | Behrends *et al.* (1996) | 600 | 11 |
| *Scirpus lacustris* | Samecka-Cymerman and Kempers (2001) | 94 | 20 |
| *Scirpus* spp. | Karpiscak *et al.* (2001a) | 67 | 14.3 |
| *Juncus effusus* | Samecka-Cymerman and Kempers (2001) | 272 | 15 |
| *Juncus* spp. | Chague-Goff (2005) | 18 | 14.0 |
| *Phalaris arundinacea* | Behrends *et al.* (1996) | 600 | 20 |
| *Phalaris arundinacea* | Vymazal and Krása (2005) | 198 | 23.9 |
| *Phalaris arundinacea* | Vymazal *et al.* (2006) | Sewage | 16.8 |
| *Phalaris arundinacea* | Samecka-Cymerman and Kempers (2001) | 311 | 20 |
| *Phragmites australis* | Vymazal *et al.* (2006) | Sewage | 12 |
| *Phragmites australis* | Behrends *et al.* (1996) | 600 | 28 |
| *Phragmites australis* | Vymazal and Krása (2005) | 198 | 28.7 |
| *Phragmites australis* | Samecka-Cymerman and Kempers (2001) | 311 | 23.6 |
| *Typha* spp. | Behrends *et al.* (1996) | 600 | 12 |
| *Typha latifolia* | Obarska-Pempkowiak *et al.* (2005) | Sewage | 10.9 |
| *Typha latifolia* | Maddison *et al.* (2005) | Sewage | 14.5 |
| *Typha latifolia* | Paredes *et al.* (2006) | 1,500 | 96 |
| *Typha domingensis* | Nolte and Associates (1998b) | 36 | 11.3 |
| *Typha domingensis* | Karpiscak *et al.* (2001a) | 67 | 3.5 |
| *Typha angustifolia* | Samecka-Cymerman and Kempers (2001) | 94 | 14 |
| *Glyceria maxima* | Obarska-Pempkowiak *et al.* (2005) | Sewage | 12.3 |
| *Anemopsis californicus* | Karpiscak *et al.* (2001a) | 67 | 15.0 |
| *Lythrum hyssopifolia* | Chague-Goff (2005) | 18 | 49 |
| *Typha latifolia* | Manios *et al.* (2003) | Sludge | 34–61 |
| *Typha latifolia* | Zhang *et al.* (1990) | 137 | 38.0 |
| **Plant Roots** | | | |
| *Scirpus acutus* | Behrends *et al.* (1996) | 600 | 23 |
| *Scirpus atovirens* | Behrends *et al.* (1996) | 600 | 17 |
| *Scirpus cyperinus* | Behrends *et al.* (1996) | 600 | 14 |
| *Scirpus* spp. | Karpiscak *et al.* (2001a) | 67 | 32.6 |
| *Juncus* spp. | Chague-Goff (2005) | 18 | 23.0 |
| *Phalaris arundinacea* | Behrends *et al.* (1996) | 600 | 48 |
| *Phalaris arundinacea* | Vymazal *et al.* (2006) | Sewage | 65 |
| *Phragmites australis* | Vymazal *et al.* (2006) | Sewage | 85 |
| *Phragmites australis* | Behrends *et al.* (1996) | 600 | 62 |
| *Typha* spp. | Behrends *et al.* (1996) | 600 | 38 |
| *Typha domingensis* | Karpiscak *et al.* (2001a) | 67 | 49.4 |
| *Glyceria maxima* | Obarska-Pempkowiak *et al.* (2005) | Sewage | 25.2 |
| *Anemopsis californicus* | Karpiscak *et al.* (2001a) | 67 | 23.7 |
| *Lythrum hyssopifolia* | Chague-Goff (2005) | 18 | 86.0 |
| *Typha latifolia* | Manios *et al.* (2003) | Sludge | 293–392 |
| *Typha latifolia* | Zhang *et al.* (1990) | 137 | 170.0 |
| *Typha latifolia* | Paredes *et al.* (2006) | 1,500 | 835 |
| *Typha latifolia*[b] | Maddison *et al.* (2005) | Sewage | 181 |
| **Plant Rhizomes** | | | |
| *Scirpus acutus* | Nolte and Associates (1998b) | 36 | 36.2 |
| *Phalaris arundinacea* | Vymazal *et al.* (2006) | Sewage | 24 |
| *Phragmites australis* | Vymazal *et al.* (2006) | Sewage | 24 |
| *Phragmites australis* | Behrends *et al.* (1996) | 600 | 23 |
| *Typha* spp. | Behrends *et al.* (1996) | 600 | 19 |
| *Typha domingensis* | Nolte and Associates (1998b) | 36 | 30.8 |
| *Typha latifolia* | Zhang *et al.* (1990) | 137 | 46.0 |

(*Continued*)

**TABLE 11.34 (CONTINUED)**
**Zinc Content of Wetland Plants and Sediments**

| Wetland Plant and Part | Reference | Water (µg/L) | Solid (µg/g) |
|---|---|---|---|
| **Sediments** | | | |
| System C | Pontier *et al.* (2001) | Urban | 252 |
| System H | Pontier *et al.* (2001) | Urban | 232 |
| Panel | Goulet and Pick (2001) | Urban | 20 |
| Monahan | Goulet and Pick (2001) | Urban | 15 |
| Falconbridge | Goulet and Pick (2001) | Urban | 31 |
| Riverwalk | Goulet and Pick (2001) | Urban | 10 |
| 11 lakes, Poland | Samecka-Cymerman and Kempers (2001) | 86–751 | 3–131 |
| Sacramento | Nolte and Associates (1998) | Secondary | 23 |
| Christchurch, New Zealand | Chague-Goff (2005) | Lagoons | 75 |
| Bielkowo, Poland | Obarska-Pempkowiak *et al.* (2005) | Sewage | 196 |
| *Typha latifolia* | Zhang *et al.* (1990) | 137 | 800 |
| *Typha latifolia* | Lan *et al.* (1990) | 1,930 | 6,863 |

[a] Water concentrations are approximate
[b] Roots and rhizomes

oxides are generally excellent scavengers for other metals, they are unstable under anoxic conditions (Knox *et al.*, 2004).

### *Zinc Sorption*

It is likely that zinc sorption to organic sediments follows a Langmuir isotherm, but water concentrations would presumably usually fall in the linear range, and hence simple proportionality is used (Equation 11.12).

Partition coefficients for zinc to aquatic sediments vary considerably. Wepener *et al.* (2000) found a range of $K_D$ from 134 to 542 L/g at river sites in South Africa. Koelmans and Radovanovic (1988) determined that most zinc partitioned to iron and manganese oxyhydroxides in lakes in the Netherlands, and measured $K_D$ of about 54 L/g. Ringqvist and Öborn (2002) determined $K_D$ = 10.8 L/g for two peats. Roberts *et al.* (2003) determined that zinc partitioned very strongly to aluminum hydroxides and silica, with essentially 100% on the solids at pH > 7.5. Algal necromass was also found to be an effective sorbent (Pascucci and Kowalak, 1999).

## Performance of Wetlands for Zinc Removal

Zinc is removed in wetlands when the incoming amounts are greater than background (Table 11.35). For 26 FWS systems, with inlet zinc ranging from 18–12,000 µg Zn/L, the median concentration reduction is 68%. Removal rates are positively correlated with inlet concentrations, with increasing removals as a response to increasing inlet concentrations (log–log $R^2 = 0.91$). A constant load reduction of 7 g/m²·d (25,550 kg/ha·yr) is the basis reported for use in design by Younger *et al.* (2002), which is much higher performance than any system in Table 11.35. The design value of 0.04 g Zn/m²·d (146 kg/ha·yr) was tentatively advanced by the PIRAMID Project of the European Union (PIRAMID Consortium, 2003a) for aerobic wetlands.

HSSF systems remove zinc with high efficiency (Table 11.35). Median removal for four systems in the Czech

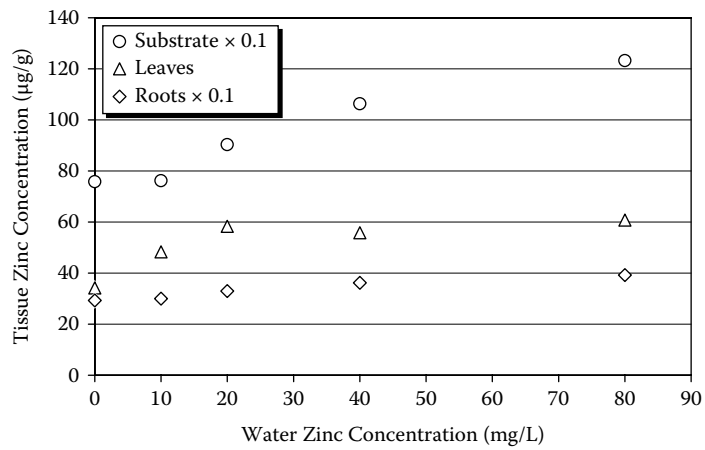



**FIGURE 11.21** The response of *Typha latifolia* tissues lead to increasing amounts of dissolved zinc in the overlying water. (Data from Manios *et al.* (2003) *Chemosphere*, 53(5): 487–494.)

## TABLE 11.35
## Zinc Reduction in Treatment Wetlands

| System | Notes | Concentration (µg/L) | | | Mass Rates (kg/ha·yr) | | | | Reference |
|---|---|---|---|---|---|---|---|---|---|
| | | In | Out | Reduction (%) | In | Out | Removal | Removal (%) | |
| **Mine Water** | | | | | | | | | |
| Albright, Pennsylvania | Combustion leachate | 83 | 37 | 55 | 21.94 | 9.78 | 12.16 | 55 | Hoover *et al.* (1998) |
| Woodcutters, NT, Australia | Metal mine water | 1,000 | 23.0 | 98 | — | — | — | — | Noller *et al.* (1994) |
| Springdale, Pennsylvania | Combustion leachate | 89 | 24 | 73 | 31.86 | 8.59 | 23.27 | 73 | Hoover *et al.* (1998) |
| Tom's Gully, NT, Australia | Metal mine water | 6,900 | 260 | 96 | — | — | — | — | Noller *et al.* (1994) |
| Dunka Mine, Minnesota | Metal mine water | 80 | 28 | 65 | — | — | — | — | Eger *et al.* (1993) |
| Orcopampa, Peru | Metal mine water | 11,290 | 757 | 93 | — | — | — | — | Unpublished data |
| Wheal Jane, United Kingdom | Tin/Zinc mine water | 79 | 45 | 43 | — | — | — | — | Younger (2000) |
| Dunka Mine, Minnesota | Metal mine water | 1,400 | 70 | 95 | — | — | — | — | Eger (1994) |
| **Landfill Leachate** | | | | | | | | | |
| New Hanover County, NC | Landfill leachate | 19 | 5 | 74 | — | — | — | — | Unpublished data |
| Monroe County, New York | Landfill leachate | 227 | 80 | 65 | — | — | — | — | Eckhardt *et al.* (1999) |
| Esval, Norway | Landfill leachate | 312 | 59 | 81 | — | — | — | — | Mæhlum *et al.* (1999) |
| **Urban Stormwater** | | | | | | | | | |
| Hidden River, Florida | Urban stormwater | 64 | 19 | 70 | 2.42 | 0.73 | 1.693 | 70 | Carr and Rushton (1995) |
| Greenwood, Florida | Urban stormwater | 147 | 72 | 51 | 13.79 | 6.75 | 7.035 | 51 | McCann and Olson (1994) |
| Sanford, Florida | Urban stormwater | 4 | — | — | 0.44 | 0.26 | 0.180 | 41 | Harper *et al.* (1986) |
| Norco, Louisiana | Refinery stormwater | 566 | 126 | 78 | — | — | — | — | Hawkins *et al.* (1997) |
| Aiken, South Carolina | Lab stormwater | — | — | 60 | — | — | — | — | Nelson *et al.* (2004) |
| Crestwood, Virginia | Urban stormwater | 131 | 96 | 27 | 0.140 | 0.099 | 0.041 | 29 | Godrej *et al.* (1999) |
| Kildare, Ireland | Highway runoff | 81–426 | 12–45 | 85 | — | — | — | — | Higgins *et al.* (2006) |
| **Municipal Wastewater** | | | | | | | | | |
| Estevan, Saskatchewan | FWS lagoon | 33 | 13 | 61 | — | — | — | — | Unpublished data |
| Sacramento, California | FWS | 35.82 | 6.74 | 81 | — | — | — | — | Nolte and Associates (1998) |
| Christchurch, New Zealand | FWS | 17.7 | 4.5 | 75 | — | — | — | — | Chague-Goff (2005) |
| Central Illinois | FAP | 64 | 23 | 64 | — | — | — | — | Jin *et al.* (2003) |
| Central Illinois | SSF | 64 | 51 | 20 | — | — | — | — | Jin *et al.* (2003) |
| Nučice, Czech Republic | HSSF/sewage | 198 | 5 | 97 | 19.6 | 0.5 | 19.1 | 97 | Vymazal and Krása (2003) |
| Mořina, Czech Republic | HSSF/sewage | 158 | 18.6 | 88 | 14.8 | 1.7 | 13.1 | 88 | Unpublished data |
| Břehov, Czech Republic | HSSF/sewage | 98 | 46 | 53 | 24.1 | 11.3 | 12.8 | 53 | Unpublished data |
| Slavošovice, Czech Republic | HSSF/sewage | 82 | 28 | 66 | 8.0 | 2.7 | 5.3 | 66 | Unpublished data |
| Wepserveen, The Netherlands | VSSF/sewage + dairy | 121 | 15 | 88 | 0.66 | 0.08 | 0.58 | 88 | Unpublished data |
| Wepserveen, The Netherlands | VSSF/sewage | 51 | 15 | 71 | 4.62 | 1.4 | 3.22 | 71 | Unpublished data |
| Nijkerk, The Netherlands | VSSF/sewage | 112 | 10 | 91 | 22.5 | 2.1 | 20.4 | 91 | Unpublished data |
| Bokrijk, Belgium | VSSF/sewage | 200 | 56 | 72 | 25.9 | 7.2 | 18.7 | 72 | Unpublished data |
| Zemst-Larbeek, Belgium | VSSF/sewage | 120 | 22 | 82 | — | — | — | — | Unpublished data |
| Sint-Niklaas-Heimolen, Belgium | VSSF/sewage | 319 | 635 | –99 | — | — | — | — | Unpublished data |
| Central Illinois | FWS | 64 | 52 | 18 | — | — | — | — | Jin *et al.* (2003) |
| San Diego, California | FWS/SSF | 399 | 63 | 84 | 54 | 11 | 43 | 79 | Sinicrope *et al.* (1992) |
| San Diego, California | FWS/SSF | 2,500 | 725 | 71 | 558 | 162 | 396 | 71 | Sinicrope *et al.* (1992) |
| Norco, Louisiana | Refinery effluent | 550 | 190 | 65 | 628 | 217 | 411 | 65 | Gillespie *et al.* (2000) |
| Norco, Louisiana | Refinery effluent | 4,150 | 2,570 | 38 | 4,741 | 2,936 | 1,805 | 38 | Gillespie *et al.* (2000) |

Republic treating municipal sewage amounted to 77% and varied between 53–97%. The median removal of zinc in VF systems equaled that for HSSF wetlands, about 77%. However, the removal varied between 91–99%.

The first-order areal removal rate constant for zinc is 28 m/yr for the FWS/HSSF data of Sinicrope *et al.* (1992). This is comparable to the results of Delgado *et al.* (1993) for hyacinths, which give $k = 20$ m/yr. The data of Gillespie *et al.* (2000) produce 55 and 121 m/yr for two different loadings. Mitchell *et al.* (2002) reported $k_V = 6\ d^{-1}$ for zinc in highway runoff wetlands.

### Example Treatment Wetlands for Zinc Removal

*Palmerton Zinc, Pennsylvania* (*Haffner, 1992, 1993*)

The Palmerton Zinc Superfund Site is a former zinc smelting operation located in Palmerton, Pennsylvania. This plant operated from 1898–1980, and caused large quantities of zinc, cadmium, lead, and copper to be emitted into the atmosphere in the vicinity of the plant. These emissions caused significant concentrations of heavy metals in the soil within an area of thousands of acres surrounding the plant, with zinc concentrations up to 3% dry weight. An estimated 30 million tons of smelting residue were deposited in a cinder bank, from which zinc leaches into surface and groundwaters. Constructed wetlands were tested as a means of removing metals from the leachate in 1991–1992. Sixteen mesocosms, 1 × 2 m, were filled with varying mixtures of peat, mushroom compost, and limestone, and planted with cattails (*Typha latifolia*). The cells were loaded at about 8 cm/d of water containing 200 mg/L zinc. Zinc removals of more than 90% were achieved over a six-month period of operation.

*Glendalough, Wicklow, Ireland*
(*Beining and Otte, 1996, 1997*)

The lead-zinc mines in the Glendalough valley operated from about 1820–1950. Drainage from tailings piles is still acting as a source of metals to a marsh situated between the tailings and the Glendalough River. Studies of the water, soils, and vegetation were conducted by University College, Dublin, researchers. Pore-water zinc concentrations decreased from 1,863 µg/L to 85 µg/L along a 500-m transect in the flow direction, while soil concentrations decreased from 16.35 mg/g to 0.78 mg/g. The principal macrophyte species were *Glyceria fluitans*, *Eriophorum angustifolia,* and *Juncus effusus*, and were apparently tolerant of the very high soil zinc levels (Beining and Otte, 1996). The tissue concentrations of zinc reflected the levels of zinc to which they were exposed (Figure 11.21). Given the very long period of operation of the mines, and the existence of a gradient in soil zinc concentrations, Beining and Otte (1997) estimated that only 26% of the capacity of the marsh to trap zinc had been utilized.

## Mercury

Mercury is released from a variety of anthropogenic sources such as solid and medical waste incineration, the combustion of fossil fuels for electric power production, and historically from gold mining processes. The methylated form biomagnifies, and can cause serious effects in top-level consumers, including humans. Wetlands provide an environment that may sometimes trap mercury, and in many cases convert a small fraction to methylmercury.

### Wetland Processing and Storage of Mercury

Mercury may exist in three main forms in aquatic and wetland systems: elemental mercury, divalent ions, and methylated. These interconvert:

$$Hg^{o} \rightleftharpoons Hg(II) \rightleftharpoons MeHg \qquad (11.48)$$

Mercury entering marshes is subject to transformation through a variety of complex biogeochemical processes (Figure 11.22). End points of these processes can result in mercury sequestration in sediments, reduction, and emission back to the atmosphere, or production of methylmercury (MeHg). MeHg is produced from inorganic mercury in anaerobic environments by SRB.

Although inorganic forms of mercury dominate the environmental cycle of mercury, some few percent can be transformed into methylmercury (SFWMD, 2002). Methylmercury is primarily produced by SRB naturally present in the sediment where oxygen is absent, and a sulfur compound (sulfate) is present. These bacteria take up inorganic mercury and manufacture methylmercury as a byproduct of the normal life processes. Studies in freshwater and marine systems have demonstrated that SRB are the major biological contributors of methylmercury formation in wetlands (e.g., SFWMD, 1999b–2005). The respiration activity of SRB has been implicated in the methylation process. Factors that facilitate methylation include the presence of organic matter, low-oxygen sediment, high microbial activity, and water-level fluctuations. Methylation rates also depend on pH, temperature, and oxygen levels. The presence of high levels of sulfide causes the precipitation of insoluble mercury sulfide:

$$Hg(II) + S^{2-} \rightarrow HgS \qquad (11.49)$$

This process effectively prevents methylation. These competing processes lead to the interesting situation in which either too little or too much sulfate inhibits methylation, but at intermediate concentrations, the sulfate promotes methylation. However, other mechanisms can cause methylation in the absence of sulfur or SRB (Loseto *et al.*, 2004).

#### *Mercury Removal Processes*

Methylmercury is readily absorbed, much faster than it is excreted, by microorganisms living in the sediments or overlying water. As these are fed on by other organisms, methylmercury becomes progressively more concentrated at each higher level of the aquatic food web. This process is termed *bioaccumulation*.

Elemental mercury is volatile, and may be lost from wetlands as a vapor. Both evaporation and transpiration are candidate

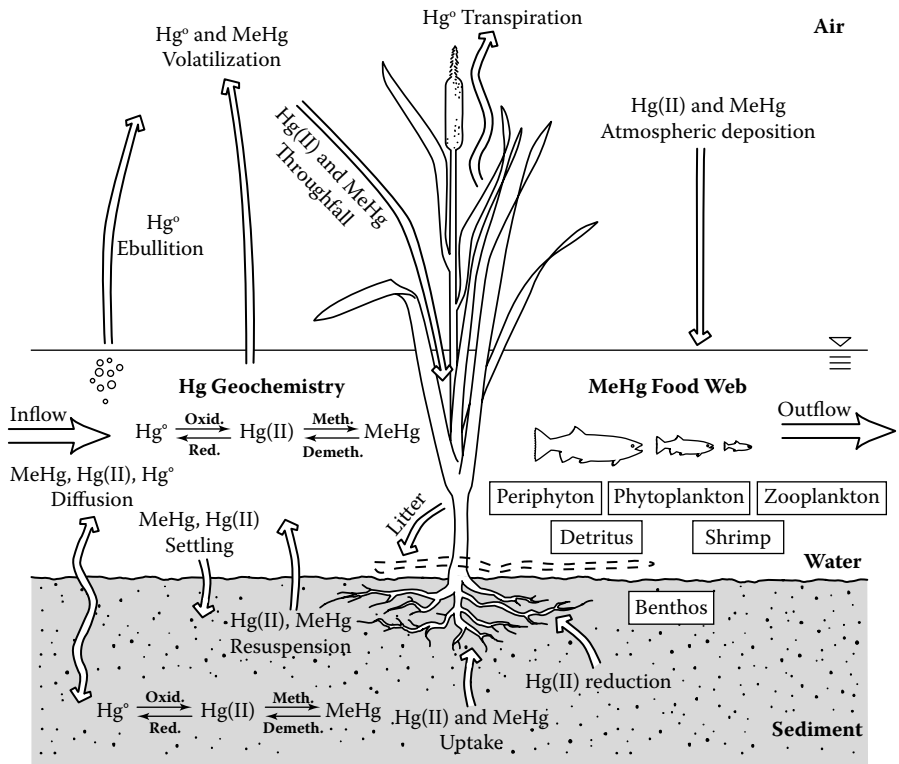



**FIGURE 11.22** Everglades mercury model processes. (Adapted from South Florida Water Management District (2000) Everglades Consolidated Report, South Florida Water Management District, West Palm Beach, Florida.)

processes, but in sum these losses are quite minor. Vaporization of mercury is also known as *evasion* or *ebullition*.

Plant uptake of mercury also occurs. Göthberg *et al.* (2004) studied water spinach (*Ipomea aquatica*), and determined the tissue mercury content under various exposures. In untreated plants, the mercury concentrations in the leaves were significantly higher than in the stems, but lower than in the roots (Table 11.36). In the mercury-treated plants, the roots always contained higher mercury concentrations than the other plant parts. Tissue concentrations increased with increasing mercury in the water. The background total mercury (THg) content of plants is typically 0.05 mg/kg dry mass. However, a concentration of 0.26 mg/kg dry mass was reported for trees in the Okefenokee Swamp and some southeastern states of the United States saltmarsh plants had concentrations as high as 1.13 mg/kg dry mass (Kadlec and Knight, 1996).

**TABLE 11.36**
**Concentrations of Mercury in Various Plant Parts of *Ipomea aquatica* Grown in Hydroponic Solutions**

| Solution (ng/L) | Leaves (mg/kg) | Stems (mg/kg) | Roots (mg/kg) |
|---|---|---|---|
| 0.0 | 0.4 | 0.2 | 0.5 |
| 10.0 | 0.8 | 0.5 | 30.7 |
| 20.1 | 1.0 | 0.7 | 61.9 |
| 40.1 | 2.0 | 2.2 | 147.8 |
| | **Distribution (%)** | | |
| 0.0 | 49.8 | 19.9 | 15.2 |
| 10.0 | 6.8 | 3.9 | 63.8 |
| 20.1 | 4.0 | 2.5 | 71.0 |
| 40.1 | 3.9 | 3.3 | 79.3 |

*Source:* Data from Göthberg *et al.* (2004) *Journal of Environmental Quality,* 33(4): 1247–1255.

### *Mercury in Wetland Sediments*

For headwater and riparian wetland sites of the Adirondack Region, New York, data indicate that retention of THg was greatest at the upland interface, and declined along the hydrologic gradient (Demers *et al.*, 2004). THg retention was greatest in the zone of water table fluctuation, between 15 cm and 35 cm, and declined to nearly zero at depths greater than 1 m. Dissolved THg concentrations were lower in shallow-peat riparian zones than in stream water. However, THg associated with fine particulate matter was an order of magnitude greater in shallow-peat than in stream water.

THg concentrations progressively declined in sediments in sequential preexisting wetlands in the Lahontan Valley of Nevada (Tuttle *et al.*, 2001). A gradient of THg concentrations in sediment was also apparent in constructed wetlands. MeHg concentrations in sediment generally corresponded to THg concentrations in sediment, with MeHg concentrations generally decreasing with distance from wetland inflow. No statistical difference in sediment MeHg concentrations was found between historical and constructed wetlands. Sediment THg concentrations typically ranged from 0.2 to 1.0 mg/kg, while methylmercury was very much lower, in the range of 1–2 μg/kg.

Okefenokee Swamp soils had a concentration of 0.54 mg/kg, and saltmarsh concentrations were as high as 1.7 mg/kg dry mass (Nixon and Lee, 1986). Delfino *et al.* (1993) found average mercury concentrations of about 0.6 mg/kg dry mass. Stable mercury isotope studies by Hintelmann *et al.* (2000) showed the calculated half-life for methylmercury in sediments was 1.7 days, suggesting a rapid turnover and low persistence of methylmercury in lake sediments.

## Performance of Wetlands for Mercury Removal

Mercury may be trapped in wetlands, and accumulate in old and new sediments. Removals range up to about 50% removal in a week in newly constructed wetlands. However, some antecedent soil and ecosystem conditions do not favor such trapping. Most wetlands cause a small fraction of the THg to be methylated.

Mercury is released from a variety of anthropogenic sources such as solid and medical waste incineration, the combustion of fossil fuels for electric power production, and historically from gold mining processes. The methylated form biomagnifies, and can cause serious effects in top-level consumers, including humans. Wetlands provide an environment that may sometimes trap mercury, and in many cases convert a small fraction to methylmercury.

The information on removal of mercury in HSSF and VF systems is quite limited especially because these systems are primarily used to treat domestic and municipal sewage where mercury concentrations are very low. Vymazal (unpublished data) found reduction of mercury concentrations in three HSSF systems in the Czech Republic between 1–65%.

## Example Treatment Wetlands for Mercury Removal

### *Savannah River, South Carolina, Pilot Project*

Pilot-scale wetlands were constructed to determine if a wetland treatment system could effectively remove low-level mercury from an outfall located at the Department of Energy's Savannah River Site (King *et al.*, 2001). These were planted with giant bulrush, *Scirpus californicus,* and surface amended with gypsum ($CaSO_4$). Nominal residence time in each tank was 24 hours with a design residence time for water passing through the first and second pair of tanks of 48 hours. The system was allowed to operate for a period of eight months prior to an examination of the relationship between mercury speciation and the biogeochemical cycling of sulfur in the sediment.

Results compiled over one year demonstrate that mercury was consistently removed from outfall waters passing through the mesocosm cells. Overall, removal efficiencies for both tanks averaged about 50% for the first year of operation. Methylmercury concentrations were collected over the same time period as THg. Overall, inflow concentrations of methylmercury were typically less than the limit of detection (<0.02 ng/L). Initial concentrations of methylmercury in the effluent ranged from 0.081 to 0.241 ng/L for the first three months of operation. However, methylmercury concentrations exceeded 1.0 ng/L after surface applications of gypsum ($CaSO_4$). Methylmercury concentrations in pore water were up to three times higher in the sulfate treatments (0.5 to 1.6 ng/L) than in the control (<0.02 ng/L to 0.5 ng/L). Mean percent methylmercury was 9.0% in the control with 18.5% and 16.6% in the low and high sulfate treatments, respectively. Methylmercury concentrations measured in surface water did not reflect the differences between the control and the sulfate treatments that were noted in pore water, and there were no differences between methylmercury flowing into and out of the systems.

### *Savannah River, South Carolina, Full-Scale Project*

The pilot project at Savannah River was followed by a full-scale FWS project (Nelson *et al.*, 2002). The treatment system consists of four pairs of 0.4-ha wetland cells (3.24 ha total), with water flowing from one cell to a second cell, then to the discharge point. The cells are vegetated with *Scirpus californicus*, and water retention time in the system is about 48 hours. Operation and maintenance of the system was minimal, and mainly consisted of checking for growth of the vegetation and free flow of the water through the system. This has provided a low-cost construction option and low-cost maintenance program for the effective treatment of large volumes of permitted water discharges from an industrial area.

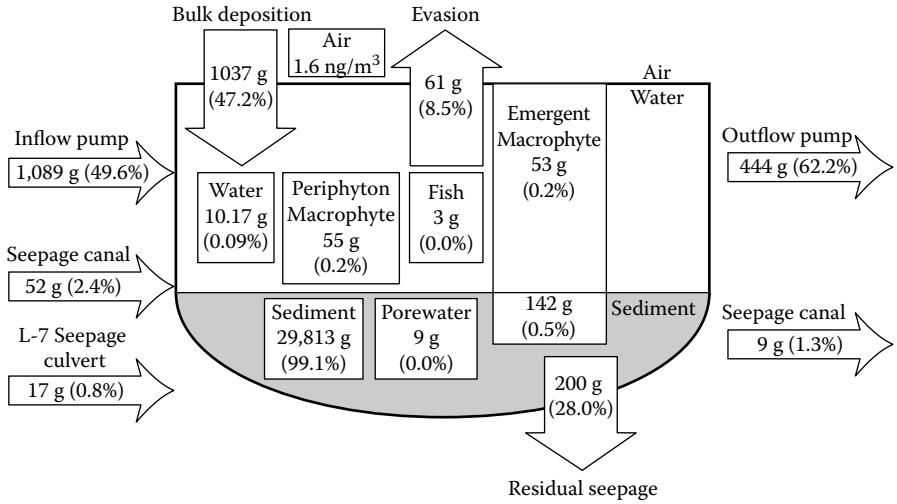
Mercury levels after passage through the first cell are typically reduced to 40% of the influent value, and reduced by an additional 30–40% after passage through the second cell in the series. The final concentration steadily went down as the wetland system matured. During the June 2001 sampling event, the THg level fell below the permit limit (13 ng/L), and has remained below the permit limit since. Performance was better than the anticipated mercury reductions based upon mesocosm testing. Methylmercury production in the system occurred, but was a small component of the THg in the effluent. Methylmercury has increased during the warm summer months, but has never exceeded 10% of the THg at the discharge point.

### *The Everglades Nutrient Removal Project (ENRP), Florida*

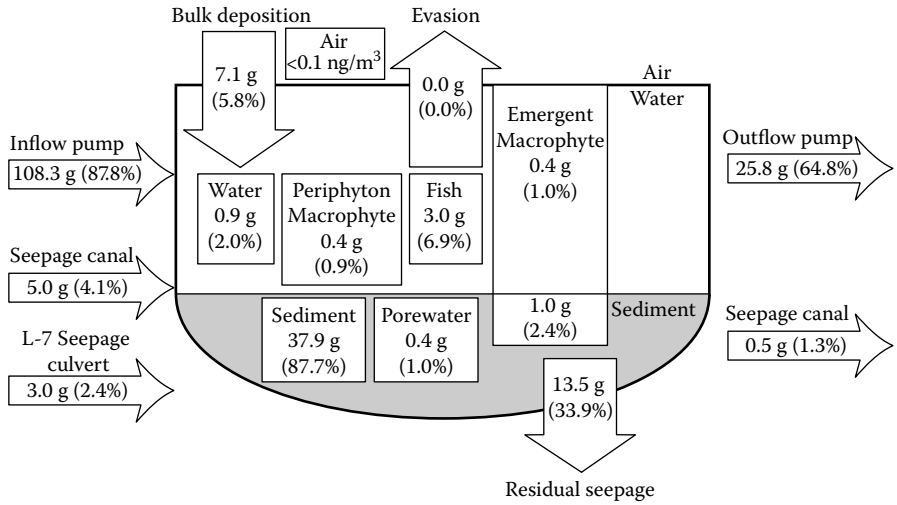
A demonstration free water surface wetland system was constructed to remove phosphorus from agricultural runoff before it entered the Everglades (Miles and Fink, 1998; Fink and Miles, 1999). The 1,545-ha system consisted of five cells, vegetated mostly with cattails (*Typha domingensis*), and operated at about two weeks' detention time. Mass budget estimates showed the inflow pump and rainfall to be the major sources of THg, while the outflow pump was the major output (Figure 11.23). The inflow pump was the major source of MeHg while the outflow pump was the major output. For the three-year period 1994–1997, based upon pumped inflows and outflows, there was 60% removal of THg, and 77% reduction in MeHg. Storages within the wetland ecosystem were overwhelmingly in the sediments: 99% for THg and 88% for MeHg.

### *The Everglades Stormwater Treatment Areas, Florida*

These free water surface constructed wetlands consist of six individual units totaling 17,000 ha (Goforth, 2001). Each of




(a) Unfiltered Total Mercury (3 year total)




(b) Unfiltered Methylmercury (3 year total)

**FIGURE 11.23** Mercury mass balances for the Everglades Nutrient Removal Project, 1994–1997. (From South Florida Water Management District (1999a) *Everglades Consolidated Report.* South Florida Water Management District, West Palm Beach, Florida.)

the six units is compartmentalized, with two to ten cells in one to three flow paths. On an annual average basis, these event-driven wetlands have design detention time of about three weeks, but under peak flow conditions, it is less than four days. Vegetation was allowed to volunteer into these large wetlands, but for some cells, submerged aquatic vegetation was fostered by maintaining deeper water. The result is a mixture of emergent, submergent, and floating plants. Five of the six units have been in operation long enough to have records of performance. Although these treatment wetlands were designed for phosphorus removal, mercury performance has also been monitored. That is because the downstream ecosystem is the Florida Everglades, a wetland complex that is notoriously plagued by high levels of mercury and methylmercury.

Operating data for mercury has not been assembled into a publication, but that data is here given in Table 11.36. On average, for the nine wetlands or flowpaths, there is no significant removal or addition of THg. On closer inspection, it is seen that STA1W exhibited good reductions of THg. STA1W had its origins in the ENRP, which it subsumed. Therefore, the observed removals are in part an extension of the performance of the ENRP. This system was built on land that had been in agriculture. In contrast, STA2 Cell 1 was a net exporter of THg, by a factor of three to one for outflow compared to inflow. This system was built on land that had not been in agriculture, and displayed a replacement of woody shrubs by soft-tissue emergent plants. For other wetlands, there is essentially no difference between inflow and outflow THg concentrations.

Methylmercury is on average higher in the STA outflows than it is in the inflows (0.54 ng/L versus 0.18 ng/L). MeHg comprised 4–21% of the inlet THg (mean 11%), but formed 12–50% of the outlet THg (mean 26%). Thus in the majority of cases, methylation was occurring in the STAs.

In connection with these Everglades wetland system studies, a very large amount of companion research has been conducted, which is described in a series of reports (e.g., SFWMD, 1999–2005). Included is the development and calibration of a detailed process model, the Everglades Mercury Cycling Model (E-MCM).

*Sacramento, California*

Free-water surface constructed wetlands were investigated at Sacramento, California, for the reduction of mercury in secondary municipal effluent (Crites *et al.*, 1997). The 8.9-ha system consisted of ten cells, vegetated with bulrushes (*Scirpus acutus*), and operated at about one week's detention time. Inlet mercury of 10.61 ng/L was reduced on average to 4.71 ng/L. Mercury in soils and vegetation was below detection limits.

## 11.8 METALLOIDS

### Arsenic

Arsenic is a naturally occurring element in surface waters, occurring as As (III) and As (V). Arsenic is considered a metalloid because it behaves both as a metal and as a nonmetal. The arsenate form predominates in aerobic waters, existing in anionic forms of either $H_2AsO_4^-$, $HAsO_4^{2-}$, or $AsO_4^{3-}$ at circumneutral pH. The chemistry of As(V) resembles that of phosphorus. Arsenite occurs under reducing conditions, as arsenous acid ($H_3AsO_3$). In the presence of sulfides, precipitation of AsS (realgar) or $As_2S_3$ (orpiment) may remove soluble $As^{3+}$ (Jiang, 2001). However, the several thioarsenic species dominate arsenic speciation at sulfide concentrations greater than about 4 mg/L at neutral pH in iron-limited environments (Wilkin *et al.*, 2003).

At higher soil redox levels (+200 to +500 mV), As(V) is the predominant species. The reduction of As(V) to As(III) occurs at redox levels corresponding within the nitrate reducing zone of soils characterized by a soil redox level of 200–300 mV (DeLaune *et al.*, 1998). As(III) is more soluble than As(V), but slow kinetics of the As(V) to As(III) reduction coupled with the release of high concentrations of Mn upon reduction made the precipitation of a $Mn_3(AsO_4)_2$ phase under reduced conditions a likely event (Masscheleyn *et al.*, 1991a; Pardue and Patrick, 1995).

A large portion of industrial arsenic in the United States is currently used as a wood preservative, but arsenic is also used in paints, dyes, metals, drugs, soaps, and semi-conductors. Arsenic is found in organometallic forms (Frank, 2005), such as methylarsonic acid ($H_2AsO_3CH_3$) and dimethylarsinic acid [$(CH_3)_2AsO_2H$], which are active ingredients in pesticides. Arsenic was a widely used pesticide prior to the introduction of dichloro-diphenyl-trichloroethane (DDT) (Knox *et al.*, 1999). Chromate-copper-arsenate (CCA) is a common wood preservative, used in bridges and in boardwalks through wetlands. Such bridging has been studied, and found to create small increases in near-field sediment concentrations of copper, chromium, and arsenic, but no biological effects were found (Brooks, 2000b). Industry activities such as copper smelting, mining, and coal burning also contribute to arsenic in our environment.

Arsenic is of concern in aqueous environments because it is a known human carcinogen and because it is acutely and chronically toxic to aquatic organisms. U.S. EPA has set the arsenic standard for drinking water at 10 µg/L to protect consumers served by public water systems from the effects of long-term, chronic exposure to arsenic. There are geographic "hot spots" in the United States that have higher levels of arsenic in well water, resulting from natural underground sources. In the Ron Phibun district of Thailand, mine tailing leachates have produced up to 1,000 µg/L, and caused arsenic poisoning in the local population (Aksorn and Visoottiviseth, 2004).

#### Wetland Processing and Storage of Arsenic

The arsenic content of some typical wetland plants is given in Table 11.38. In general, tissue concentrations are low when water concentrations are low. When the water contains elevated arsenic, some plants respond by accumulating arsenic, notably *Lemna* spp. (Figure 11.24). Such bioaccumulation makes harvest an inviting option for removal, but harvest has not in general been successful in treatment wetlands, including those with *Lemna* spp.

The arsenic content of peat from Finnish bogs was found to be in the range 0.1–8.0 µg/g (Frank, 2005). This bog arsenic

**TABLE 11.37**
**Mercury Entering and Leaving the Various Stormwater Treatment Areas in Florida**

| System | Period | Total Hg Inlet (ng/L) | Total Hg Outlet (ng/L) | Methyl Hg Inlet (ng/L) | Methyl Hg Outlet (ng/L) |
|---|---|---|---|---|---|
| STA1W | 1999–2005 | 4.20 | 0.84 | 0.18 | 0.10 |
| STA2 Cell 1 | 2000–2005 | 1.38 | 4.34 | 0.18 | 2.18 |
| STA2 Cell 2 | 2000–2005 | 1.38 | 1.56 | 0.18 | 0.43 |
| STA2 Cell 3 | 2000–2005 | 1.38 | 1.00 | 0.18 | 0.17 |
| STA3/4 Cell 1 | 2004–2005 | 1.52 | 1.47 | 0.09 | 0.30 |
| STA3/4 Cell 1 | 2004–2005 | 2.17 | 1.33 | 0.13 | 0.40 |
| STA3/4 Cell 1 | 2004–2005 | 1.51 | 1.73 | 0.12 | 0.37 |
| STA5 | 1999–2005 | 1.92 | 1.49 | 0.27 | 0.28 |
| STA6 | 1998–2005 | 1.47 | 1.76 | 0.31 | 0.59 |
| **Mean** | | **1.88** | **1.72** | **0.18** | **0.54** |

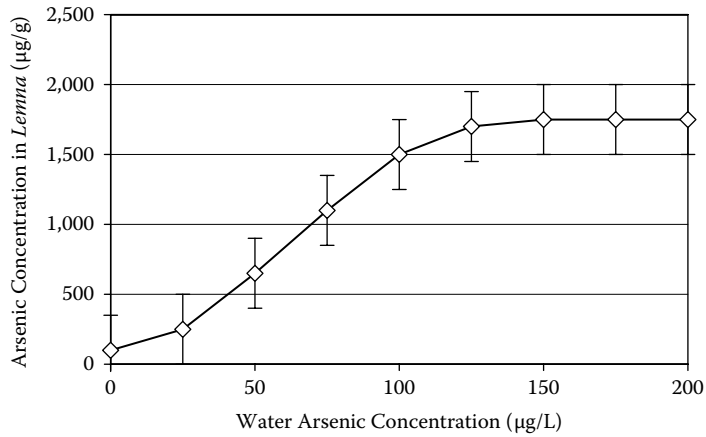



**FIGURE 11.24** General trend of arsenic in duckweed at two mine tailings sites, at Lengenfeld and Neuenalz-Mechelgrün mines in Germany. Error bars are approximate indications of data scatter. (Data from Mkandawire and Dudel (2005) *Science of the Total Environment,* 336: 81–89.)

is mostly of atmospheric origin, because there are not other sources. However, wetlands impacted by mining and industrial sources have much higher levels of arsenic in their sediments (Table 11.38).

The speciation of arsenic in wetland sediments is complex. Keon *et al*. (2001) tested an eight-step procedure on arsenic-contaminated peats from the Woburn, Massachusetts, mine tailings wetland. Very little of the arsenic was ionically bound (4%), but the majority was organically bound (56%) and could be removed by phosphate exchange. Another major fraction was associated with acid-volatile sulfides and oxides of iron and manganese (30%). Only 10% was associated with pyrites and silicates, or present as orpiment. Fox and Doner (2003) studied arsenic speciation in the wetland sediments at the Corcoran, California, agricultural drainage site. They found the following approximate proportions: waters soluble (12%), adsorbed (29%), carbonate-bound (34%), organically bound (8%), and residual (17%).

### *Arsenic Removal Processes*

The adsorption of As(V) and As(III) on various Fe(III) and Al hydroxides has been investigated extensively employing both macroscopic and spectroscopic experiments. Iron oxides are a good sorbent for As(V) (Katsoyiannis and Zouboulis, 2004). The solid phase speciation of As(V) on various Fe and Al (hydr)oxides has been extensively characterized. Arsenate adsorbs via a ligand exchange reaction replacing surface

**TABLE 11.38**
**Arsenic Content of Example Wetland Plants**

| Location | Wetland | Plant | Water (µg/L) | Above (µg/g) | Below (µg/g) | Reference |
|---|---|---|---|---|---|---|
| Sacramento, California | FWS CW | *Scirpus acutus* | 2.4 | 1.29 | 4.21 | Nolte and Associates (1998b) |
| Oxnard, California | FWS CW SAV | *Potemogeton pectinatus* | 2.5 | 2.30 | 0.49 | CH2M Hill (2005) |
| Oxnard, California | FWS CW | *Typha latifolia* | 2.5 | 0.82 | 1.47 | CH2M Hill (2005) |
| Oxnard, California | FWS CW | *Scirpus americanus* | 2.5 | 1.70 | 1.40 | CH2M Hill (2005) |
| Oxnard, California | VSSF CW | *Juncus balticus* | 2.5 | 4.50 | 3.00 | CH2M Hill (2005) |
| Lengenfeld, Germany | FAP | *Lemna* spp. | 3 | 30 | — | Mkandawire and Dudel (2005) |
| Lengenfeld, Germany | FAP | *Lemna* spp. | 265 | 1,540 | — | Mkandawire and Dudel (2005) |
| Wheal Jane, United Kingdom | Hybrid | *Phragmites australis* | 2.5 | 0.02 | 0.99 | Barley *et al.* (2005) |
| Bangkok, Thailand | Hydroponic | *Typha latifolia* | 1,000 | 5.19 | 79.94 | Aksorn and Visoottiviseth (2004) |
| Bangkok, Thailand | Hydroponic | *Canna* spp. | 1,000 | 11.49 | 58.89 | Aksorn and Visoottiviseth (2004) |
| Bangkok, Thailand | Hydroponic | *Colocasia esculenta* | 1,000 | 3.33 | 194.87 | Aksorn and Visoottiviseth (2004) |
| Bangkok, Thailand | Hydroponic | *Heliconia spittacorum* | 1,000 | 7.20 | 82.81 | Aksorn and Visoottiviseth (2004) |
| Bangkok, Thailand | Hydroponic | *Thalia dealbato* | 1,000 | 7.03 | 122.90 | Aksorn and Visoottiviseth (2004) |
| Laboratory | FWS | *Juncus effusus* | 500 | 0.07 | 3.53 | Buddhawong *et al.* (2005) |
| Laboratory | SSF | *Juncus effusus* | 500 | 0.15 | 7.20 | Buddhawong *et al.* (2005) |
| Laboratory | Hydroponic | *Juncus effusus* | 500 | 0.15 | 0.33 | Buddhawong *et al.* (2005) |
| Břehov, Czech Republic | HSSF | *Phragmites australis* | Sewage | 0.26 | 3.2 | Vymazal *et al.* (2006) |
| Břehov, Czech Republic | HSSF | *Phalaris arundinacea* | Sewage | 0.26 | 3.2 | Vymazal *et al.* (2006) |
| Mesocosm, Mexico | HSSF | *Schoenoplectus americanus* | 1,000 | — | 52 | Alacrón-Herrera *et al.* (2006) |

*Note:* FAP = floating aquatic plant.

$OH^-$ groups (Williams *et al*., 2003), for instance,

$$FeOH + AsO_4^{3-} + 2H^+ \rightleftharpoons FeHAsO_4^- + H_2O \quad (11.50)$$

Smith *et al*. (2002) measured linear partition coefficients for As(V) for a specific soil type (an Alfisol), with low amounts of iron and aluminum, to be 1.7–3.5 L/kg. In contrast, for two other soil types (a Vertisol and an Oxisol), with high iron and aluminum (7–12 g/kg), the values were 23–62 L/kg. The presence of added phosphorus reduced the partition coefficients. The Freundlich parameters for a mineral soil (Inceptisol) with very high iron (26 g/kg) from eastern Tennessee were measured to be $K_F$ = 278 L/kg and $n$ = 0.32 (Williams *et al*., 2003). Humic substances do not have much effect on the sorption of either As(III) or As(V) to ferrihydrite (Gräfe *et al*., 2002). Arsenate anion exchange processes are analogous to those of phosphate, with competition favoring phosphorus compounds (Mkandawire and Dudel, 2005). Arsenate can also be adsorbed by clays, particularly by kaolinite and montmorillonite at low pH (Knox *et al*., 1999).

Sediment resuspension provides a mechanism for remobilization of sediment-bound metals, including arsenic (Linge and Oldham, 2002). The movement of anoxic materials into the oxic water column causes the oxidation of sulfide compounds.

## Performance of Wetlands for Arsenic Removal

The observed removals of arsenic in constructed wetlands are very much dependent on the source water (Table 11.39). The higher concentrations in industrial, mining, and leachate waters allows for measurable and high reductions. However, urban stormwaters and treated municipal wastewaters do not contain appreciable arsenic, and removals are low, and in some cases are influenced by method detection limits. The median concentration reduction in 22 FWS systems is 29%, and the maximum is 99%. Alacrón-Herrera *et al*. (2006) found in three mesocosm subsurface flow constructed wetlands in Mexico an average arsenic removal of 90%. Ninety-nine percent of arsenic retained in the system was found in the gravel and only 1% was found plant biomass. However, the results from three operational HSSF systems in the Czech Republic showed median effect of −380% (Table 11.39). Surprisingly, also results from VF systems from Belgium and the Netherlands (Table 11.39) indicated that arsenic was washed out of the systems with arsenic concentration increase at the outflow between 14–170%. Arsenic that is adsorbed to Mn, Fe, and Al precipitates under oxic conditions is released when these precipitates got reduced under low redox potential condions. This would explain the increase of arsenic concentration in HSSF systems, but it seems that VF systems are not fully oxidized and contain reduced zones.

## Example Treatment Wetlands for Arsenic Removal

Arsenic is usually associated with other metallic contaminants, such as lead and zinc, because it is a component of some mined minerals. However, arsenic can be the primary target of treatment wetland systems.

### *The Trail, British Columbia, Wetland System (Duncan, 2002; Duncan et al., 2004)*

A vegetation-capped landfill, near a smelter in Trail, British Columbia, produces a leachate that contains arsenic and toxic metals. The design and construction of a retrofitted new "arsenic only" test facility are presented. The multistage treatment system consists of two anaerobic bioreactors (AB1 and AB2) and three subsurface flow wetlands in series, draining to a final holding pond-wetland. The anaerobic cells, 510 and 450 $m^2$, respectively, contained a meter of standing water over a substrate comprised of biosolids, sand, and manure together with limestone. The deep water was to provide winter protection from freezing temperatures. The HSSF cells, 50, 50, and 300 $m^2$, respectively, were vegetated with a mixture (HSSF 1), *Calamagrostis canadensis* (HSSF 2), and *Typha latifolia* (HSSF 3). The system treats about 8 $m^3$/d, for an overall hydraulic loading of 0.6 cm/d. Although the original system, built in 1998, targeted lead and zinc, as well as arsenic, the system was retrofitted in 1999 to focus on arsenic (Duncan, 2002). The removal rates for the complete system were 99.1%, 99.2%, and 94% for arsenic, cadmium, and zinc, respectively, over the period 2000–2001. The first two cells removed over 94.3%, 92.6%,

**TABLE 11.39**
**Arsenic Content of Example Wetland Sediments**

| Location | Notes | Water Source | Arsenic (µg/g) | Reference |
|---|---|---|---|---|
| Tres Rios, Arizona | C2 | Municipal | 10.0 | Wass, Gerke, and Associates (2002) |
| Roxboro, North Carolina | Reservoir | Power plant cooling | 1.25 | Masscheleyn *et al.* (1991a) |
| N. Bull Island, Co. Dublin, Ireland | Salt marsh | — | 2–36 | Doyle and Otte (1997) |
| Sacramento, California | Pilot | Secondary | 1.0–3.0 | Nolte and Associates (1998b) |
| Oxnard, California | Pilot | Backwash | 2.7–3.2 | CH2M Hill (2005) |
| Tulare Lake, California | *Spartina alterniflora* | Agricultural drainage | 9.7 | Fox and Doner (2002) |
| Tulare Lake, California | *Bolboschoenus maritimus* | Agricultural drainage | 14.8 | Fox and Doner (2002) |
| Woburn, Massachusetts | Peatland | Industrial contamination | 2161 | Keon *et al.* (2001) |
| Wheal Jane, United Kingdom | Aerobic | Mine water | 7400 | Swash and Monhemius (2005) |
| Wheal Jane, United Kingdom | Anaerobic | Mine water | 30 | Swash and Monhemius (2005) |
| Lake Yangebup, Australia | Lake/Wetland | Industrial inputs | 23 | Linge and Oldham (2002) |

**TABLE 11.40**
**Performance of Example Treatment Wetlands for Arsenic Reduction**

| System | Water Treated | Concentration (µg/L) In | Out | Reduction (%) | Reference |
|---|---|---|---|---|---|
| **Mine Water** | | | | | |
| Albright, Pennsylvania | Coal combustion leachate | 3.00 | 2.00 | 33 | Hoover *et al.* (1998) |
| Springdale, Pennsylvania | Coal combustion leachate | 61.00 | 5.00 | 92 | Hoover *et al.* (1998) |
| Orcopampa, Peru | Metal mine water | 138.00 | 37.00 | 73 | Unpublished data |
| Champion Paper, Florida | Pulp and paper | 9.00 | 9.30 | — | NADB database (1998) |
| Campbell Mine, Ontario | Gold mine water | 24.00 | 11.00 | 54 | Ontario Mining Association (2005) |
| Woodcutters, NT, Australia | Copper mine water | 5.00 | 2.00 | 60 | Noller *et al.* (1994) |
| Tom's Gully, NT, Australia | Metal mine water | 68.00 | 0.50 | 99 | Noller *et al.* (1994) |
| Trail, British Columbia | Metal mine water | 99,600 | 950 | 99 | Duncan *et al.* (2004) |
| **Landfill Leachate** | | | | | |
| New Hanover Co., North Carolina | Landfill leachate | 127.00 | 9.00 | 93 | Unpublished data |
| Isanti-Chisago, Minnesota | Landfill leachate | 6.92 | 4.16 | 40 | Kadlec (2003c) |
| **Stormwater** | | | | | |
| Everglades, Florida | Agricultural runoff | 3.0 | 2.1 | 30 | NADB database (1998) |
| Tarrant County, Texas | River, Train 1 | 3.7 | 3.0 | 19 | NADB database (1998) |
| Tarrant County, Texas | River, Train 2 | 3.7 | 2.3 | 38 | NADB database (1998) |
| Tarrant County, Texas | River, Train 3 | 3.7 | 2.7 | 27 | NADB database (1998) |
| **Municipal Wastewater** | | | | | |
| Estevan, Saskatchewan | FWS lagoon | 4.73 | 5.11 | –8 | Unpublished data |
| Sacramento, California | FWS | 2.37 | 2.80 | –18 | Nolte and Associates (1998b) |
| Hayward, California | Secondary | 1.90 | 3.80 | –100 | NADB database (1998) |
| Oxnard, California | FWS/SSF Train 1 | 2.50 | 2.50 | 0 | NADB database (1998) |
| Oxnard, California | FWS/SSF Train 2 | 2.50 | 2.50 | 0 | NADB database (1998) |
| Tres Rios, Arizona | FWS Cobble | 3.50 | 3.50 | 0 | NADB database (1998) |
| Tres Rios, Arizona | FWS Hayfield | 3.50 | 3.50 | 0 | NADB database (1998) |
| Tres Rios, Arizona | FWS C2 | 4.50 | 3.80 | 16 | Wass, Gerke, and Associates (2002) |
| Mořina, Czech Republic | HSSF | 1.0 | 4.8 | –380 | Unpublished data |
| Břehov, Czech Republic | HSSF | 0.8 | 6.5 | –713 | Unpublished data |
| Slavošovice, Czech Republic | HSSF | 5.5 | 2.0 | 64 | Unpublished data |
| Wepserveen, The Netherlands | VSSF | 2.0 | 2.5 | –25 | Unpublished data |
| Bokrijk, Belgium | VSSF | 5.0 | 13.5 | –170 | Unpublished data |
| Zemst-Larbeek, Belgium | VSSF | 3.5 | 4.0 | –14 | Unpublished data |
| Sint-Niklass-Heimolen, Belgium | VSSF | 5.0 | 9.3 | –86 | Unpublished data |
| Power station process water | FWS | 4.3 | 5.8 | 0 | Jensen *et al.* (2006) |

and 68.3% for arsenic, cadmium, and zinc. Arsenic removal was achieved mostly in the first two cells (Table 11.40). Over the period of record, concentrations were reduced by 99%. Companion research projects were undertaken at the University of Guelph, Ontario. Greenhouse studies tested a variety of plants for three effluent concentrations. This plant research showed that they could survive and sequester metals if the effluent metal concentration was reduced by 90%.

## Boron

Boron (B) is a naturally occurring element that is found in the form of borates in the oceans, sedimentary rocks, coal, shale, and in some soils. Borates are released naturally into the atmosphere and aquatic environment from oceans, geothermal steams, and weathering of clay-rich sedimentary rocks. Boron is also released to a lesser extent from anthropogenic sources. In the United States, 12% of $B_2O_3$ is consumed in detergents and bleaches; the percentage is even higher in Europe. Borax has been used as a laundry additive since about 1900. Glass and related uses consume about half of the borates as boric anhydride ($B_2O_3$). Fiberglass accounts for the largest share, comprising about 77% of the total (Woods, 1994).

### Wetland Processing and Storage of Boron

The aqueous chemistry of the borates is dependent on concentration and pH. Dilute aqueous boric acid solutions are comprised solely of $B(OH)_3$ and $B(OH)_4^-$ species. Freshwaters normally range from less than 0.01 mg/L to 0.66 mg/L, with higher concentrations in regions of sea water intrusion (Boyd and Walley, 1972). In Europe, freshwaters range from 3.3 µg/L (lakes in Finland) to 367 µg/L in Portugal (Wyness *et al.*, 2003). Boron in rainfall in the southeastern U.S. is at about 6 µg/L. Boron concentrations in soils range from 10 to 300 mg/kg with an average of 30 mg/kg. Concentrations of boron in

surface freshwaters typically range from less than 0.1 mg/L to 0.5 mg/L; much higher concentrations are measured in a few areas, depending on the geochemical nature of the drainage catchment. Boron accumulates in both aquatic and terrestrial plants, but it does not appear to be biomagnified through the food chain.

Boron is an essential micronutrient for higher plants, with interspecies differences in the levels required for optimum growth (Power and Woods, 1997). There is typically a small concentration range between deficiency and toxicity; however, toxicity owing to excess boron is much less common in the environment than boron deficiency. Borate, like silicate, is an essential micronutrient for the growth of small aquatic organisms, such as diatoms. Phytoplankton can tolerate up to 10 mg/L boron. Data for aquatic invertebrates suggests that the no observable effect levels are 13.6 mg/L for freshwater organisms (Eisler, 1990).

The boron content of some typical wetland plants and sediments is given in Table 11.41. In natural wetlands, plant tissues range from 1.2 to 11.3 µg/g across 22 species (Boyd and Walley, 1972). There is a clear indication that *Lemna* spp. accumulates considerably more boron than other plants. Glandon and McNabb (1978) observed that as water concentrations increased from 0.1 to 1.0 mg/L, *Lemna minor* responded with tissue boron increasing tenfold. Submerged aquatic plants appear to bioaccumulate boron in amounts intermediate between emergent plants and *Lemna* spp.

### *Boron Removal Processes*

Boron adsorbs on both crystalline and amorphous iron and aluminum oxides (Goldberg, 1997). The optimum pH is 8–9, and the mechanism involves ligand exchange with surface hydroxyl groups. Clay minerals exhibit strong binding for boron, with an optimum pH = 8. However, clays sorb less boron than oxyhydroxides on a per gram basis (Goldberg, 1997). Calcareous soil data are found to fit the Freundlich isotherm, with $n = 0.78$ and $K_F = 2.6$ (units mg/kg and mg/L). Alleoni and de Camargo (2000) found $1.24 < K_F < 4.84$ and $0.28 < n < 0.68$ for two clay soils, and $0.40 < K_F < 2.64$ and $0.338 < n < 0.67$ for two sandy loams. Peat has a greater affinity for boron, with partition coefficients in the range of 5–50 L/kg measured by Sartaj and Fernandes (2005) and Sartaj *et al.* (1999). Comerton *et al.* (2002) measured Freundlich parameters for three peats, and determined and $4.6 < K_F < 14.1$ and $0.66 < n < 0.90$ (see Equation 11.13).

## Performance of Wetlands for Boron Removal

The observed removals of boron in constructed wetlands are dependent on the source water (Table 11.42). The higher concentrations in industrial, mining, and leachate waters produce some reductions. However, urban stormwaters and treated municipal wastewaters do not contain appreciable boron, and removals are low or negative. The median concentration reduction in 13 systems is 5%, and the maximum is 91%, the latter achieved by a vertical peat filter. However, the removal efficiency varies widely among systems with –100% to +91% (Table 11.42).

## Example Treatment Wetlands for Boron Removal

Boron is usually associated with other contaminants, and is rarely the primary target of treatment wetland systems.

### *The Huneault Landfill Wetland System (Sartaj et al., 1999)*

The Huneault Landfill, located in Gloucester, Ontario, produces about 57,000 $m^3$/yr of leachate. The 40-ha system receives construction/demolition waste as well as industrial, commercial, and institutional materials. This is different from many landfills, which receive putrescible domestic waste. As a consequence, the leachate is low in BOD and ammonia, but does contain salts, metals, and other inorganic materials. Boron was determined to be the target contaminant, and a wetland system was constructed to treat one third of the total flow, during the frost-free period of 142 days. The existing facilities at the site, prior to wetland construction, included two ponds. The system is comprised of a vertical flow peat filter followed by three FWS cells in series. The peat filter had 1.4 m of organic and an area of 5,580 $m^2$, and a hydraulic loading of 2.4 cm/d. The ensuing FWS cells had a combined surface area of 4,374 $m^2$, and operated at 15 cm depth, for

**TABLE 11.41**
**Performance of the Trail, British Columbia, Wetland, Treatment System for Arsenic Removal (mg/L)**

| Year | 1999 | 2000 | 2001 | 2002 | 2003 | 5/14/01–1/20/02 |
|---|---|---|---|---|---|---|
| Input | 76.4 | 47.2 | 95.6 | 82.9 | 108 | 99.6 |
| Anaerobic #1 | 14.4 | 8.8 | 27.2 | 34.4 | 20 | 28.2 |
| Anaerobic #2 | NB | 11.7 | 5.8 | 8.2 | 10 | 5.7 |
| HSSF1 | — | — | — | — | — | 1.6 |
| HSSF2 | — | — | — | — | — | 1.1 |
| HSSF3 | — | — | — | — | — | 1.04 |
| Pond | 0.3 | 1.6 | 0.7 | 1.5 | 0.5 | 0.95 |

*Note:* NB = cell not yet built.

*Source:* Data from Duncan *et al.* (2004) In *Waste Processing and Recycling in Mineral and Metallurgical Industries V*, Metallurgical Society of the Canadian Institute of Mining, Metallurgy, and Petroleum, Montreal, Quebec, Canada, pp. 469–484.

**TABLE 11.42**
**Tissue and Sediment Concentrations of Boron in Example Wetland Systems**

| Location | Notes | Plant | Water (µg/L) | Above (µg/g) | Below (µg/g) | Reference |
|---|---|---|---|---|---|---|
| **Plants** | | | | | | |
| Belding, Michigan | Facultative pond | *Ceratophyllum demersum* | 520 | 62 | — | Glandon and McNabb (1978) |
| Belding, Michigan | Facultative pond | *Potamogeton pectinatus* | 520 | 327 | — | Glandon and McNabb (1978) |
| Belding, Michigan | Facultative pond | *Potamogeton foliosus* | 520 | 37 | — | Glandon and McNabb (1978) |
| Oxnard, California | FWS CW SAV | *Potemogeton pectinatus* | 1.2 | 292 | 191 | CH2M Hill (2005) |
| Oxnard, California | FWS CW | *Typha latifolia* | 1.2 | 30 | 19 | CH2M Hill (2005) |
| Oxnard, California | FWS CW | *Schoenoplectus americanus* | 1.2 | 11 | 13 | CH2M Hill (2005) |
| Oxnard, California | VSSF CW | *Juncus balticus* | 1.2 | 31 | 12 | CH2M Hill (2005) |
| Widows Creek, Alabama | FWS CW | *Typha latifolia* | 2,000 | 46 | 113 | Ye *et al.* (2001a,b) |
| Widows Creek, Alabama | FWS CW | *Juncus effusus* | 2,000 | 41 | 248 | Ye *et al.* (2001a,b) |
| Olentangy River, Ohio | FWS mesocosms | *Schoenoplectus tabernaemontani* | — | 14 | 8 | Ahn *et al.* (2001) |
| Japan | FWS mesocosms | *Zizania latifolia* | 500 | 25 | — | Enari *et al.* (2004) |
| Japan | FWS mesocosms | *Zizania latifolia* | 5,000 | 110 | — | Enari *et al.* (2004) |
| Czech Republic | HSSF CW | *Phragmites australis* | — | 2.1 | 3.2 | Vymazal *et al.* (2006) |
| Czech Republic | HSSF CW | *Phalaris arundinacea* | — | 2.9 | 4.7 | Vymazal *et al.* (2006) |
| Germany | SSF CW | *Typha angustifolia* | — | 24.5 | — | Seidel (1966) |
| Germany | SSF CW | *Scirpus lacustris* | — | 14.6 | — | Seidel (1966) |
| Germany | SSF CW | *Glyceria aquatica* | — | 15.0 | — | Seidel (1966) |
| Kiryat Sde Boker, Israel | FWS FAP | *Lemna gibba* | 2,000 | 1,250 | — | del Campo and Oron (2002) |
| Kiryat Sde Boker, Israel | FWS FAP | *Lemna gibba* | 10,000 | 1,850 | — | del Campo and Oron (2002) |
| Belding, Michigan | Facultative pond | *Lemna minor* | 520 | 1,515 | — | Glandon and McNabb (1978) |
| **Sediments** | | | | | | |
| Oxnard, California | Pilot | Backwash | 1.2 | — | 19–25 | CH2M Hill (2005) |
| Widows Creek, Alabama | Cattail marsh | Coal ash leachate | 2,000 | — | 500 | Ye *et al.* (2001a,b) |
| Olentangy River, Ohio | FWS mesocosms | Topsoil | — | — | 2.1 | Ahn *et al.* (2001) |

a detention time of 5.3 days. The peat filter was effective in reducing boron from 15 mg/L to 1.34 mg/L during the operating season of 1996. However, some bypassing was observed to have occurred, and the influent to the three wetland cells averaged 4.9 mg/L. Background levels in the FWS wetlands were 0.6–1.0 mg/L. Most of the boron was absorbed by the peat, which had a projected life of only four years.

## Selenium

Selenium is a metalloid, with reactivity and chemistry similar to sulfur. Four oxidation states exist: selenide (-II), elemental selenium (O), selenite (IV), and selenate (VI). Selenium soil solution chemistry is controlled by pH and redox conditions; however, organic carbon and microbial transformation also influence the availability of selenium for plant uptake (Zhang and Moore, 1997). Under most conditions, inorganic selenium in solution is found in one of two oxidation states, as selenite ($SeO_3^{2-}$) and as the more mobile selenate ($SeO_4^{2-}$). Selenate is the dominant form at high redox and high pH, is more readily taken up by plants, and considered to be the most bioaccumulative inorganic ion; however, organic selenium has been shown to be more mobile and bioaccumulative than selenate. Selenium readily forms organic complexes, such as dimethyl selenide and selenomethionine.

U.S. EPA (2002a) has set a selenium continuous concentration criterion of 5 µg/L to protect aquatic life. Biological effects thresholds for selenium are between 30 and 370 mg/kg dry weight for 300 plankton, benthic invertebrates, and certain forage fish; 4 mg/kg dry weight for some freshwater and anadromous fish; and 3–10 mg/kg dry weight for some aquatic birds (Lemly *et al.*, 1993). The toxicity of the volatile selenium compounds, such as dimethylselenide (DMSe), are 600–700 times lower than the inorganic forms (DWTTC, 1999). Of all the priority and nonpriority pollutants, selenium has the narrowest range of what is beneficial for biota and what is detrimental (U.S. EPA, 2004a).

### Wetland Processing and Storage of Selenium

The selenium concentrations in vegetation in treatment wetlands exposed to selenium are typically in the range of 1–20 µg/g for both aboveground and belowground parts (Table 11.43). However, the fallen litter at Corcoran, California, contained 10–20 µg/g. Treatment wetland sediments have 1–10 µg/g (Table 11.44). Although these values are low, there is still a possibility that bioaccumulation can reach unacceptable levels under some circumstances. However, the results of the Richmond and Corcoran, California, studies have been encouraging, because toxic effects have not been identified.

#### *Selenium Removal Processes*

Constructed wetlands remove selenium by reduction to insoluble forms which are deposited in the sediments, by

## TABLE 11.43
## Boron Removal Performance of Example Treatment Wetlands

| System | Notes | Concentration (µg/L) In | Out | Reduction (%) | Reference |
|---|---|---|---|---|---|
| **Mine Water** | | | | | |
| Springdale, Pennsylvania | Coal ash leachate | 15,920 | 14,030 | 12 | Hoover *et al.* (1998) |
| Widows Creek, Alabama | Coal ash leachate | 2,340 | 1,680 | 28 | Ye *et al.* (2001a, b) |
| **Landfill Leachate** | | | | | |
| Gloucester, Ontario | Construction debris leachate | 15,000 | 1,340 | 91 | Sartaj *et al.* (1999) |
| **Municipal Wastewater** | | | | | |
| Estevan, Saskatchewan | FWS lagoon | 1.13 | 1.04 | 8 | Unpublished data |
| Hayward, California | Secondary | 1.90 | 3.80 | –100 | NADB database (1998) |
| Tres Rios, Arizona | FWS (cobble site) | 0.38 | 0.36 | 5 | NADB database (1998) |
| Tres Rios, Arizona | FWS (hayfield site) | 0.38 | 0.36 | 5 | NADB database (1998) |
| Oxnard, California | FWS/SSF Train 1 | 1.20 | 1.22 | –1 | CH2M Hill (2005) |
| Oxnard, California | FWS/SSF Train 2 | 1.20 | 1.57 | –31 | CH2M Hill (2005) |
| Mořina, Czech Republic | HSSF | 423 | 567 | –34 | Unpublished data |
| Břehov, Czech Republic | HSSF | 333 | 128 | 62 | Unpublished data |
| Slavošovice, Czech Republic | HSSF | 240 | 35 | 85 | Unpublished data |
| **Power Station Process Water** | | | | | |
| Australia | FWS | 6.5 | 6.9 | 0 | Jensen *et al.* (2006) |
| **Median (Overall)** | | | | **5** | |

## TABLE 11.44
## Selenium Content of Wetland Plants

| Location | Wetland Type | Plant | Water (µg/L) | Above (µg/g) | Below (µg/g) | Reference |
|---|---|---|---|---|---|---|
| Tres Rios, Arizona | FWS CW | *Typha* spp. | 5.2 | 0.93 | 1.32 | Wass, Gerke, and Associates (2002) |
| Tres Rios, Arizona | FWS CW | *Scirpus olneyi* | 5.2 | 0.90 | 0.99 | Wass, Gerke, and Associates (2002) |
| Tres Rios, Arizona | FWS CW | *Scirpus validus* | 5.2 | 1.16 | 0.39 | Wass, Gerke, and Associates (2002) |
| Tres Rios, Arizona | FWS CW | *Hydrocotyle* spp. | 5.2 | 1.04 | 1.86 | Wass, Gerke, and Associates (2002) |
| Oxnard, California | FWS CW SAV | *Potemogeton pectinatus* | 3.8–9.0 | 3.32 | 1.77 | CH2M Hill (2005) |
| Oxnard, California | FWS CW | *Typha latifolia* | 3.8–9.0 | 1.48 | 2.68 | CH2M Hill (2005) |
| Oxnard, California | FWS CW | *Scirpus americanus* | 3.8–9.0 | 3.24 | 5.09 | CH2M Hill (2005) |
| Oxnard, California | VSSF CW | *Juncus balticus* | 3.8–9.0 | 1.50 | 1.59 | CH2M Hill (2005) |
| Juelich, Germany | SSF | *Phragmites australis* | 20.4 | 1.80 | 0.80 | Shardendu *et al.* (2003) |
| Břehov, Czech Republic | HSSF | *Phragmites australis* | Sewage | 0.73 | 0.96 | Vymazal *et al.* (2006) |
| Břehov, Czech Republic | HSSF | *Phalaris arundinacea* | Sewage | 0.7 | 0.88 | Vymazal *et al.* (2006) |
| Juelich, Germany | SSF | *Typha latifolia* | 20.4 | 1.20 | 2.20 | Shardendu *et al.* (2003) |
| Richmond, California | FWS | *Scirpus maritimus* | 20–30 | 5.20 | 15.50 | Hansen *et al.* (1998) |
| Richmond, California | FWS | *Typha* spp. | 20–30 | 6.10 | 14.00 | Hansen *et al.* (1998) |
| Richmond, California | FWS | *Cotula coronopifolia* | 20–30 | 7.80 | 15.10 | Hansen *et al.* (1998) |
| Richmond, California | FWS | *Distichlis spicata* | 20–30 | 7.10 | 11.00 | Hansen *et al.* (1998) |
| Richmond, California | FWS | *Polypogon monspieliensis* | 20–30 | 7.90 | 20.70 | Hansen *et al.* (1998) |
| Australia | FWS | *Typha orientalis* | 108–320 | 1.6–34 | 31–474 | Jensen *et al.* (2006) |
| Imperial, California | Marsh | *Scirpus* spp. | 6.5 | 0.86 | — | Unpublished data |
| Brawley, California | Marsh | *Scirpus* spp. | 10.0 | 0.67 | — | Unpublished data |
| Corcoran, California[a] | Marsh | *Bolboschoenus maritimus* | 11–14 | 14.6 | — | Gao *et al.* (2003b) |
| Corcoran, California[a] | Marsh | *Scirpus balticus* | 5–11 | 15.3 | — | Gao *et al.* (2003b) |
| Corcoran, California[a] | Marsh | *Spartina alterniflora* | 7–14 | 18.4 | — | Gao *et al.* (2003b) |
| Corcoran, California[a] | Marsh | *Polypogon monspeliensis* | 14–16 | 19.4 | — | Gao *et al.* (2003b) |
| Corcoran, California[a] | Marsh | *Distichlis spicata* | 9–17 | 10.8 | — | Gao *et al.* (2003b) |
| Corcoran, California[a] | Marsh | *Typha latifolia* | 11–14 | 16.8 | — | Gao *et al.* (2003b) |
| Corcoran, California[a] | Marsh | *Ruppia maritima* + *Scirpus* | 11–13 | 11.3 | — | Gao *et al.* (2003b) |
| Corcoran, California[a] | Marsh | *Schoenoplectus* + *Typha* | 9–11 | 12.2 | — | Gao *et al.* (2003b) |

[a] Values are for leaf litter (plant detritus).

accumulation into plant tissues, and by volatilization to the atmosphere, i.e., through biological volatilization of plants, plant/microbe associations, and microbes alone.

### *Selenium Reduction*

Water column removal processes include reduction of selenate and selenite to elemental selenium, and subsequent sequestration in plants and sediments. The sequence is:

$$SeO_4^{2-} \rightarrow SeO_3^{2-} \rightarrow Se^o \rightarrow Se^{2-} \qquad (11.51)$$

Lin and Terry (2003) determined that surface sediments at the Corcoran wetlands, receiving water containing Se(VI), contained Se(VI):Se(IV):Se(O):Se(-II) = 0:13:41:46. In *Polypogon monspeliensis* aboveground tissues, the proportions were 32:17:27:24.

Sediments provide a reducing environment, and can produce and contain selenide and elemental selenium. The elemental forms have a distinctive red color, and can be produced using litter from rice plants (*Oryza sativa)* (Zhang and Frankenberger, 2003a). Because Se(O) is insoluble in aquatic systems, reduction of Se(VI) to Se(O) is a potentially useful technique for bioremediation. However, removal of Se(O) from the drainage water is slow, possibly because of the low precipitation rate of colloidal Se(O) due to its small size, ranging from 0.2 to 0.4 μm. Detention times of 10–16 days were found to give 89–92% reduction of Se(VI) to Se(O) in mesocosm trains (Zhang and Frankenberger, 2003b). Without straw as a trapping aid, about half the colloidal Se(O) was susceptible to partial oxidation to Se(IV) (Zhang *et al*., 2004).

Dissimilatory selenate reduction is carried out by anaerobic bacteria that use selenate or selenite in the oxidation of organic matter:

$$2CH_2O + SeO_4^{2-} \rightarrow 2HCO_3^- + H_2Se \qquad (11.52)$$

where $CH_2O$ represents organic matter.

Zhang and Moore (1997) describe the process via Freundlich isotherms, with $0.77 < n < 0.93$ and $0.90 < K_F < 1.00$ (see Equation 11.13).

### *Volatilization*

There is evidence that significant losses of selenium, through biologically mediated methylation and volatilization, can occur in wetlands. Upon release from organisms, selenium may be converted to the volatile dimethyl selenide form (Masscheleyn *et al*., 1991a; Masscheleyn and Patrick, 1993). This removal route involves conversion to organic forms such as dimethylselenide, which can volatilize from the wetlands, mediated by plants (Azaizeh *et al*., 2003):

$$CH_2O + SeO_4^{2-} \rightarrow (CH_3)_2Se \qquad (11.53)$$

The field measurements of Hansen *et al*. (1998) indicated volatilization rates of 60–190 μg/m²·d. Plexiglas chambers that covered whole plants were being used to monitor selenium volatilization rates. However, Lin and Terry (2003) measured lower rates, 5–30 μg/m²·d, at the Corcoran, California, site, also using flux chambers. This represented only about 1–5% of the applied selenium (Gao *et al*., 2003a, b).

### *Sorption*

Selenite has a stronger affinity for sorption sites than Se(VI) and thus is readily adsorbed to sediments and soil constituents such as Fe-, Mn-, and Al-oxyhydroxides, which are the major adsorbents for removing soluble selenium from water.

### *Plant Uptake*

Carvalho and Martin (2001) studied the hydroponic uptake of selenite by cattails *(Typha domingensis)*, duckweed (*Lemna obscura*), *Hydrillaverticillata*, and swamp lilies (*Crinum americanum).* In one week, all four were able to remove 50–100% of selenium solutions of 1–200 mg/L. Tissue concentrations in *Lemna* and *Hydrilla* were increased to 0.1–1.3 μg/g, and to 30–80 μg/g in cattails and swamp lilies. Jensen *et al*. (2006) reported concentrations of selenium up to 200 and 1,400 μg/g in stalks and roots of *Typha orientalis* growing in FWS systems treating power station process water in Australia. The high concentrations of selenium in roots of *Typha* did not appear to have a detrimental effect on the plant growth. However, mass balances on field-scale operating systems over several years indicate that very little of the applied selenium load is stored in plant tissues (Gao *et al*., 2003a,b).

## Performance of Wetlands for Selenium Removal

Although FWS wetlands process selenium, the observed removals in constructed wetlands have not been outstanding (Table 11.45). The median concentration reduction in 15 FWS systems is 25%, and the maximum is 68%. However, results from operational HSSF systems indicate substantial increase of selenium concentration at the outflow as compared to the inflow, with median increase of 292%. (Table 11.45). This is unexpected because with decreasing redox potential values selenium solubility decreases due to formation of achavalite (FeSe) and ferroselite ($FeSe_2$) (Masscheleyn *et al*., 1991a,b).

Figure 11.25 summarizes the inflow and outflow dissolved selenium concentrations at the Imperial, California, wetland. The project was primarily intended to reduce pathogens and nutrients, but selenium was a concern because the influent was agricultural runoff from arid lands. This system had a sedimentation basin, followed by a four-cell series FWS wetland. There was considerable open water, and conditions are generally well-oxygenated. The overall concentration reduction was 32% over a three-year period of record, although load reductions were greater (47–59%) because of infiltration (TTI and WMS, 2006). The first-order rate constant for removal of selenium was calculated to be 12 m/yr for the Imperial, California, wetland. A companion system was located at Brawley, California, also in the Imperial Valley. This system contained a sedimentation basin and two FWS cells in series. Data on the inflow and outflow dissolved

## TABLE 11.45
## Selenium Content of Wetland Sediments

| Location | Wetland Details | Water Source | Water (µg/L) | Sediments (µg/g) | Reference |
|---|---|---|---|---|---|
| Tres Rios, Arizona | C2 | Municipal | — | 4.3 | Wass, Gerke, and Associates (2002) |
| Roxboro, North Carolina | Reservoir | Power plant cooling | — | 1.2 | Masscheleyn *et al.* (1991b) |
| Oxnard, California | Pilot | Backwash | 3.8–9.0 | 0.40–0.62 | CH2M Hill (2005) |
| Juelich, Germany | Pilot | Dosed | — | 1.0–2.8 | Shardendu *et al.* (2003) |
| Richmond, California | *Scirpus maritimus* | Refinery | 20–30 | 4.6 | Hansen *et al.* 1998 |
| Richmond, California | *Typha* spp. | Refinery | 20–30 | 5.9 | Hansen *et al.* (1998) |
| Richmond, California | *Cotula coronopifolia* | Refinery | 20–30 | 6.7 | Hansen *et al.* (1998) |
| Richmond, California | *Distichlis spicata* | Refinery | 20–30 | 2.3 | Hansen *et al.* (1998) |
| Richmond, California | *Polypogon monspieliensis* | Refinery | 20–30 | 2.1 | Hansen *et al.* (1998) |
| Martinez, California | *Distichlis spicata* | Runoff | 0.1–0.2 | 0.88–1.4 | Zawislanski *et al.* (2001) |
| Martinez, California | *Scirpus-Typha-Spartina* | Runoff | 0.1–0.2 | 0.56–0.58 | Zawislanski *et al.* (2001) |
| Stewart Lake, Utah | *Microcosms* | Lake sediment | — | 9.96 | Zhang *et al.* (2004) |
| Tulare Lake, California | *Microcosms* | Lake sediment | — | 0.44 | Zhang *et al.* (2004) |
| Imperial, California | Sedimentation basin | Agricultural runoff | 8.7 | 2.15 | Unpublished data |
| Imperial, California | Marsh | Agricultural runoff | 6.5 | 1.53 | Unpublished data |
| Brawley, California | Sedimentation basin | Agricultural runoff | 10.9 | 1.26 | Unpublished data |
| Brawley, California | Marsh | Agricultural runoff | 10.0 | 1.34 | Unpublished data |
| Corcoran, California[a] | *Bolboscoenus maritimus* | Agricultural runoff | 11–14 | 8.5 | Gao *et al.* (2003b) |
| Corcoran, California[a] | *Scirpus balticus* | Agricultural runoff | 5–11 | 12.5 | Gao *et al.* (2003b) |
| Corcoran, California[a] | Open water | Agricultural runoff | 14–16 | 7.0 | Gao *et al.* (2003b) |
| Corcoran, California[a] | *Spartina alterniflora* | Agricultural runoff | 7–14 | 10.0 | Gao *et al.* (2003b) |
| Corcoran, California[a] | *Polypogon monspeliensis* | Agricultural runoff | 14–16 | 11.3 | Gao *et al.* (2003b) |
| Corcoran, California[a] | *Distichlis spicata* | Agricultural runoff | 9–17 | 7.5 | Gao *et al.* (2003b) |
| Corcoran, California[a] | *Typha latifolia* | Agricultural runoff | 11–14 | 8.0 | Gao *et al.* (2003b) |
| Corcoran, California[a] | *Ruppia maritima* + *Scirpus* | Agricultural runoff | 11–13 | 7.3 | Gao *et al.* (2003b) |
| Corcoran, California[a] | *Schoenoplectus* + *Typha* | Agricultural runoff | 9–11 | 8.6 | Gao *et al.* (2003b) |
| Corcoran, California[a] | *Typha latifolia* | Agricultural runoff | 7–9 | 8.0 | Gao *et al.* (2003b) |

[a] Values are for newly deposited materials.

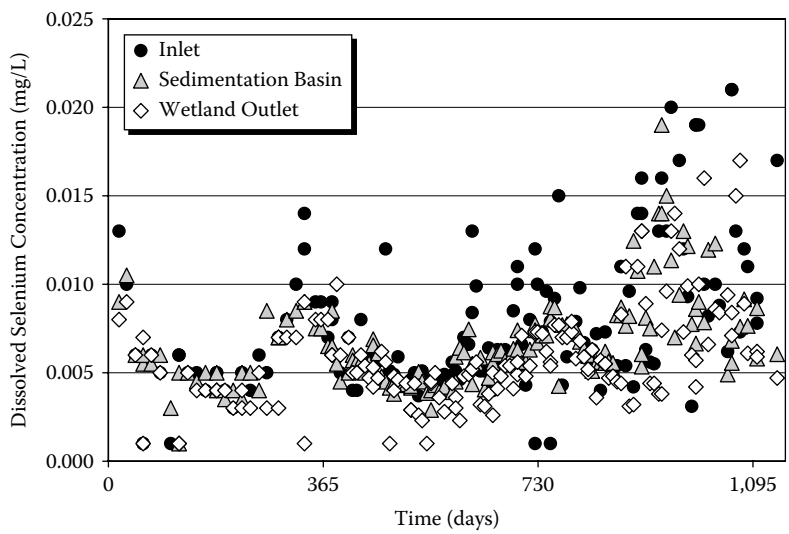



**FIGURE 11.25** Selenium water concentrations through the Imperial, California, treatment wetland, 2001–2003. The mean inlet was 8.7 µg/L, 7.1 µg/L out of the sedimentation basin, and 5.9 µg/L out of the wetland. (Unpublished data from Imperial Irrigation District.)

**TABLE 11.46**
**Performance of Treatment Wetlands in Selenium Removal**

| System | Notes | Concentration (µg/L) In | Out | Reduction (%) | References |
|---|---|---|---|---|---|
| **Mine Water** | | | | | |
| Albright, Pennsylvania | Coal combustion leachate | 4.00 | 2.00 | 50 | Hoover *et al.* (1998) |
| Springdale, Pennsylvania | Coal combustion leachate | 2.00 | 2.00 | 0 | Hoover *et al.* (1998) |
| Richmond, California | Refinery wastewater | 25.00 | 8.00 | 68 | Hansen *et al.* (1998) |
| **Landfill Leachate** | | | | | |
| New Hanover County, North Carolina | | 76.00 | 21.00 | 72 | Unpublished data |
| **Stormwater** | | | | | |
| Imperial, California | Agricultural runoff | 7.10 | 5.90 | 17 | Unpublished data |
| Brawley, California | Agricultural runoff | 10.10 | 9.90 | 2 | Unpublished data |
| Corcoran, California (10 wetlands, 1997) | Agricultural runoff | 22.00 | 17.50 | 20 | Lin and Terry (2003) |
| Corcoran, California (10 wetlands, 1998) | Agricultural runoff | 21.00 | 14.00 | 33 | Lin and Terry (2003) |
| Corcoran, California (10 wetlands, 1999) | Agricultural runoff | 16.00 | 9.00 | 44 | Lin and Terry (2003) |
| **Municipal Wastewater** | | | | | |
| Estevan, Saskatchewan | FWS lagoon | 1.70 | 1.80 | –6 | Unpublished data |
| Sacramento, California | FWS | 0.60 | 0.50 | 17 | CH2M HILL (1998) |
| Oxnard, California | FWS/SSF Train 1 | 12.00 | 9.00 | 25 | CH2M Hill (2005) |
| Oxnard, California | FWS/SSF Train 2 | 12.00 | 6.00 | 50 | CH2M Hill (2005) |
| Tres Rios, Arizona | FWS C2 | 5.20 | 5.00 | 4 | Wass, Gerke, and Associates (2002) |
| Mořina, Czech Republic | HSSF | 0.54 | 2.12 | –292 | Unpublished data |
| Břehov, Czech Republic | HSSF | 0.4 | 4.0 | –900 | Unpublished data |
| Slavošovice, Czech Republic | HSSF | 1 | 2.6 | –160 | Unpublished data |
| **Power Station Process Water** | | | | | |
| Australia | FWS | 146 | 104 | 29 | Jensen *et al.* (2006) |

selenium concentrations for the Brawley wetland showed essentially no concentration reduction, but load reductions were (29–60%) because of infiltration. First-order rate constants for removal of selenium were nominally 0.5 m/yr.

The mass balance for the ten-cell Corcoran project defines the fate of removed selenium over the history of that project (Gao *et al.*, 2003a; Tables 11.46 and 11.47). Approximately 39% of the added selenium passed through the wetlands, mostly in surface flow. Of the remaining 61%, most was founding surface sediments (new deposits 18%) and soils (0–20 cm layer 33%). Only 2% was found in plant litter, and less than 1% in plants. Flux measurements indicated that about 2% was lost to volatilization.

Further definition of the fate of removed selenium was obtained by several investigators, via sequential extraction of the sediments. For instance, Gao *et al.* (2003b) determined the fractionation of Se species in the 0–5 cm mineral sediments at Corcoran. The largest fraction was elemental Se(O) ($Na_2SO_3$ extractable) (46.6%), followed by organic matter associated selenium (NaOH extractable) (33.7%). Other extractable forms of selenium were small proportions of total selenium: KCl-extractable (soluble) selenium (1.2%), adsorbed selenium ($PO_4$-extractable) (2.5%), and carbonate-associated selenium (NaOAc-extractable) (1.2%). Residual (nonextractable) selenium was 15%.

**TABLE 11.47**
**Selenium Mass Balance on the Combined Ten Cells at Corcoran, California, May 1997 through September 2000 (in percent)**

| | | | |
|---|---|---|---|
| **Total In** | | | **100** |
| *Outflows* | Surface outflow | 35 | |
| | Seepage | 4 | |
| *Total out with water* | | | 39 |
| *Volatilization* | | | 2 |
| *Internal storage* | Surface soils 0–20 cm | 33 | |
| | Surface sediments | 18 | |
| | Litter | 2 | |
| | Plants | <1 | |
| | Water | <1 | |
| *Total storage* | | | 53 |
| *Unaccounted* | | | 6 |
| **Total Out** | | | **100** |

*Source:* Data from Gao *et al.* (2003a) *Journal of Environmental Quality* 32(4): 1557–1570; and Gao *et al.* (2003b) *Water, Air, and Soil Pollution* 144: 263–284.

### *The Kesterson Debacle—A Case Study*

A case study from the San Joaquin Valley in California is a prime example of wetland soil and surface water contamination derived from agricultural practices. Seleniferous soils leached by agricultural drainage waters, at a selenium concentration of 300 µg/L, were directed by tile drains and transported to the Kesterson National Wildlife refuge, a series of man-made marshes that were originally constructed between 1969 and 1975. The impoundment of selenium-rich waters led to soil concentrations 100 times higher than those in nearby control sites. Elevated selenium concentrations in soil solution led to high concentrations of selenium in plants, invertebrates, reptiles, fish, mammals, and birds (Ohlendorf *et al.*, 1986).

Emergent plants at Kesterson contained about 52.1 mg/kg (dry mass) of selenium compared to 0.43 mg/kg dry weight at the nearby Volta Wildlife Area, which did not receive agricultural drainage waters. The most pronounced effects were seen in waterfowl and shore birds, with common occurrences of embryonic mortality and deformity, and mortality in adult birds (Ohlendorf *et al.*, 1990). The "Kesterson Effect" of selenium bioconcentration in arid land wetlands receiving agricultural drain waters is considered to be a widespread problem in the American West (Lemly *et al.*, 1993).

## Example Treatment Wetlands for Selenium Removal

The Kesterson situation generated intense interest in learning the details of selenium processing in wetlands. Studies at existing constructed wetlands were undertaken, and new pilot systems were built for that express purpose. These studies have allowed the forecasting of potential effects of other constructed wetland projects, which proved to be acceptable for the Irvine Ranch Water District project (Horne, 2004; Fleming-Singer and Horne, 2006).

### *The Richmond Refinery Wetland (Duda, 1992; Hansen et al., 1998)*

The Chevron oil company conducted a study in California of a 36-ha FWS constructed wetland located adjacent to San Francisco Bay. The system consisted of three cells, with saltmarsh bulrush (*Scirpus maritimus*), cattails (*Typha* spp.), brass buttons (*Cotula coronopifolia*), and rabbitfoot grass (*Polypogon monspeliensis*). Flows of about 6,500 $m^3$/d created nominal detention of about seven to ten days. Analysis of the wetland inlet and outlet waters showed that the constructed wetland was successful in removing 89% of the selenium from the wastewater passing through it. Inflow selenium concentrations of 20–30 µg/L decreased to <5 µg/L in the outflow (Hansen *et al.*, 1998). Measurements of selenium volatilization showed that biological volatilization might have accounted for as much as 10–30% of the total selenium removed. Rates of volatilization were much higher in vegetated areas than in nonvegetated areas. Azaizeh *et al.* (1997) determined that the process was bacterially mediated, and that selenomethionine was the principal volatilized species, followed by selenite.

### *The Corcoran, California, Demonstration Project (Lin and Terry, 2003; Gao et al., 2003a, b)*

A flow through experimental wetland system was studied from 1996 through 1999, which was designed to remove selenium from agricultural drainage water in the Tulare Lake Drainage District of California. The system consisted of ten FWS cells, 15 × 76 m, operated at 7–21 days nominal detention time. The cells were established with bulrush (*Schoenoplectus robustus*), saltmarsh bulrush (*Bolboscoenus maritimus*), Baltic rush (*Scirpus balticus*), saltgrass (*Distichlis spicata*), cordgrass (*Spartina alterniflora*), rabbitfoot grass (*Polypogon monspeliensis*), widgeon grass (*Ruppia maritima*), tules (*Scirpus acutus*), and cattails (*Typha latifolia*), plus a water-only control. In 1999, the wetland cells reduced total selenium by 32–65% in concentration and 43–89% in mass. The nonvegetated cell showed the least removal of selenium. The incoming water was dominated by selenate (Se(VI), 91%), with smaller percentages of selenite (Se(IV), 7%) and organic selenium (org-Se-(II), 2%). Surface water, sediment pore water, plant tissue, and sediment samples are being collected monthly to determine the removal efficiency, fate of selenium, and the oxidation state of water soluble selenium. The outflow contained an average of 47% Se(VI), 32% Se(IV), and 21% org-Se, indicating reduction processes.

The surface sediment was the largest sink for selenium. The highest selenium concentration was found in litter, followed by the fine organic sediment layer. The sediment selenium concentration decreased with increasing sediment depth. Fractionation of surface sediment (0–5 cm) revealed that elemental selenium was the largest fraction (average 47%), followed by organic matter-associated selenium (34%). Soluble, adsorbed, and carbonate-associated selenium accounted for 1.2%, 3.1%, and 2.5%, respectively. The major selenium removal mechanism was the reduction of selenate to elemental selenium and immobilization into the sediment organic phase.

In conjunction with this field-scale project, a mesocosm-scale project was conducted at the University of California–Riverside (DWTTC, 1999). This project was designed to study issues such as replication and leaching losses. Analysis of the influent and effluent indicated selenium load reduction of 42% in the vegetative treatments and 28% in the nonvegetative treatments.

## SUMMARY

Constructed wetlands are being considered as an option for treating a variety of elements including salts and metals. Design of wetland systems to treat these elements is currently limited by a small but growing knowledge of the factors that control removal rates and of the biological effects of concentrating many of these elements in wetlands. However, for those substances associated with mine waters—notably iron, manganese, copper, nickel, and lead—there is now a large body of knowledge and alternative design recommendations.

This chapter summarizes the current knowledge about the fate and effects of halogens, sulfur, heavy metals, and

metalloids in natural and constructed wetlands. From this review, it is clear that wetlands cannot be used for consistent removal of most salts. This limitation exists because the typical surface water concentrations of most salts exceed growth requirements and because salts are soluble and tend to leach from sediments and tissues. In short, there are no microbial removal mechanisms except for sulfate reduction, and removal is to storage in the wetland sediments.

Wetlands are, however, effective at retaining significant loads of several trace metals. Sustainable metal uptake occurs primarily in the wetland sediments. However, this storage capacity in preexisting wetland sediments and biota will eventually be exceeded, and sooner at high influent loadings. Thereafter, there remains only a lesser removal potential to the newly forming sediments. It would be desirable that metal levels be reduced to levels that will not result in chronically toxic conditions to biota. In constructed FWS wetland treatment systems, this is not always practical. SSF wetlands eliminate most biological contact but may require some effort to rebuild or clean out if the bed becomes saturated with metals. In another approach, constructed wetlands may be used for a limited time depending on the cumulative loading of one or more metals. Limits on metal loadings to soils are based on accumulated concentrations that are kept below toxicity thresholds of sensitive biota. When the allowable loading limit is reached, these sites are abandoned or cleaned out to an alternate burial site. In many cases, however, calculation of actual metals loading rates indicates that the system's life is on the order of many decades or longer. Actual burial of metals by accreting sediments may further extend wetland treatment system life expectancies.

# 12 Pathogens

Pathogens are present in untreated domestic wastewaters as well as in runoff waters from animal sources. These organisms range from submicroscopic viruses to parasitic worms visible to the unaided eye and represent important components of water quality. Table 12.1 lists some of the most prevalent human pathogens associated with domestic wastewater. These pathogens are divided into five groups: viruses, bacteria, fungi, protozoans, and helminths. The density of these organisms in raw wastewater varies geographically.

Viruses are submicroscopic, nonliving particles of genetic material that are enclosed in a sheath. Viruses cannot divide and reproduce alone, but they can infect host organisms and reproduce to very large populations at the expense of the host organism. Over 100 virus types are known to occur in human feces, with minimum infective doses as low as one organism for some species. Bacteria are universally present in human feces, with normal populations of about $10^{11}$ organisms per gram (Leclerc *et al.*, 1977). Although most of these organisms live symbiotically with their hosts, a number of species are known human pathogens and occur with great frequency in infected individuals. Human parasites derived from wastewater-related infections include protozoa and helminths. Two common protozoan parasites are *Entamoeba histolytica* and *Giardia lamblia*, which both cause diarrhea in infected humans. The phylum *Aschelminthes* (cavity worms) includes all parasitic worms incapable of adult life without a host organism. A number of helminths, including tapeworms and flukes, are found in infected humans and can be spread through wastewater pathways. For a summary discussion of environmentally transmitted pathogens, see Maier *et al.* (2000).

The efficiencies of conventional treatment technologies that reduce pathogens to noninfective levels have been studied thoroughly, and wastewater treatment plants regularly add processes to accomplish necessary removals (Metcalf and Eddy Inc., 1991; Crites and Tchobanoglous, 1998). The most common add-on disinfection processes are chlorination, ozonation, and ultraviolet irradiation.

Because of its low cost and proven effectiveness, chlorination has been the disinfection method of choice for many years. However, negative side effects of effluent chlorination have become apparent in the past two decades. Residual free chlorine harms a variety of aquatic organisms and causes chronic and acute toxicity to microorganisms and fish. Also, when it comes in contact with organic compounds in wastewater or receiving waters, free chlorine forms trihalomethanes and other organochlorine compounds known to be carcinogenic. These findings have resulted in the increased use of dechlorination techniques and in the development of the ozonation and ultraviolet disinfection technologies.

Animals are also a source of pathogenic organisms. Farm animals produce feces with high numbers of bacteria. Cows, sheep, pigs, and poultry produce $10^5$–$10^7$ fecal coliforms per gram, and $10^6$–$10^8$ fecal streptococci per gram. Cats, dogs, mice, and chipmunks produce similar concentrations (Maier *et al.*, 2000). Waterfowl and other wetland birds also contribute coliforms to the wetland environment and may do so in great quantities during migratory concentrations. Beavers, muskrats, and other warm-blooded wetland animals are also producers of enteric pathogens. Beavers have been implicated in transmission of *Giardia*. About half the animals in Colorado were found to be infected and shedding $10^8$ cysts per animal per day. Ninety-five percent of muskrats in this study were found to be infected with *Giardia* (Erlandsen, 1995).

Natural treatment technologies have the potential to reduce populations of enteric pathogens because of natural die-off rates and hostile environmental conditions. Wetlands have been found to reduce pathogen populations with varying but significant degrees of effectiveness. This chapter reviews the pathogens typically found in wastewater and describes the effect that treatment wetlands have on the pathogen populations passing through them.

## 12.1 INDICATOR ORGANISMS AND MEASUREMENT

Measurement of human pathogenic organisms in natural and wastewaters is expensive and technically challenging. Consequently, it has been customary to first look for indicator organisms that are easy to monitor and correlate with populations of pathogenic organisms. No perfect indicators have been found, but the coliform bacteria group has long been used as the first choice among indicator organisms. Typical indicator organisms are listed in Table 12.2.

In the United States, total coliforms were the first approved indicator organisms. These coliforms include bacterial species that are rod-shaped, stain gram-negative, do not form spores, are facultatively anaerobic, and ferment lactose with gas production in 48 hours at a temperature of 35°C. Because many members of the group are not limited to fecal sources, methods have been developed to differentiate organisms of fecal origin. Fecal coliforms are separated from total coliforms by their ability to ferment lactose with gas production in 24 hours at a temperature of 44.5°C.

**TABLE 12.1**
**Some Human Pathogens Typical of Domestic Wastewater**

| | Pathogen | Illness |
|---|---|---|
| **Viruses** | Adenovirus (31 types) | Respiratory disease |
| | Enteroviruses (67 types) | Diarrhea, respiratory disease, polio |
| | Hepatitis A | Infectious hepatitis |
| | Norwalk agent | Gastroenteritis |
| | Rotavirus | Diarrhea |
| | Reovirus | Gastroenteritis |
| | HIV | AIDS |
| **Bacteria** | *Campylobacter jejuni* | Diarrhea |
| | *Escherichia coli* | Diarrhea |
| | *Legionella pneumophila* | Fever, respiratory tract infections |
| | *Leptospira* (150 spp.) | Leptospirosis |
| | *Salmonella typhi* | Typhoid fever |
| | *Salmonella* (~1,700 spp.) | Salmonellosis |
| | *Shigella* (4 spp.) | Diarrhea, dysentery |
| | *Vibrio* spp. | Cholera, diarrhea |
| | *Yersinia* spp. | Yersiniosis |
| **Fungi** | *Aspergillus fumigatus* | Aspergillosis |
| | *Candida albicans* | Fungal infections |
| **Protozoa** | *Balantidium coli* | Diarrhea, dysentery |
| | *Cryptosporidium parvum* | Diarrhea |
| | *Entamoeba histolytica* | Diarrhea, dysentery |
| | *Giardia lamblia* | Diarrhea |
| **Helminths** | *Ascaris lumbricoides* | Roundworm |
| | *Clonorchis sinensis* | Bile duct infection |
| | *Diphyllobothrium latum* | Fish tapeworm |
| | *Enterobius vericularis* | Pinworm |
| | *Fasciola hepatica* | Liver fluke |
| | *Fasciolopsis buski* | Intestinal fluke |
| | *Hymenolepis nana* | Dwarf tapeworm |
| | *Necator americanus* | Hookworm |
| | *Opisthorchis* spp. | Bile duct infection |
| | *Schistosoma* spp. | Schistosomiasis |
| | *Taenia* spp. | Tapeworm |
| | *Trichuris trichura* | Whipworm |

*Source:* Data from Krishnan and Smith (1987) In *Aquatic Plants for Water Treatment and Resource Recovery*. Reddy and Smith (Eds.), Magnolia Publishing, Orlando, Florida, pp. 855–878; Shiaris (1985) In *Ecological Considerations in Wetlands Treatment of Municipal Wastewaters*. Godfrey *et al*. (Eds.), Van Nostrand Reinhold, New York, pp. 243–261; Leclerc *et al.* (1977) In *Bacterial Indicators/Health Hazards Associated with Water*. Hoadley and Dutka (Eds.), American Society for Testing and Materials (ASTM), Philadelphia, pp. 23–36; Cabelli (1977) In *Bacterial Indicators/Health Hazards Associated with Water.* Hoadley and Dutka (Eds.) American Society for Testing and Materials (ASTM): Philadelphia, pp. 222–238; Metcalf and Eddy Inc. (1991) *Wastewater Engineering, Treatment, Disposal, and Reuse.* Tchobanoglous and Burton (Eds.), Third Edition, McGraw-Hill, New York; Crites and Tchobanoglous (1998) *Small and Decentralized Wastewater Management Systems.* McGraw-Hill, New York.

**TABLE 12.2**
**Indicator Microorganisms and Their Abundance in Feces and Raw Domestic Wastewater**

| | Water Concentration (CFU/100 mL) | Fecal Coliform Density (#/gram × $10^6$) |
|---|---|---|
| **Fecal Coliforms** | | |
| Human | $10^6$–$10^8$ | 13 |
| Pig | — | 3.3 |
| Cat | — | 7.9 |
| **Fecal Streptococci** | | |
| Human | $10^4$–$10^6$ | 3 |
| Pig | — | 84 |
| Cat | — | 27 |
| *Clostridium perfringens* | $10^3$–$10^5$ | — |
| *Pseudomonas aeruginosa* | $10^3$–$10^5$ | — |
| Salmonella | $10^2$–$10^4$ | — |
| *Cryptosporidium parvum* | 10–$10^3$ | — |
| *Giardia lamblia* | $10^3$–$10^4$ | — |
| *Ascaris lumbricoides* | 10–$10^3$ | — |
| Coliphages | $10^2$–$10^4$ | — |

*Source:* Data from Crites and Tchobanoglous (1998) *Small and Decentralized Wastewater Management Systems.* McGraw-Hill, New York; Maier *et al.* (2000) *Environmental Microbiology.* Academic Press, San Diego, California.

An even narrower group, *Escherichia coli*, is being used more frequently as an indicator organism, because it can readily be separated from the rest of the fecal group, and because several strains are capable of causing severe human health problems. However, as these bacteria also originate in other warm-blooded animals, *E. coli* is not diagnostic of human fecal contamination alone. The coliform groups in general have an indicator disadvantage in that they may regrow in aquatic environments.

The fecal streptococcus group is used to confirm the origins of fecal contamination. A high ratio of fecal coliform to fecal streptococcus (FC/FS > 4) is regarded as an indication of human origin, whereas a low ratio (FC/FS < 0.7) is indicative of animal pollution (Clausen *et al*., 1977). However, the validity of this separation has been subject to question (Gerba, 2000). Fecal streptococcus is found in the feces of humans and other warm-blooded animals including birds and mammals. These bacteria are found frequently in waters receiving fecal contamination and are not believed to multiply in natural or polluted waters and soils. Because fecal streptococci bacteria seem to survive longer than fecal coliforms in receiving waters, they are used as a second indicator of fecal contamination. Because bacterial die-off affects the ratio, it is only applicable to fecal pollution within 24 hours of discharge.

It is generally recognized that fecal coliforms are not suitable indicators of viral contamination in surface water receiving domestic waste because some viruses are more resistant to chlorination and environmental deactivation than bacteria (Kraus, 1977; Gersberg *et al*., 1987; Gerba, 2000). Bacteriophages (viruses that infect bacteria, such as coliphage MS-2) have been used as viral indicators in wetland treatment systems (Gersberg *et al*., 1987; Schuerman *et al*., 1989).

Enumeration of bacteriophages is technically simpler and more rapid than enumeration of the target pathogenic viruses. Also, MS-2 is nearly the same size as enteroviruses and is more resistant to ultraviolet (UV) light, heat, and disinfection than most enteric viruses. In a free water surface (FWS) study at San Jacinto, California, Chendorain *et al.* (1998) added cultured MS-2 virus to the wetland influent wastewater to determine die-off rates.

## 12.2 PATHOGEN REMOVAL PROCESSES

The observed net removal of pathogenic organisms in treatment wetlands is the result of a variety of processes, some that remove pathogens and some that introduce pathogens. Generally speaking, removal mechanisms are much more prevalent than reintroduction mechanisms, and most constructed wetlands demonstrate large net removals of pathogenic organisms down to a nonzero background concentration ($C^*$).

### Solar Disinfection

UV radiation is a potent agent for killing bacteria in FWS wetlands. An exponential relation is commonly used to describe the inactivation in disperse batch cultures:

$$N = N_o e^{(-k_i I t)} \tag{12.1}$$

where

$I$ = intensity of UV light in solution, J/m$^2$·d
$k_i$ = inactivation rate coefficient, m$^2$/J
$N_o$ = initial number of dispersed organisms
$N$ = surviving number of dispersed organisms
$t$ = time, d

However, most organisms are present in association with wastewater particulates, rather than in a dispersed phase (Emerick *et al.*, 2000). Consequently, UV inactivation rates are lowered from shielding of the organisms by the particle structures. Equation 12.1 is still applied, but rate constants are much lower than for dispersed-phase populations. Emerick *et al.* (2000) found little variation in rates for fecal coliforms across 11 treatment technologies, with $0.022 < k_i < 0.054$ m$^2$/J for activated sludge, trickling filter, aerated, and facultative pond effluents.

The UV wavelengths that are absorbed by microorganisms, and hence are effective in disinfection, comprise the range 240–280 nm (Crites and Tchobanoglous, 1998). Lamps used for in-line disinfection of wastewaters are designed to provide radiation in this range. The effectiveness of the radiation depends on water quality factors, such as optical absorbance and suspended solids content. Typical UV dosages are 100–850 J/m$^2$ to provide suitable disinfection of fecal coliforms to the range between 1,000 and 23 CFU/100 mL (Crites and Tchobanoglous, 1998).

In a natural system, such as a pond or an open-water wetland, sunlight provides a source of ultraviolet (UV) radiation. However, the fraction of the incoming solar radiation that is in the UV range is small, typically less than 1%. Therefore, solar inactivation rates based on total solar radiation are much smaller than those determined for UV lamp sources, and these are designated $k_S$. For example, Davies-Colley *et al.* (1999) present pond water data for *E. coli* for which $k_S \approx 0.7$ m$^2$/MJ, which is about 100,000 times lower than for a UV lamp. Salih (2003) determined a similar value for *E. coli*, $k_S \approx 1.5$ m$^2$/MJ. Davies-Colley *et al.* (1999) also demonstrated that dissolved oxygen and pH had important influences on rates of inactivation. These authors inferred that both direct hit radiation damage and photooxidation were caused by incoming radiation.

The overall solar inactivation rate for a shallow water body (pond) is further reduced by the lack of penetration of light (Mayo, 1995). The top few centimeters of a stabilization pond get the required radiation, but deeper sections do not. Accordingly, the solar inactivation rate constant is approximately inversely proportional to the pond depth:

$$k_S = \frac{k'_S}{K_L h} \tag{12.2}$$

where

$h$ = water depth, m
$k_S$ = overall solar inactivation rate coefficient, m$^2$/J
$k'_S$ = intrinsic solar inactivation rate coefficient, m$^2$/J
$K_L$ = light attenuation coefficient, m$^{-1}$

Mayo (1995) reported values of $0.012 < hk_S < 0.016$ m$^3$/MJ for five stabilization pond studies. Additionally, high-rate ponds (0.2–0.3 m depth and a few days' detention) have been shown to have $k_S = 0.0675$ m$^2$/MJ (Davies-Colley *et al.*, 2003) and $k_S = 0.083$ m$^2$/MJ (Craggs *et al.*, 2004).

Bacteriophages are also susceptible to UV radiation. Jolis (2002) gives a range of inactivation rate coefficients of $0.0054 < k_i < 0.0106$ m$^2$/J for irradiation from UV lamp sources. This slower rate is reflected in higher UV lamp dosages needed to inactivate viruses, typically six to eight times higher than for coliforms (Crites and Tchobanoglous, 1998). As for bacteria, natural systems receive solar radiation that contains small amounts of UV, which is effective in inactivation of bacteriophages. Davies-Colley *et al.* (1999) provide pond data for an F-DNA phage ($k_S \approx 0.7$ m$^2$/MJ) and for an F-RNA phage ($k_S \approx 0.6$ m$^2$/MJ). The inactivation of phages by sunlight has been found to be a good rate analog for the inactivation of human pathogenic polio, echo, and coxsackie viruses (Fujioka and Yoneyama, 2002).

Solar disinfection depends on the sunlight reaching and penetrating into the water column. Dense vegetation intercepts sunlight in the wetland environment and diminishes the potential for solar disinfection. This negative effect is putatively present for emergent and floating plant communities. It may or may not be present in submersed algal or macrophyte communities where subsurface oxygenation may occur. The presence of enhanced amounts of dissolved

oxygen (DO) fosters the photooxidation route of disinfection and may compensate for reduced penetration of UV radiation (Davies-Colley *et al*., 1999; Sun *et al*., 2003).

However, when pond performance is compared to vegetated wetland performance for fecal coliform reduction across large numbers of both types of systems receiving varying numbers of pathogens, little difference can be seen in the general level of reduction (Kadlec, 2005e). Other mechanisms of organism mortality and removal, beyond just solar disinfection, become operative in the wetland environment.

## Predation

Most pathogens are food for nematodes, rotifers, and protozoa (Decamp and Warren, 1998). Among these, rotifers and flagellated and ciliated protozoa have been implicated as important contributors to the reduction of bacteria in treatment wetlands (Panswad and Chavalprit, 1997; Laybourn-Parry *et al*., 1999; Proakis, 2003; Stott and Tanner, 2005) as well as in natural aquatic systems (Menon *et al*., 2003). While pathogenic organisms span a wide size range (0.2–100 μm), so do the associated predator/grazing communities (Figure 12.1). They are found in secondary wastewaters in considerable numbers; for instance, Fox *et al*. (1981) reported up to about 50 rotifers per liter in such effluents (Figure 12.2).

Proakis (2003) determined that rotifers consumed enterococci at the rate of about 600/h when the bacterial density was approximately $10^6$ per 100 mL. From this rate, it was determined that rotifers, at a density of ten per mL, could suitably disinfect stormwater in a 1.2-hour detention in a marsh. The next question, then, is about concentrations of predators that can be expected in the wetland waters.

Decamp and Warren (1998) found that the ciliate *Paramecium* consumed more than 100 *E. coli* per hour when the bacterial density was approximately $10^8$ per 100 mL. This was interpreted to require 20 *Paramecium* per mL to remove about $2 \times 10^6$ *E. coli* per 100 mL in eight hours' detention. This concentration of *Paramecium* was comfortably within the observed concentration range in the SSF wetland from which the protozoa were taken.

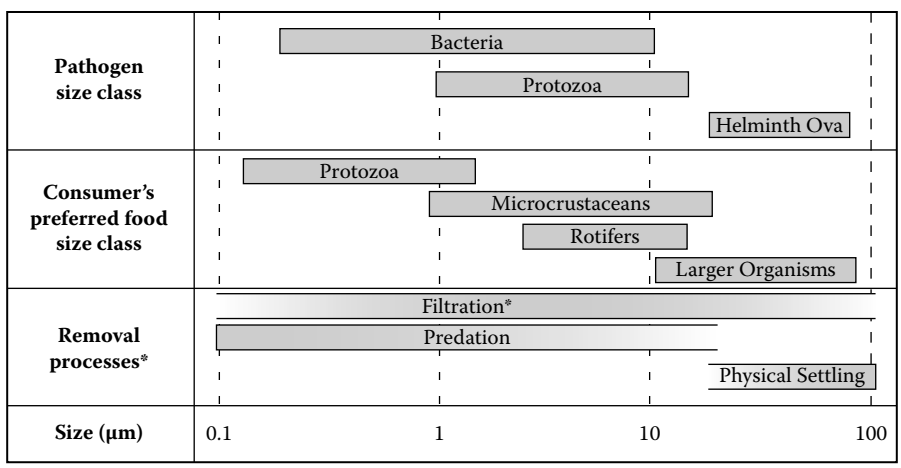
Heterotrophic nanoflagellates (HNAN) were found to dominate the protozoan community in both lagoons and grass filters treating sewage (Laybourn-Parry *et al*., 1999). Ciliates contributed to only about 10% of the grazer populations. The grass filters harbored more numbers (approximately $10^8$ per 100 mL) of these bacterial predators than the lagoons (approximately $10^7$ per 100 mL).

The overall effect of grazing on bacterial numbers in aquatic systems was quantified by Menon *et al*. (2003). Grazing by protozooplankton was responsible for more than 90% of the overall mortality rate of both fecal and autochthonous bacteria in the river Seine. For example, whereas in the presence of protozoa the overall first-order volumetric mortality coefficient was $34 \times 10^{-3}$ h$^{-1}$, it was $2 \times 10^{-3}$ h$^{-1}$ in the absence of protozoa.

Grazing is generally higher at higher temperatures (Laybourn-Parry *et al*., 1999; Menon *et al*., 2003), but temperature coefficients for wetland environments are not available. The aforesaid studies indicate a strong potential for pathogen removal through predation in constructed wetlands. They do not, unfortunately, allow quantification of this removal mechanism. Whereas predation/grazing is an important removal mechanism (especially in FWS wetlands), it is not the only removal mechanism. Settling and filtration also play important roles in pathogen reduction, and these removal mechanisms have a cumulative effect in treatment wetland systems.

## Settling and Filtration

A measurable proportion of wastewater microorganisms are found either associated with particulates or as aggregates of




*Note: Filtration processes are system dependent.

**FIGURE 12.1** Pathogen and predator size chart.

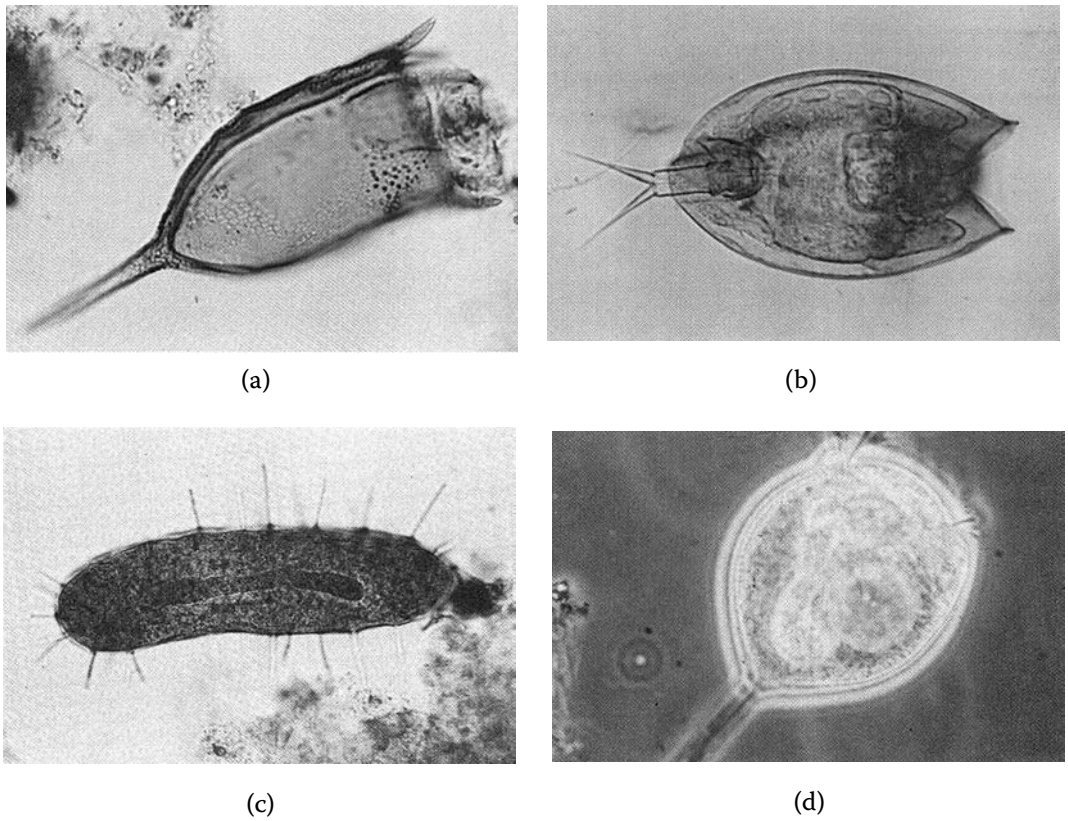



**FIGURE 12.2** (a) Rotifer *Keratella* spp. 200 µm. (b) Rotifer *Lepadella* spp. 100 µm. (c) Protozoan *Discophrya* spp. 80 µm. (d) Protozoan *Vorticella* spp. 65 µm. (From Fox *et al.* (1981) *Sewage Organisms: A Color Atlas.* Lewis Publishers, Boca Raton, Florida. Reprinted with permission.)

many organisms. For the activated sludge process, the fraction of particles with associated coliform ranges from <1% to 24% (Loge *et al.*, 2002). This fraction declines with the residence time (mean cell residence time) of organics in the activated sludge process. The loss of coliform organisms inside the activated sludge process has been speculatively attributed to predation by protozoans (Loge *et al.*, 2002). Organisms associated with particles are far less susceptible to UV or solar disinfection, presumably because of shielding effects (Emerick *et al.*, 2000).

Bacteria may clump together to form aggregates. These are highly amorphous and porous, but the internal voids are apparently filled with exopolymeric material (Li and Yuan, 2002). Sedimentation of these particles contributes to their removal from the water column, with removals of 25–75% due to gravitational sedimentation in conventional treatment plants (Metcalf and Eddy Inc., 1991). Tests of algal settling ponds produced 40% removal of *E. coli* by sedimentation (Davies-Colley *et al.*, 2003).

In the wetland environment, submersed plant parts and their associated biofilms form "sticky traps" for particles, including all sizes of microorganisms (see Chapter 7). These biofilms are capable of trapping considerable numbers of organisms (Flood and Ashbolt, 2000; Stott and Tanner, 2005). Such biofilms are enhanced in quantity by larger quantities of submersed surfaces and exposure to light. There may be an optimal plant density that allows light and provides the necessary surfaces for biofilm growth.

### Mortality and Regrowth

Bacteria, protozoa, helminths, and viruses typically do not survive longer than about 30 days in freshwater environments and about 50 days in soil environments (Crites and Tchobanoglous, 1998). Similar survivals might therefore be predicted for wetlands, but there are many site-specific factors and processes which may materially increase or decrease survival.

A part of the overall removal of pathogens consists of the death or inactivation of organisms for reasons other than radiation damage or predation. This has been called the *dark death rate* (Khatiwada and Polprasert, 1999b). Mortailty has been found to be approximately 25% of the overall removal rate in ponds (Craggs *et al.*, 2004), with a similar fraction in a *Typha* FWS wetland in Thailand (Khatiwada and Polprasert, 1999b). However, care must be taken to separate sedimentation and filtration from mortality in this dark death rate. For instance, Khatiwada and Polprasert (1999b) estimated that only 6.5% of the overall fecal coliform removal rate was due to temperature-modulated death of organisms.

There is no definitive study of the temperature effect on mortality of bacteria in wetlands. Estimates of temperature

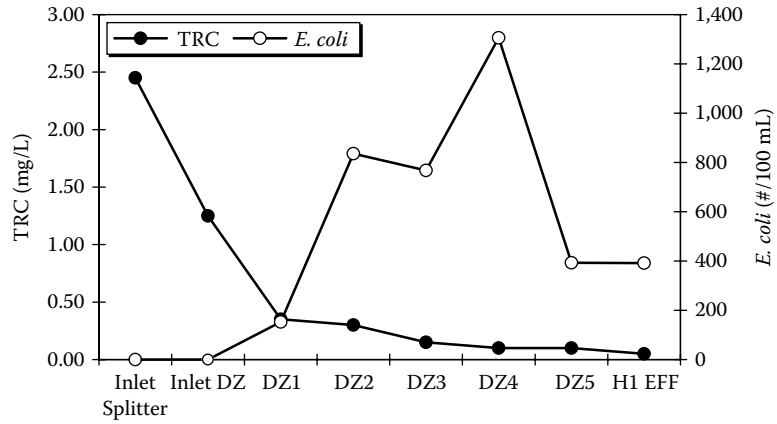



**FIGURE 12.3** Longitudinal profiles of total residual chlorine (TRC) and *E. coli* through Tres Rios Hayfield, Arizona, wetland H1.

coefficients (θ) for ponds range from 1.000 (no effect) to 1.070 (strong effect).

Very low temperatures may result in freezing conditions. Ice formation followed by a thaw drastically reduces the populations of pathogens in water. For instance, Torrella *et al*. (2003) found that only 1.2% of fecal coliforms survived four days of freezing, but more coliphages survived. However, care must be taken in enumeration, because organisms may be viable after freezing, but unculturable in routine tests (Parker and Martel, 2002). Further, many bacteria survive the snowmaking process. Less than one log loss was measured for fecal streptococcus due to snow formation and melting (Parker *et al*., 2000).

To complicate matters even more, some indicator organisms are capable of regrowing and multiplying in an aquatic environment. All members of the coliform group have been observed to regrow in natural surface water (Gerba, 2000). Such regrowth is fostered by high concentrations of organic matter and by elevated temperatures. This phenomenon is recognizable when disinfected wastewater is introduced into treatment wetlands, and bacterial populations increase along the direction of flow (Figure 12.3).

## Reintroduction

Indicator organisms can be produced by sources other than incoming wastewaters. This is particularly true for the coliform group, which may originate from many different warm-blooded animals that frequent wetlands. As an example, consider the performance of the Orlando Easterly FWS Wetlands (Figure 12.4). Incoming fecal coliforms are very low, but animal (bird) populations are high, especially near the outlet. Background geometrical annual averages range from 50 to 142 per 100 mL. Monthly values show an exceedance frequency of 40% of the permit limit of 100 CFU/100 mL. The five-year median fecal coliform was 64 CFU per 100 mL and FS was 106 CFU per 100 mL. The ratio FC/FS was 0.6; thus, the presumption may be made that the source of contamination is animal rather than human (criterion = FC/FS < 0.7). In this case, the treatment wetland has been measured to have very large bird populations, which also supports the concept of reintroduction of pathogens from avian populations (U.S. EPA, 1993a; U.S. EPA, 1993d).

These naturally-caused bacterial populations are generally low, but they may be variable and seasonally high because of wildlife activity patterns. Total and coliform bacteria have been measured in natural wetlands that receive no wastewater. For example, Fox *et al*. (1984) measured between 109 and 456 CFU/100 mL of fecal coliforms in cypress wetlands in Florida that were not receiving any external water inputs. Fecal coliform concentration in lake water passing through a wetland in Montreal, Quebec, increased from 40 to 110 CFU/100 mL (Vincent, 1992). Because natural sources of coliforms and fecal streptococcus bacteria are found in all wetlands open to wildlife, outflow indicator bacteria populations in treatment wetlands cannot be consistently reduced to near zero unless disinfection is used.

Subsurface flow (SSF) wetlands provide much less wildlife habitat than FWS systems; however, their use by wildlife cannot

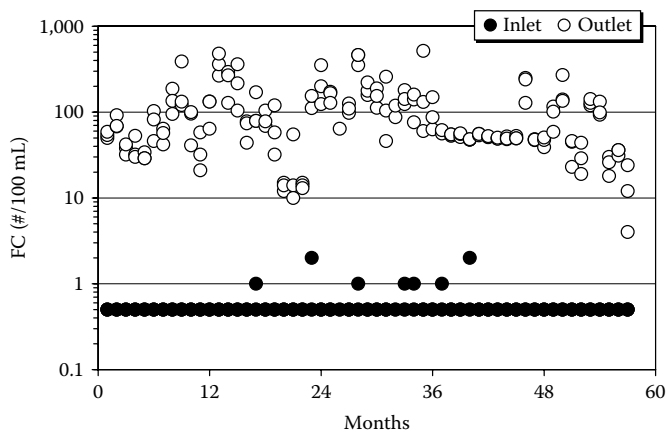



**FIGURE 12.4** Fecal coliforms entering and leaving the Orlando Easterly Wetland, in Florida during 1993–1997. The incoming wastewater is disinfected, with only a few detections of the indicator organisms. The balance of the nondetect samples are plotted as half the detection limit of 1.0 per 100 mL. (Unpublished data from city of Orlando.)

be excluded. For instance, a HSSF wetland in Arizona, fed with disinfected potable water, produced on an average, effluent concentrations of 130 total coliforms (CFU/100 mL) and 22.3 fecal coliforms (CFU/100 mL). This reintroduction of indicator organisms was attributed to wildlife use of the system. Because wildlife use of SSF wetlands has been observed on many occasions (Wallace *et al.*, 2001), it is to be expected that these treatment wetlands will also deliver a nonzero background concentration ($C^*$) for indicator organisms. However, wildlife use is so restricted that backgrounds are likely of little consequence.

## 12.3 FECAL COLIFORM REMOVAL IN FWS WETLANDS

The most common indicator organism group is the fecal coliform group and consequently the wetland database is largest for this group. Further, several attempts have been made to extract quantitative generalities about the removal of this indicator group in FWS treatment wetlands.

### First-Order Removal Models

The complexity of the suite of processes responsible for reducing indicator bacteria in FWS wetlands is considerable, with the many interacting processes described previously. Although some attempts have been made to combine these into a single removal model (Mayo, 1995; Bahlaoui *et al.*, 1998; Khatiwada and Polprasert, 1999b), data have been insufficient to provide robust calibrations of combination models. Information on light exposure and penetration, vegetative cover, and predator density is typically lacking. Therefore, at this point in the evolution of treatment wetland technology, only very simple global removal models may be considered.

### Model Structure

First-order models have been used to describe reductions of indicator bacterial populations in lagoons and wetlands:

$$\text{First-order (volumetric)} \quad \left(\frac{C_o - C^*}{C_i - C^*}\right) = \left(1 + \frac{k_V \tau}{N}\right)^{-N} \qquad (12.3)$$

$$\text{First-order (areal)} \quad \left(\frac{C_o - C^*}{C_i - C^*}\right) = \left(1 + \frac{k_A \tau}{Nh}\right)^{-N} \qquad (12.4)$$

$$\text{Plug-flow limit} \quad \left(\frac{C_o - C^*}{C_i - C^*}\right) = \exp\left(-\frac{k_A}{q}\right) \qquad (12.5)$$

where

$C_i$ = inlet concentration, #/100mL
$C_o$ = outlet concentration, #/100mL
$C^*$ = background concentration, #/100mL
$h$ = water depth, m
$k_A$ = areal rate coefficient, m/d
$k_V$ = volumetric rate coefficient, $d^{-1}$
$N$ = number of TIS
$\tau$ = nominal detention time, d

The first question that arises is whether to use Equation 12.3 or Equation 12.4, the former being volume-specific and the later being area-specific. In the situation of unvegetated systems, von Sperling and Mascarenhas (2005) provide an unequivocal answer: no difference was found in doubling the depth (80 cm versus 40 cm) and thus doubling the detention time for *E. coli* removal. The implications are that $k_V$ is inversely proportional to depth and, hence, the areal rate coefficient in Equation 12.4 is constant. The same inverse proportion is found for the Arcata, California, pilot data, in which six pairs of emergent FWS wetland cells were run at two depths and different hydraulic loadings (data from Gearheart *et al.*, 1983). Thus, for both ponds and wetlands, there is strong evidence that the areal rate coefficient ($k_A$) is constant, whereas the volumetric rate coefficient ($k_V$) decreases with increasing depth. In this book, the areal rate coefficient will be used and presumed constant over some modest depth range, such as 20–80 cm. However, FWS volumetric rate coefficients, when they are reported elsewhere, may be easily converted to an areal basis simply by dividing by the free water depth.

A far more serious question unique to pathogen removal is the internal pattern of the flow (number of tanks in series, $N$) to be assigned to the system for which $k$ is to be determined or used. For other measures of wetland water quality improvement, such as biochemical oxygen demand (BOD), total suspended solids (TSS), and nutrient removal, reductions are rarely greater than 90–95% (less than one $\log_{10}$ removal). For pathogens, reductions often reach 99.99% (two to three $\log_{10}$ removal) or more. This is precisely the region of performance in which reductions are extremely sensitive to the number of tanks-in-series ($N$TIS) value (see Chapter 6). Equation 12.4 has been graphed for the case of a negligibly small $C^*$ ($C^* \ll C_o$) in Figure 12.5.

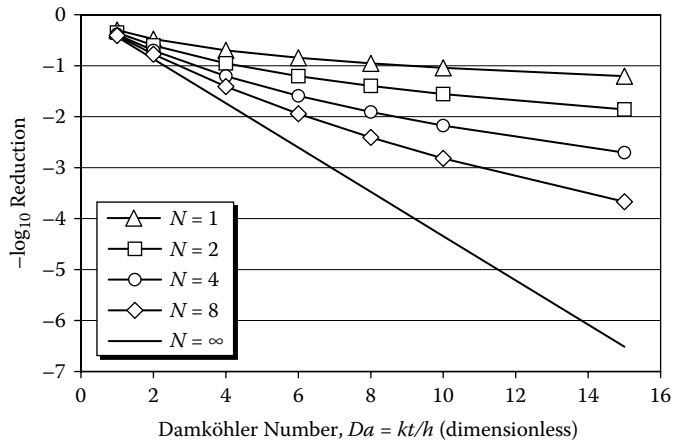



**FIGURE 12.5** Reductions according to a first-order model in the high removal range.

The Damköhler number ($Da = k_A\tau/h$) embodies the intrinsic rate of removal together with the detention time and depth. Independent of these factors, the wetland may have differing degrees of hydraulic efficiency (see Chapter 2), as indicated by different values of *N*. Figure 12.5 indicates that the number of $\log_{10}$ reductions increases steeply with number of tanks in series (*N*) at any given detention time (Damköhler number).

The underlying phenomenon is hydraulic short-circuiting. Small elements of flow can carry enough organisms to the outlet, without treatment, to jeopardize the high levels of removal that prevail for the remainder (majority) of the flow. Conversely, if a wetland exhibits performance of, say, two $\log_{10}$ reduction, the rate coefficient is extremely sensitive to whether the *N*TIS value is 4, 8, or ∞ (plug flow); the computed areal rate coefficient ($k_A$) changes by approximately 100% if $N = 4$ as opposed to $N = \infty$ (plug flow) (Figure 12.5).

Data from waste stabilization ponds provides confirmation of this phenomenon. Lloyd *et al*. (2003) studied fecal coliform removal in a sequence of pond modifications, each designed to improve the hydraulic efficiency. The levels of contamination were in the range of $10^4$–$10^6$ fecal coliforms per 100 mL, outlet to inlet; thus, in this instance, *C** is negligible. The results may be summarized as follows:

| | | |
|---|---|---|
| Open pond, aspect 4:1 | 90.26% | (1.01 $\log_{10}$) |
| Channelized, aspect 36:1 | 96.03% | (1.42 $\log_{10}$) |
| Channelized, aspect 36:1, wind break | 98.13% | (1.78 $\log_{10}$) |
| Batch reduction (plug-flow equivalent) | 99.7% | (2.6 $\log_{10}$) |

Although tracer tests were not performed to determine the number of tanks in series, it is clear that the increasing aspect ratio and decreasing wind mixing had very large effects on the performance of the pond. These results correspond nicely to the theoretical projections of Figure 12.5 for a Damköhler number of about 6.0. From these considerations, it may be concluded that the reduction of pathogens in FWS wetlands is critically dependent on the internal flow patterns.

## Longitudinal Profiles

Irrespective of the degree of hydraulic efficiency in a particular wetland, global first-order models forecast a decline in pathogens through the wetland, provided the incoming levels exceed the regrowth and reintroduction potentials. This theory corresponds to observations for a number of wetlands, such as the Byron Bay, Australia, FWS system (Figure 12.6). There is a drop of about two $\log_{10}$ over the first third of that system, followed by a plateau at levels of 300–2,000 fecal coliforms per 100 mL, depending upon the year in question. Avian use of the outlet portions of the system may account for the plateaus. For most FWS wetland systems, complete elimination of fecal coliforms has not been achieved, and therefore the inclusion of a *C** value in the first-order model is a necessity.

Because of residual indicator bacteria populations in all wetlands, bacteria removal efficiency is a function of the inflow bacteria population. Removal efficiency is typically high at high-inflow populations but declines to negative efficiencies when inflow populations are lower than the *in situ* bacteria production and addition rates. For instance, at the Titusville, Florida, wetland treatment system, inflow to the wetland contains essentially no indicator bacteria because of a high level of pretreatment and disinfection (less than 1 CFU/100 mL); however, the wetland outflow contains small numbers of fecal coliforms (63 CFU/100 mL) and fecal streptococci (170 CFU/100 mL), presumably because of wetland bird use (unpublished data from Titusville, Florida).

## Implications for Design

There are two important points to be kept in mind when extracting information from the literature for use in design: (1) literature die-off rates are often computed using the plug-flow equivalent of Equations 12.3 and 12.4, and (2) the use of that plug-flow formulation in design in an extrapolation mode is extremely dangerous, because it creates large overestimates of reductions.

Data from 28 FWS systems, totaling 47 wetland years, was used to estimate fecal coliform areal rate coefficients ($k_A$) for a presumed $C^* = 40$ CFU/100 mL. These systems were selected for having inlet fecal coliform concentrations of at least 1,000 CFU/100 mL, thus eliminating systems with low influent fecal coliform concentrations, whether due to pretreatment disinfection or other factors. The hydraulic parameter *N*, representing the number of tanks in series, was relaxed to the more general case of *P*, which accounts not only for the nonideal hydraulics but also treatment effects as well ($P < N$), as discussed in Chapter 6. Because the critically important *P*TIS values were not known, three different assumptions were made:

*P*TIS = ∞ (plug flow; known to be wrong, but in common use)
*P*TIS = 3 (a modest degree of departure from both plug flow and complete mixing)
*P*TIS = 1 (complete mixing)

The resulting annual areal rate coefficients were

| | | |
|---|---|---|
| *P*TIS = ∞ | $k_A \pm$ s.e. = 85 ± 17 m/yr | (median = 40 m/yr) |
| *P*TIS = 3 | $k_A \pm$ s.e. = 288 ± 73 m/yr | (median = 83 m/yr) |
| *P*TIS = 1 | $k_A \pm$ s.e. = 16,300 ± 9,500 m/yr | (median = 1,030 m/yr) |

It is seen that the presumed degree of nonideal flow has a very large effect on the calibrated areal rate coefficients, as has been theoretically explained previously. This wide disparity exists for each individual wetland as well as for the means and medians of the N = 28 dataset.

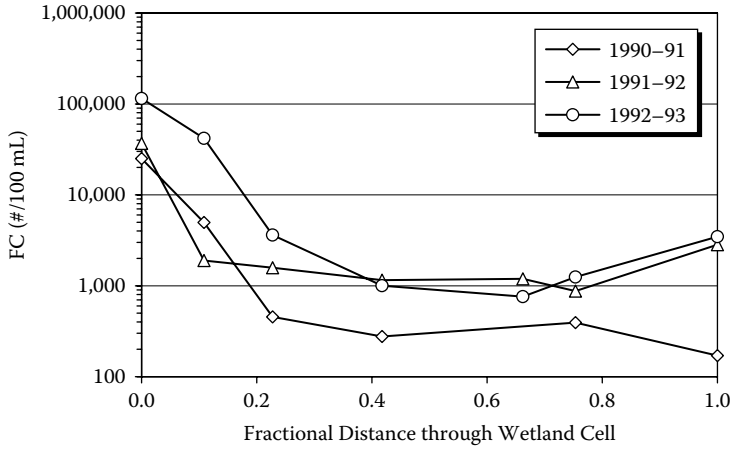



**FIGURE 12.6** Average annual longitudinal profiles of fecal coliforms in the Byron Bay, Australia, treatment wetlands. (Unpublished data from Byron Bay operations.)

The danger of extrapolation is seen in an example for which we assume $C^* = 0$ for simplicity. Suppose that the given wetland is producing 1.0 $\log_{10}$ reduction at a hydraulic loading rate of 10 cm/d and the wetland hydraulics can be characterized by $P$TIS = 3. The plug flow $k_A = 84$ m/yr (areal die-off coefficient) (Equation 12.5), and the 3TIS $k_A = 126$ m/yr (Equation 12.4). It is desired to improve performance by lowering the hydraulic loading rate to 2 cm/d. According to the plug flow model (Equation 12.5), performance should increase to 5.0 $\log_{10}$ reduction. But, according to the 3TIS model (Equation 12.4), performance would increase to only 2.49 $\log_{10}$ reduction. Thus, even though the plug flow rate coefficient is typically considered to be "conservative," it is in error by more than a factor of 300.

It is concluded that first-order modeling is not useful in design unless the hydraulic characterization of the wetland is taken into account.

Nonetheless, it is appropriate to explore the rates of reduction of pathogens in FWS systems, and here the indicator fecal coliforms is used to identify the intersystem variability. Table 12.3 shows the frequencies of reductions and areal $k$-values for a set of FWS wetlands that are in the mode of removal, i.e., they receive a load of organisms at or above the expected background, here taken to be 40 CFU/100 mL. For a presumed $P$TIS = 3, the median reduction is 1.54 $\log_{10}$, and the median $k = 83$ m/yr.

## Seasonal and Stochastic Effects

There are no pronounced seasonal or temperature effects for pathogen removal in the FWS treatment wetland datasets that are currently available. Computation of cosine trends in outlet concentrations typically produces a low amplitude, about 12% in $\log_{10}$, with the annual maximum in September (Table 12.4). Ten years' data from Estevan, Saskatchewan, showed an increase in fecal coliforms in the wetland (0.26 $\log_{10}$ increase), but a decrease in total coliforms (2.01 $\log_{10}$ reduction). However, the seasonal trend for both displays higher summer values (Figure 12.7).

### *Temperature Coefficients*

There are conflicting potential effects of temperature on pathogen reduction. Cold temperatures are inimical to

**TABLE 12.3**
**Annual Reduction of Fecal Coliforms in FWS Wetlands**

**Stipulations**

1. Annual averages are used in calculations.
2. For $k$-value calculations, the following $P$-$k$-$C^*$ parameters are selected:
   a. $C^* = 40$ #/100 mL
   b. $P = 3$ TIS
3. Ranges of variables:

| | HLR (cm/d) | $\log_{10}$ FC In (CFU/100 mL) | $\log_{10}$ FC Out (CFU/100 mL) |
|---|---|---|---|
| **Mean** | 3.8 | 5.21 | 3.28 |
| **Median** | 4.6 | 4.76 | 3.25 |
| **Max** | 24.3 | 6.89 | 5.43 |
| **Min** | 0.2 | 3.02 | 1.14 |

**Results (N = 47 wetland·years for 23 wetlands).**

| Percentile | $\log_{10}$ Reduction (CFU/100 mL) | Rate Constant (m/yr) |
|---|---|---|
| 0.05 | –0.16 | 11 |
| 0.10 | 0.17 | 27 |
| 0.20 | 0.64 | 43 |
| 0.30 | 1.00 | 49 |
| 0.40 | 1.31 | 68 |
| 0.50 | 1.54 | 83 |
| 0.60 | 1.77 | 115 |
| 0.70 | 1.96 | 177 |
| 0.80 | 2.26 | 321 |
| 0.90 | 2.72 | 856 |
| 0.95 | 3.12 | 1,365 |

**TABLE 12.4**
**Annual Cyclic Trends and Trend Multipliers for the $\log_{10}$ of Concentration (CFU/100 mL) of Pathogenic Bacteria in the Effluent from FWS Wetlands**

| System | Years | $\log_{10}$ Mean | $\log_{10}$ Amplitude | Yearday Maximum | $R^2$ | Excursion Frequency 80% | 90% | 95% | 99% |
|---|---|---|---|---|---|---|---|---|---|
| **Fecal Coliforms** | | | | | | | | | |
| Estevan, Saskatchewan | 10 | 0.46 | 1.55 | 230 | 0.493 | 1.00 | 1.62 | 1.93 | 2.37 |
| Imperial, California | 4 | 2.00 | 0.17 | 223 | 0.247 | 1.01 | 1.21 | 1.30 | 1.36 |
| Brawley, California | 4 | 1.99 | 0.19 | 234 | 0.203 | 1.00 | 1.24 | 1.32 | 1.50 |
| Boggy Gut, North Carolina | 5 | 2.67 | 0.08 | 266 | 0.082 | 0.98 | 1.16 | 1.25 | 1.33 |
| Kohukohu, New Zealand[a] | 10 | 2.87 | 0.11 | 265 | 0.434 | 0.98 | 1.23 | 1.29 | 1.31 |
| Bennett, Colorado | 4 | 2.59 | 0.17 | 176 | 0.170 | 1.06 | 1.23 | 1.34 | 1.36 |
| Delta, Colorado | 4 | 2.52 | 0.11 | 166 | 0.057 | 1.02 | 1.24 | 1.33 | 1.38 |
| La Veta, Colorado | 4 | 2.84 | 0.11 | 247 | 0.233 | 0.98 | 1.11 | 1.20 | 1.27 |
| Ouray, Colorado | 10 | 2.88 | 0.12 | 231 | 0.095 | 1.04 | 1.21 | 1.33 | 1.35 |
| Silt, Colorado | 4 | 2.60 | 0.28 | 142 | 0.299 | 1.01 | 1.23 | 1.44 | 1.53 |
| Augusta, Georgia | 5 | 1.16 | 0.08 | 315 | 0.027 | 0.96 | 1.36 | 1.53 | 1.76 |
| **Total Coliforms** | | | | | | | | | |
| Estevan, Saskatchewan | 10 | 2.62 | 0.29 | 217 | 0.664 | 1.03 | 1.19 | 1.29 | 1.39 |
| Imperial, California | 4 | 3.02 | 0.07 | 211 | 0.068 | 1.06 | 1.11 | 1.20 | 1.26 |
| Brawley, California | 4 | 3.17 | 0.02 | 318 | 0.010 | 1.01 | 1.11 | 1.23 | 1.28 |
| ***E. coli*** | | | | | | | | | |
| Imperial, California | 4 | 1.86 | 0.14 | 224 | 0.202 | 1.04 | 1.19 | 1.25 | 1.31 |
| Brawley, California | 4 | 1.58 | 0.25 | 233 | 0.382 | 1.01 | 1.18 | 1.33 | 1.37 |
| Brighton, Ontario | 6 | 2.10 | 0.12 | 53 | 0.039 | 0.94 | 1.49 | 1.62 | 1.66 |
| **Median** | **4.0** | **2.61** | **0.12** | **231** | **0.19** | **1.01** | **1.22** | **1.31** | **1.36** |
| **Mean** | **5.9** | **2.39** | **0.24** | **232** | **0.22** | **1.01** | **1.23** | **1.36** | **1.46** |

*Note:* The multipliers that represent the frequency of excursions from the trend are also shown. For instance, the 99th percentile median $\log_{10}$ of concentration is 1.36 times higher than the trend. If the outlet trend value is 1,000 per 100 mL, then one time out of twenty the outlet may reach $1{,}000^{1.36} = 12{,}022$ per 100 mL. Trend multiplier is $(1 + \Psi)$; see Equation 6.61.

[a] Date adjusted 182 days

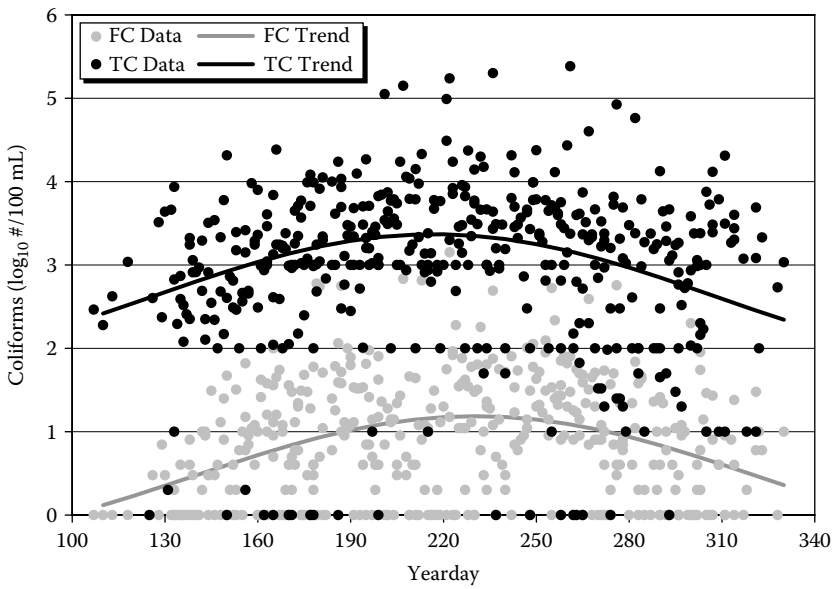



**FIGURE 12.7** Folded time series of $\log_{10}$ fecal and total coliforms at the Estevan, Saskatchewan, FWS wetlands. The wetlands operate during the unfrozen season. Note a midsummer peak in both. Fecal coliforms showed an increase of 0.26 $\log_{10}$, whereas total coliforms decreased 2.01 $\log_{10}$. (Unpublished data from city of Estevan.)

**TABLE 12.5**
**Temperature Coefficients for First-Order Plug Flow *k*-values for FWS Wetlands**

| System | Vegetation | HLR (cm/d) | T Range (°C) | $\log_{10}$ Inlet (CFU/100mL) | $\log_{10}$ Outlet (CFU/100 mL) | Annual Areal *k* (m/yr) | Theta |
|---|---|---|---|---|---|---|---|
| **Fecal Coliforms** | | | | | | | |
| Listowel 1, Ontario | *Typha* | 2.72 | 1–18 | 3.24 | 1.78 | 35 | 0.972 |
| Listowel 2, Ontario | *Typha* | 2.77 | 1–18 | 3.24 | 2.66 | 16 | 0.957 |
| Listowel 3, Ontario | *Typha* | 1.99 | 1–18 | 3.24 | 1.73 | 28 | 0.966 |
| Listowel 4, Ontario | *Typha* | 1.86 | 1–18 | 5.35 | 2.10 | 51 | 0.997 |
| Listowel 5, Ontario | *Typha* | 2.46 | 1–18 | 5.35 | 3.36 | 41 | 1.002 |
| Richmond, New South Wales | Open water | 6.41 | 10–26 | 6.13 | 4.22 | 93 | 1.011 |
| Richmond, New South Wales | *Myriophyllum* | 7.34 | 10–26 | 6.13 | 4.26 | 107 | 1.030 |
| Imperial, California | *Schoenoplectus* | 6.77 | 10–31 | 3.71 | 2.00 | 275 | 0.963 |
| Brawley, California | *Schoenoplectus* | 8.87 | 10–31 | 4.95 | 1.99 | 242 | 0.951 |
| **Total Coliforms** | | | | | | | |
| Imperial, California | *Schoenoplectus* | 6.77 | 10–31 | 4.57 | 3.02 | 248 | 0.956 |
| Brawley, California | *Schoenoplectus* | 8.87 | 10–31 | 5.37 | 3.17 | 178 | 0.958 |
| ***E. coli*** | | | | | | | |
| Imperial, California | *Schoenoplectus* | 6.77 | 10–31 | 3.05 | 1.88 | 189 | 0.960 |
| Brawley, California | *Schoenoplectus* | 8.87 | 10–31 | 4.70 | 1.61 | 257 | 0.953 |
| **Mean** | | | | | | | **0.975** |
| **Median** | | | | | | | **0.963** |

*Note:* For all but Listowel 1, 2, and 5 and Richmond Open Water, there is evidence that near-plug-flow prevailed.

the survival of organisms that originate in warm-blooded organisms, and thus human pathogens usually do not survive well outside the body. Survival times in waters are typically 10–30 days (Crites and Tchobanoglous, 1998). On the other hand, the activity of predatory organisms, such as protozoa, is lessened at cold temperatures, although direct lethality is unlikely (Davies-Colley *et al*., 2005). Removal by adsorption is expected to be insensitive to temperature but slightly less effective under cold conditions. The very high temperature coefficients suggested for ponds and FWS wetlands ($\theta = 1.19$) (U.S. EPA, 2000a; Shilton and Mara, 2005) are unlikely to prevail in the subsurface environment and do not apply. A number of FWS wetlands have a median of $\theta = 0.963$ (Table 12.5).

For example, the reductions in fecal coliforms for Listowel Systems 4 and 5 show slight but opposite trends with temperature, with a low degree of variability explained (low $R^2$) (Figure 12.8). System 4 shows $\theta = 0.985$, whereas System 5 shows $\theta = 1.021$, where $\theta$ is the modified Arrhenius temperature coefficient. Part of these trends are due to flow changes

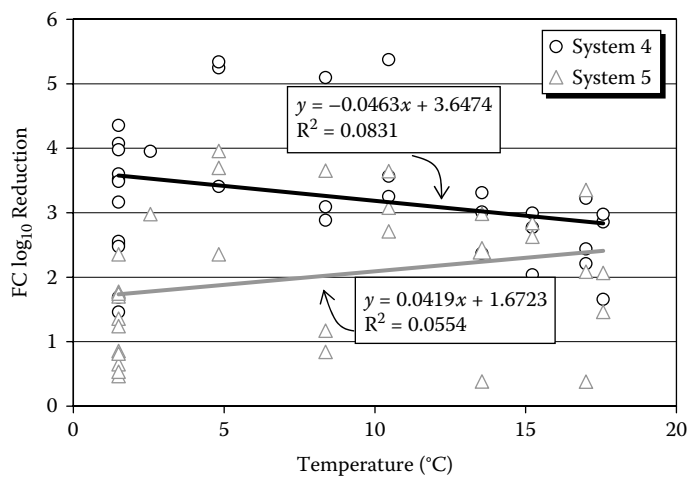



**FIGURE 12.8** Dependence of $\log_{10}$ reduction of fecal coliform for Listowel, Ontario, Systems 4 and 5. Three years of monthly data are represented. (Data from Herskowitz (1986) *Listowel Artificial Marsh Project Report*. Ontario Ministry of the Environment, Water Resources Branch: Toronto, Ontario.)

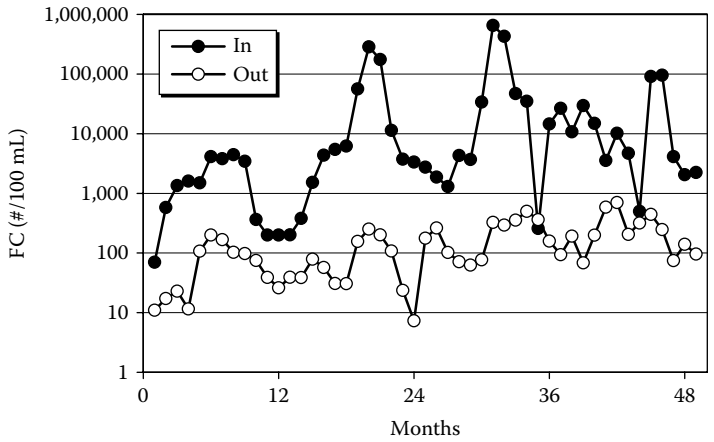



**FIGURE 12.9** Time series of monthly fecal coliforms entering and leaving the Imperial, California, treatment wetlands. (From unpublished data.)

from summer to winter, and thus the rate constants exhibit lesser temperature dependence ($\theta = 0.997$ and 1.002, respectively, Table 12.5).

Stochastic variability in the bacterial indicator organisms leaving FWS wetlands is very large. Fecal coliform counts are due to internal loading within the wetland as well as declines in populations entering with wastewater. For example, at the Imperial, California, wetland, the four-year time series of monthly incoming fecal coliforms varied over three orders of magnitude (factor of 1,000) and the wetland effluent fecal coliforms varied over two orders of magnitude (factor of 100) (Figure 12.9). Such time series may be expressed as probability density plots, such as those for Listowel Systems 4 and 5 (Figure 12.10). For these side-by-side wetlands, the variability of fecal coliform in the lagoon source water was much less than in the wetland effluents, which display a spread of more than three orders of magnitude. It is also noteworthy that the wetland of greater aspect ratio (System 4) shows a greater reduction.

Excursions around mean performance should be acknowledged in design and regulation. Trends in outlet pathogens can form the basis, or predictions from a first order or other model. Table 12.4 shows example trend fits for several FWS wetlands, together with the multipliers necessary to contain excursions of a given frequency of occurrence. For instance, the 95th percentile median $\log_{10}$ of concentration is 1.31 times higher than the trend. If the outlet trend value is 1,000 CFU/100 mL, then 1 out of 20 times, the outlet may reach $1{,}000^{1.31} = 8{,}511$ CFU/100 mL.

## Input–Output Relation for Fecal Coliforms

FWS treatment wetlands reduce high numbers of incoming fecal coliforms. Wetland data has been acquired for a number of systems, with examples listed in Table 12.6.

Input–output annual data for 256 wetland years is displayed in Figure 12.11. The scatter in these data is quite high

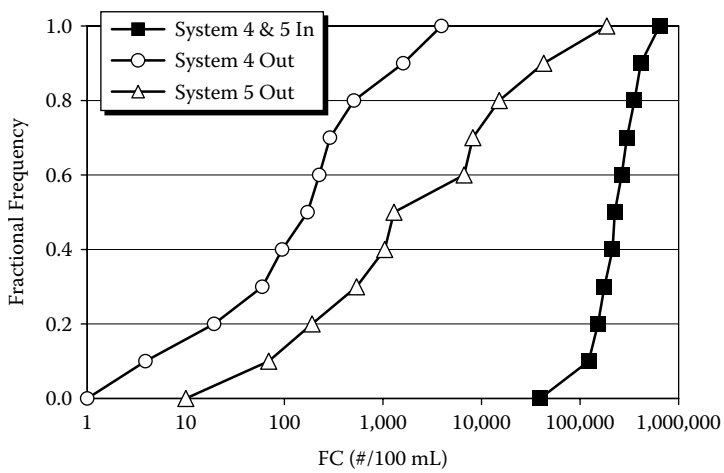



**FIGURE 12.10** Probability densities for monthly fecal coliforms entering and leaving Listowel, Ontario, Systems 4 and 5. System 4 had aspect ratio L:W = 84:1, whereas System 5 had aspect L:W = 5:1. (Data from Herskowitz (1986) *Listowel Artificial Marsh Project Report*. Ontario Ministry of the Environment, Water Resources Branch: Toronto, Ontario.)

**TABLE 12.6**
**Reduction of Fecal Coliforms in FWS Wetlands**

| Site | Cell | Time | Inlet (CFU/100 mL) | Outlet (CFU/100 mL) | Reduction ($log_{10}$) | Reference |
|---|---|---|---|---|---|---|
| Titusville, Florida | All | 2002 | 0 | 48.18 | −2.32 | Unpublished data |
| Orlando Easterly, Florida | 1–12 | Annual 92 | 1 | 24 | −1.53 | NADB v.2 (1998) |
| Sea Pines, North Carolina | T:1 | Annual 92 | 1 | 829 | −2.79 | NADB v.2 (1998) |
| Hillsboro, Oregon | T:1–17 | Annual 89 | 1 | 189 | −2.15 | NADB v.2 (1998) |
| Tarrant, Texas | WC6 | 1993–96 | 23 | 193 | −0.92 | Unpublished data |
| Benton, Kentucky | 1 | April 88–October 89 | 27 | 82 | −0.47 | NADB v.2 (1998) |
| Cannon Beach, Oregon | 1 | Annual 89 | 46 | 31 | 0.18 | NADB v.2 (1998) |
| Estevan, Saskatchewan | All | 2000 | 47 | 16 | 0.47 | Unpublished data |
| Montreal, Quebec | 2 | 1991 | 78 | 91 | −0.07 | Vincent (1992) |
| West Jackson County, Mississippi | 1–2 | Annual 90 | 95 | 578 | −0.79 | NADB v.2 (1998) |
| Brenton Cattle, Iowa | 1 | Annual 92 | 143 | 38 | 0.57 | NADB v.2 (1998) |
| Purdue University, Indiana | A3 | Annual 95 | 175 | 20 | 0.95 | NADB v.2 (1998) |
| Tom Brothers Farm, Indiana | 1 | Annual 95 | 236 | 57 | 0.62 | NADB v.2 (1998) |
| Orange County, Florida | 2 | Annual 92 | 256 | 150 | 0.23 | NADB v.2 (1998) |
| Crittenden Co., Kentucky | 1 | Annual 92 | 346 | 2 | 2.20 | NADB v.2 (1998) |
| Listowel, Ontario | 1 | 1982 | 2,596 | 52 | 1.70 | NADB v.2 (1998) |
| University of SW Louisiana | 2 | Annual 1994–95 | 3,716 | 1,599 | 0.37 | NADB v.2 (1998) |
| Arcata Pilot, California | 4 | 1981–82 | 5,131 | 1255 | 0.61 | Gearheart *et al.* (1989) |
| Arcata, California | All | — | 12,500 | 316 | 1.60 | Gearheart *et al.* (1983) |
| Lakeland, Florida | 1 | Annual 89 | 26,977 | 115 | 2.37 | NADB v.2 (1998) |
| University of Connecticut | 1 | Annual 1994/95 | 28,811 | 1,265 | 1.36 | NADB v.2 (1998) |
| Brookhaven, New York | T:1–5 | Annual 77 | 42,819 | 302 | 2.15 | NADB v.2 (1998) |
| Imperial, California | All | 2003 | 102,366 | 229 | 2.65 | Unpublished data |
| Pembroke, Kentucky | 2 | Annual 89 | 111,601 | 282 | 2.60 | NADB v.2 (1998) |
| Byron Bay, Australia | WB6–7 | 1992–93 | 114,707 | 3470 | — | Unpublished data |
| Cobalt, Ontario | All | 1983–84 | 159,301 | 1,087 | 2.17 | Miller (1989) |
| Brawley, California | All | 2001 | 164,697 | 178 | 2.97 | Unpublished data |
| Crow Edge, United Kingdom | 3 | 1998 | 170,943 | 5,310 | 1.51 | Perkins and Hunter (2000) |
| Sand Mountain, Alabama | 1 | Annual 1990–91 | 175,164 | 3,005 | 1.77 | NADB v.2 (1998) |
| Sand Mountain, Alabama | 1 | LTM 1990–92 | 175,164 | 3,115 | 1.75 | NADB v.2 (1998) |
| Listowel, Ontario | 4 | 1982 | 201,417 | 140 | 3.16 | NADB v.2 (1998) |
| Columbia, Missouri | All | 1994–2000 | 237,077 | 6414 | — | Unpublished data |
| Essex County, Ontario | 1 | Annual 94 | 1,030,000 | 11,999 | 1.93 | NADB v.2 (1998) |
| Oregon State University, Oregon | 1 | Annual 94 | 1,198,419 | 138,916 | 0.94 | NADB v.2 (1998) |
| Neshaminy, Pennsylvania | All | — | 1,290,600 | 5,600 | 2.36 | Unpublished data |
| Harriman, New York | 1 | Annual 81 | 1,953,329 | 14,180 | 2.14 | NADB v.2 (1998) |
| Waldo, Florida | All | — | 7,700,000 | 270,000 | 1.46 | Schuerman *et al.* (1989) |

and is reflective of two points discussed previously. First, there is a nonzero background for this indicator organism group, and that background is itself quite variable. Disinfected influents tend to emerge from an FWS wetland in the range 10–1,000 CFU/100 mL, with a median of 60 CFU/100 mL for those wetlands that receive a disinfected inflow. Second, when inlet concentrations are above about 1,000 CFU/100 mL, there is a $log_{10}$ reduction of about two, but results are widely variable from wetland to wetland. The addition of the hydraulic loading rate to the variables, in the form of a loading plot, does little or nothing to reduce the scatter.

Ponds are also effective in lowering fecal coliform, over a very wide range of inlet concentrations (Kadlec, 2005e). Over a range of nine orders of magnitude of inlet fecal coliform, an average reduction of just over one $log_{10}$ is achieved in ponds of varying size, loading, and configuration. Wetlands have typically been exposed to lesser inlet fecal coliform concentrations than ponds, also spanning seven orders of magnitude, but including the very low inlet values associated with disinfected source waters. For an inlet range of $10^4$–$10^6$ CFU per 100 mL, wetlands appear to be slightly superior in removal to ponds. But at lower inlet counts, wetland performance decreases to a condition of regrowth and reintroduction.

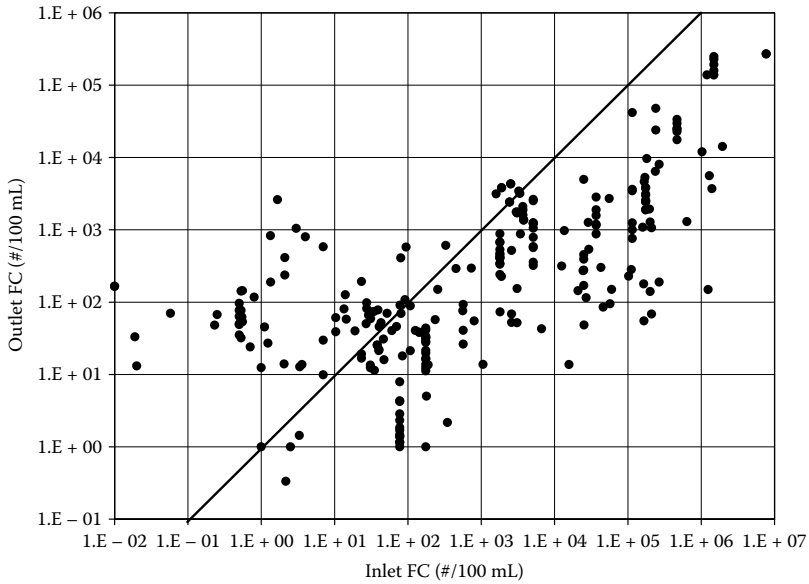



**FIGURE 12.11** Outlet fecal coliforms versus inlet fecal coliforms for FWS wetlands. There are 256 annual data points for 79 different wetlands.

## 12.4 REMOVAL OF OTHER INDICATOR BACTERIA IN FWS WETLANDS

### Total Coliforms

The coliform group has been used as the standard for assessing fecal contamination of waters throughout the twentieth century (Gerba, 2000). Although total coliforms have been widely used in older studies and monitoring, many of the organisms in this broad group are not limited to fecal sources. Accordingly, the subgroup of fecal coliforms has become the preferred indicator, which includes the genera *Escherichia* and *Klebsiella*. A simple lab test allows differentiation.

Here, it is noted that FWS treatment wetlands reduce high numbers of incoming total coliforms. Wetland data have been acquired for a number of systems, with examples listed in Table 12.7. As for fecal coliforms, wetlands reintroduce total coliforms to a disinfected influent, to a limited degree, as evidenced by the Orlando Easterly Wetland (OEW) data. For nondisinfected influents, there are reductions of one to two $\log_{10}$.

**TABLE 12.7**
**Reduction of Total Coliforms in FWS Wetlands**

| System | Years | $C_i$ (CFU/100 mL) | $C_o$ (CFU/100 mL) | Reduction ($\log_{10}$) | Reference |
|---|---|---|---|---|---|
| OEW, Florida | 5 | 2 | 225 | –2.18 | Unpublished data |
| Byron Bay, Australia | 3 | 29,225 | 3,520 | 0.92 | Unpublished data |
| Estevan, Saskatchewan | 10 | 37,076 | 3,072 | 1.08 | Unpublished data |
| Arcata, California | 1 | 87,525 | 47,139 | 0.27 | Gearheart *et al.* (1983) |
| Arcata, California | 1 | 87,525 | 37,872 | 0.36 | Gearheart *et al.* (1983) |
| Arcata, California | 1 | 87,525 | 28,242 | 0.49 | Gearheart *et al.* (1983) |
| Richmond, Australia | 2 | 3,800,000 | 69,200 | 1.74 | Bavor *et al.* (1988) |
| Richmond, Australia | 2 | 3,800,000 | 105,000 | 1.56 | Bavor *et al.* (1988) |
| Waldo, Florida | 1 | 9,200,000 | 600,000 | 1.19 | Scheuerman *et al.* (1989) |
| Tucson, Arizona | — | 10,000,000 | 83,000 | 2.08 | Karpiscak *et al.* (1996) |
| Waldo, Florida | 1 | 22,000,000 | 1,400,000 | 1.20 | Unpublished data |

### Fecal Streptococcus

Fecal streptococcus belongs to the genera *Enterococcus* and *Streptococcus*. These are considered to have advantages over fecal coliforms as an indicator because (1) they rarely multiply in water, (2) are more resistant to environmental stress, and (3) generally persist longer in the environment (Gerba, 2000). A high ratio of fecal streptococci to fecal coliforms is taken as strong evidence that the source is of human origin.

Inflow and outflow densities of fecal streptococcus bacteria have been monitored at a number of pilot and full-scale wetland treatment systems. Table 12.8 summarizes these data for some of the wetland treatment systems receiving municipal wastewater, pretreated to differing extents. Data for disinfected influents again show reintroduction in treatment wetlands, with effluent numbers in the range 50–200 CFU per 100 mL. For nondisinfected influents, there are reductions of one to two $\log_{10}$.

### *Escherichia coli*

*Escherichia coli* is found in the intestinal tract of all warm-blooded animals and is usually considered harmless (Gerba, 2000). However, several strains are capable of causing gastroenteritis and include organisms that may cause hemorrhagic colitis, a dangerous disease, especially for the elderly. For this reason, *E. coli* has found favor in some instances as a replacement for fecal coliforms as an indicator organism.

*E. coli* has the ability to regrow or to be reintroduced in FWS wetlands. Considerable numbers were found in the wetland effluents at Tres Rios, Arizona, for a disinfected influent (Table 12.9). Conversely, large reductions may occur when the wetland influent has large numbers. Such reductions may be quite large for highly compartmentalized wetlands of high aspect ratio, for instance, the Brawley, California, wetlands.

### Miscellaneous Bacteria

*Clostridium perfringens* is exclusively of fecal origin. It is regarded as particularly resistant to pH and temperature stress and was therefore chosen for monitoring at the Listowel, Ontario, system (Herskowitz, 1986). *Salmonella* is a large group, all of which are pathogenic to humans. *Salmonella* causes typhoid and paratyphoid fevers, and is found in a large variety of both warm-blooded and cold-blooded animals. *Yersinia enterocolitica*, a bacterium that causes gastroenteritis, was found in large numbers in the Listowel lagoons. Because it survives well at cold temperatures, it was monitored in the Listowel project. *Pseudomonas aeruginosa* is associated with the disease *otitis externa*, or "swimmers ear" (Herskowitz, 1986). All these organisms, plus others, have been found to be reduced in FWS treatment wetlands (Table 12.10).

## 12.5 PARASITE AND VIRUS REMOVAL IN FWS WETLANDS

### Parasites

Parasites include protozoa and helminths. In the United States, the two protozoa of special interest are *Giardia lamblia* and *Cryptosporidium parvum*. The former is the most frequently identified intestinal parasite in the United States (Maier *et al.*, 2000). The latter was implicated in an outbreak in 1993 in Milwaukee, Wisconsin, in which over 400,000 people were infected. *Giardia* is of special interest in connection with wetlands because beavers and muskrats are believed to be reservoirs of infection. Worldwide, *Ascaris lumbricoides* is the most prevalent parasitic infection, with an estimated 22% of the world population affected (Maier *et al.*, 2000). Adult *Ascaris* are roundworms which inhabit the intestine, but they are ingested as eggs. Two other prevalent infectious worms are *Trichuris trichiuria* and *Taenia saginata*, and these are of special concern in warm climates.

FWS wetlands reduce the numbers of these parasitic organisms by reducing the number of eggs or cysts that survive in the aquatic environment (Table 12.11). One mechanism is presumably sedimentation of eggs, followed by eventual inactivation. Nelson (2003) reports egg numbers in pond sludges ranging from less than ten per dry gram in France, to thousands of *Ascaris* eggs per dry gram in Brazil. Inactivation may take more than a year. However, Stott *et al.* (2001) determined that oocysts of *Cryptosporidium parvum* were effectively ingested by protozoans, notably *Paramecium caudatum* and *Stylonychia mytilus*. Therefore, predation may also be a factor in the observed reductions.

### Viruses

The average viral content of domestic sewage in the United States is about 7,000 particles per liter (Gersberg and Gearheart, 1989). Because the potential infective dose of viruses can be so low (<10 particles), and viruses are generally hardier in natural environments than bacterial pathogens, there has been considerable concern about their fate in wetland and conventional treatment systems. However, because viruses are more costly and difficult to monitor analytically, they have not received as much study as bacterial indicator populations. Existing research indicates that wetlands are generally hostile environments for viruses (Table 12.12).

## TABLE 12.8
## Reduction of Fecal *Streptococcus* in FWS Wetlands

| Site Name | Cell | Time Period | Inlet (CFU/100 mL) | Outlet (CFU/100 mL) | Reduction ($log_{10}$) | Reference |
|---|---|---|---|---|---|---|
| OEW, Florida | To MM7[a] | 1993 | 1 | 59 | −1.91 | Unpublished data |
| OEW, Florida | To MM7 | 1994 | 1 | 85 | −2.20 | Unpublished data |
| OEW, Florida | To MM7 | 1995 | 1 | 61 | −2.08 | Unpublished data |
| OEW, Florida | To MM7 | 1996 | 1 | 193 | −2.59 | Unpublished data |
| OEW, Florida | To MM7 | 1997 | 1 | 203 | −2.60 | Unpublished data |
| OEW, Florida | To HS10[b] | 1993 | 1 | 111 | −2.18 | Unpublished data |
| OEW, Florida | To HS10 | 1994 | 1 | 118 | −2.34 | Unpublished data |
| OEW, Florida | To HS10 | 1995 | 1 | 2 | −0.60 | Unpublished data |
| OEW, Florida | To HS10 | 1996 | 1 | 209 | −2.62 | Unpublished data |
| OEW, Florida | To HS10 | 1997 | 1 | 84 | −2.21 | Unpublished data |
| Titusville, Florida | All | 1997 | 1 | 124 | −2.09 | Unpublished data |
| Titusville, Florida | All | 1998 | 1 | 129 | −2.11 | Unpublished data |
| Titusville, Florida | All | 1999 | 1 | 175 | −2.24 | Unpublished data |
| Titusville, Florida | All | 2000 | 1 | 108 | −2.03 | Unpublished data |
| Titusville, Florida | All | 2001 | 1 | 107 | −2.03 | Unpublished data |
| Titusville, Florida | All | 2002 | 1 | 84 | −1.92 | Unpublished data |
| Boggy Gut, South Carolina | All | — | 2 | 1,972 | −2.99 | NADB database (1998) |
| Arcata, California | 1 | 1980–1981 | 545 | 304 | 0.25 | Gearheart *et al.* (1983) |
| Arcata, California | 2 | 1980–1981 | 545 | 418 | 0.11 | Gearheart *et al.* (1983) |
| Arcata, California | 3 | 1980–1981 | 545 | 371 | 0.17 | Gearheart *et al.* (1983) |
| Arcata, California | 4 | 1980–1981 | 545 | 463 | 0.07 | Gearheart *et al.* (1983) |
| Arcata, California | 5 | 1980–1981 | 545 | 195 | 0.45 | Gearheart *et al.* (1983) |
| Arcata, California | 6 | 1980–1981 | 545 | 211 | 0.41 | Gearheart *et al.* (1983) |
| Arcata, California | 7 | 1980–1981 | 545 | 265 | 0.31 | Gearheart *et al.* (1983) |
| Arcata, California | 8 | 1980–1981 | 545 | 320 | 0.23 | Gearheart *et al.* (1983) |
| Arcata, California | 9 | 1980–1981 | 545 | 194 | 0.45 | Gearheart *et al.* (1983) |
| Arcata, California | 10 | 1980–1981 | 545 | 172 | 0.50 | Gearheart *et al.* (1983) |
| Arcata, California | 11 | 1980–1981 | 545 | 160 | 0.53 | Gearheart *et al.* (1983) |
| Arcata, California | 12 | 1980–1981 | 545 | 176 | 0.49 | Gearheart *et al.* (1983) |
| Cherry Burton, United Kingdom | All | — | 650 | 136 | 0.68 | Cooper *et al.* (1996) |
| Crow Edge, United Kingdom | 1 | 1998 | 5,050 | 219 | 1.36 | Perkins and Hunter (2000) |
| Crow Edge, United Kingdom | 2 | 1998 | 6,575 | 644 | 1.01 | Perkins and Hunter (2000) |
| Crow Edge, United Kingdom | 3 | 1998 | 6,075 | 239 | 1.41 | Perkins and Hunter (2000) |
| Crow Edge, United Kingdom | 4 | 1998 | 6,375 | 163 | 1.59 | Perkins and Hunter (2000) |
| Brookhaven, New York | All | — | 6,780 | 739 | 0.96 | NADB database (1998) |
| Brookhaven, New York | All | — | 6,780 | 2,300 | 0.47 | NADB database (1998) |
| Rathermon, Ireland | All | — | 23,700 | 515 | 1.66 | Costello (1989) |
| Byron Bay, Australia | Research | 1990–1993 | 25,570 | 1,915 | 1.13 | Unpublished data |
| Byron Bay, Australia | To WB2–3 | 1990–1993 | 25,570 | 4,839 | 0.72 | Unpublished data |
| Byron Bay, Australia | To WB3–4 | 1990–1993 | 25,570 | 2,378 | 1.03 | Unpublished data |
| Byron Bay, Australia | To WB4–5 | 1990–1993 | 25,570 | 1,358 | 1.27 | Unpublished data |
| Byron Bay, Australia | To WB5–6 | 1990–1993 | 25,570 | 654 | 1.59 | Unpublished data |
| Byron Bay, Australia | To WB6–7 | 1990–1993 | 25,570 | 1,522 | 1.23 | Unpublished data |
| Cobalt, Ontario | All | 1983–1984 | 34,654 | 3,456 | 1.00 | Miller (1989) |
| Mpophomeni, South Africa | All | — | 37,000 | 1,300 | 1.45 | Wood and Hensman (1989) |
| Waldo, Florida | All | — | 60,000 | 400 | 2.18 | Scheuerman *et al.* (1989) |
| Mpophomeni, South Africa | All | — | 160,000 | 3,400 | 1.67 | Wood and Hensman (1989) |
| Columbia, Missouri | All | 1994–2000 | 367,200 | 5,468 | 1.83 | Unpublished data |

[a]There are 15 cells prior to MM7

[b]There are 17 cells prior to HS10

**TABLE 12.9**
**Reduction of *E. coli* in FWS Treatment Wetlands**

| Site Name | System | Year | Inlet (CFU/100 mL) | Outlet (CFU/100 mL) | Reduction ($log_{10}$) | Reference |
|---|---|---|---|---|---|---|
| Tres Rios, Arizona | H1 | 1995 | 0 | 878 | −3.26 | Unpublished data |
| Tres Rios, Arizona | H2 | 1995 | 0 | 585 | −3.08 | Unpublished data |
| Tres Rios, Arizona | C1 | 1997 | 1 | 1,715 | −3.36 | Unpublished data |
| Tres Rios, Arizona | C2 | 1997 | 1 | 4,440 | −3.77 | Unpublished data |
| Tres Rios, Arizona | H1 | 1996 | 1 | 2,421 | −3.23 | Unpublished data |
| Tres Rios, Arizona | H2 | 1996 | 1 | 9,387 | −3.82 | Unpublished data |
| Tres Rios, Arizona | C1 | 1996 | 3 | 3,746 | −3.13 | Unpublished data |
| Tres Rios, Arizona | C2 | 1996 | 3 | 1,815 | −2.81 | Unpublished data |
| Tres Rios, Arizona | H1 | 1997 | 4 | 468 | −2.10 | Unpublished data |
| Tres Rios, Arizona | H2 | 1997 | 4 | 4,172 | −3.05 | Unpublished data |
| Tres Rios, Arizona | C1 | 1995 | 5 | 16 | −0.54 | Unpublished data |
| Tres Rios, Arizona | C2 | 1995 | 5 | 29 | −0.81 | Unpublished data |
| Brighton, Ontario | All | 2004 | 84 | 50 | 0.22 | Unpublished data |
| Brighton, Ontario | All | 2002 | 150 | 101 | 0.17 | Unpublished data |
| Brighton, Ontario | All | 2003 | 167 | 340 | −0.31 | Unpublished data |
| Brighton, Ontario | All | 2001 | 1,379 | 672 | 0.31 | Unpublished data |
| Imperial, California | All | 2001 | 391 | 66 | 0.77 | Unpublished data |
| Imperial, California | All | 2004 | 910 | 95 | 0.98 | Unpublished data |
| Imperial, California | All | 2002 | 1,252 | 34 | 1.56 | Unpublished data |
| Imperial, California | All | 2005 | 1,264 | 55 | 1.36 | Unpublished data |
| Imperial, California | All | 2003 | 2,108 | 140 | 1.18 | Unpublished data |
| Oxelösund, Sweden | All | 1996 | 1,900 | 10 | 2.28 | Stenström and Carlander (2001) |
| Brawley, California | All | 2002 | 13,031 | 26 | 2.70 | Unpublished data |
| Brawley, California | All | 2001 | 14,564 | 73 | 2.30 | Unpublished data |
| Brawley, California | All | 2004 | 104,418 | 47 | 3.35 | Unpublished data |
| Brawley, California | All | 2003 | 248,172 | 42 | 3.77 | Unpublished data |
| Duplin, North Carolina | All | — | 200,000 | 2,000 | 2.00 | Hill and Sobsey (2001) |
| Slingsby, United Kingdom | All | — | 16,644 | 2,343 | 0.85 | Cooper *et al.* (1996) |
| Cherry Burton, United Kingdom | All | — | 23,536 | 14,480 | 0.21 | Cooper *et al.* (1996) |
| Bishop Wilton, United Kingdom | All | — | 30,699 | 8,505 | 0.56 | Cooper *et al.* (1996) |
| Carperby, United Kingdom | All | — | 36,630 | 14,816 | 0.39 | Cooper *et al.* (1996) |
| Wetwang, United Kingdom | All | — | 68,743 | 12,521 | 0.74 | Cooper *et al.* (1996) |

Scheuerman *et al.* (1989) found that bacteriophages and enteroviruses in a natural cypress wetland declined with distance from an outfall, indicating that wetland virus removals can be effective if residence time is adequate. In this test, bacteriophages were removed at a slower rate than bacterial indicators and enteroviruses. Enteroviruses declined at the fastest rate and were not observed to accumulate in downstream sediments. High removals of enteroviruses were also found at the Houghton Lake, Michigan, natural treatment wetland (unpublished data) and the Gainesville, Florida, natural treatment wetland (Wellings, 1977). Most recent research has focused on the reduction of bacteriophages in wetlands (Table 12.12), with reductions of greater than 90% at a number of geographically distinct sites.

### Wildlife Pathogens

Wildlife pathogens are also potentially associated with treatment wetlands as they are with natural wetlands. Avian

**TABLE 12.10**
**Reduction of Miscellaneous Bacteria in FWS Wetlands**

| Site Name | System | Organism | Inlet (CFU/100 mL) | Outlet (CFU/100 mL) | Reduction ($log_{10}$) | Reference |
|---|---|---|---|---|---|---|
| Listowel, Ontario | 1 | *Clostridium perfringens* | 1,036 | 10 | 2.02 | Herskowitz (1986) |
| Listowel, Ontario | 2 | *Clostridium perfringens* | 1,036 | 9 | 2.06 | Herskowitz (1986) |
| Listowel, Ontario | 3 | *Clostridium perfringens* | 1,036 | 6 | 2.24 | Herskowitz (1986) |
| Listowel, Ontario | 4 | *Clostridium perfringens* | 13,148 | 4 | 3.52 | Herskowitz (1986) |
| Listowel, Ontario | 5 | *Clostridium perfringens* | 13,148 | 19 | 2.84 | Herskowitz (1986) |
| Duplin, North Carolina | — | *Clostridium perfringens* | 12,000 | 500 | 1.38 | Hill and Sobsey (2001) |
| Oxelösund, Sweden | — | *Clostridium perfringens* | 1,600 | 21 | 1.88 | Stenström and Carlander (2001) |
| Glendale, Arizona | — | *Clostridium perfringens* | 16,300 | 13,100 | 0.09 | Karpiscak *et al.* (2001) |
| Listowel, Ontario | 1 | *Pseudomonas aruginosa* | 10 | 1 | 1.00 | Herskowitz (1986) |
| Listowel, Ontario | 2 | *Pseudomonas aruginosa* | 10 | 3 | 0.52 | Herskowitz (1986) |
| Listowel, Ontario | 3 | *Pseudomonas aruginosa* | 10 | 1 | 1.00 | Herskowitz (1986) |
| Listowel, Ontario | 4 | *Pseudomonas aruginosa* | 662 | 2 | 2.52 | Herskowitz (1986) |
| Listowel, Ontario | 5 | *Pseudomonas aruginosa* | 662 | 9 | 1.87 | Herskowitz (1986) |
| Listowel, Ontario | 1 | *Salmonella* spp. | 10 | 0 | BDL[a] | Herskowitz (1986) |
| Listowel, Ontario | 2 | *Salmonella* spp. | 10 | 1 | 1.00 | Herskowitz (1986) |
| Listowel, Ontario | 3 | *Salmonella* spp. | 10 | 0 | BDL[a] | Herskowitz (1986) |
| Listowel, Ontario | 4 | *Salmonella* spp. | 228 | 0 | BDL[a] | Herskowitz (1986) |
| Listowel, Ontario | 5 | *Salmonella* spp. | 228 | 2 | 2.06 | Herskowitz (1986) |
| Arcata, California | — | *Salmonella* spp. | 100 | 5 | 1.30 | Gearheart *et al.* (1983) |
| Tucson, Arizona | — | *Salmonella* spp. | — | — | 4.16 | Karim *et al.* (2004) |
| Duplin, North Carolina | — | *Salmonella* spp. | 400 | 10 | 1.60 | Hill and Sobsey (2001) |
| Listowel, Ontario | 1 | *Yersinia entercolitica* | 4,241 | 506 | 0.92 | Herskowitz (1986) |
| Listowel, Ontario | 2 | *Yersinia entercolitica* | 4,241 | 247 | 1.23 | Herskowitz (1986) |
| Listowel, Ontario | 3 | *Yersinia entercolitica* | 4,241 | 330 | 1.11 | Herskowitz (1986) |
| Listowel, Ontario | 4 | *Yersinia entercolitica* | 28,602 | 563 | 1.71 | Herskowitz (1986) |
| Listowel, Ontario | 5 | *Yersinia entercolitica* | 28,602 | 2,224 | 1.11 | Herskowitz (1986) |
| Duplin, North Carolina | — | Enterococci | 150,000 | 20,000 | 0.88 | Hill and Sobsey (2001) |
| Tucson, Arizona | — | Enterococci | 69,400 | 18,200 | 0.58 | Karpiscak *et al.* (2001) |
| Oxelösund, Sweden | — | Fecal enterococci | 480 | 0.5 | 2.98 | Stenström and Carlander (2001) |
| Tucson, Arizona | — | *Listeria* spp. | 29,300 | 20,000 | 0.17 | Karpiscak *et al.* (2001) |

[a] BDL—Below detection limit.

cholera (caused by *Pasteurella multocida*) and avian botulism (caused by *Clostridium botulinum*) have been found in association with sewage lagoons and natural and treatment wetlands (Friend, 1985). Conditions that favor transmission are exposed mudflats together with stressed and overcrowded bird populations. The nematode *Eustronglydes ignotus* has been identified as a potential threat to wading birds that feed and nest near nutrient-rich sites. However, extensive research on the fish that are intermediate hosts have shown no potential vector in Florida wetlands treating agricultural runoff (SFWMD, 1997).

## 12.6 FECAL COLIFORM REMOVAL IN SSF WETLANDS

The most common indicator group is fecal coliforms; consequently, the performance database for SSF wetlands is largest for this indicator group. Global removals (input–output data), for example, in HSSF and VF wetlands, are shown in Table 12.13.

For a larger database of 130 wetlands or operating conditions, the median global removal was a 1.82 $log_{10}$ reduction in fecal coliforms. Generally, fecal coliform removal in HSSF

**TABLE 12.11**
**Reduction of Parasites in FWS Wetlands**

| System | Vegetation | Organism | HRT (h) | Assay | Inlet | Outlet | Removal Efficiency | Reference |
|---|---|---|---|---|---|---|---|---|
| Tucson, Arizona | Duckweed | *Giardia* | — | Cysts/L | 15.7 | 0.6 | 96% | Karpiscak *et al.* (1996) |
| Tucson, Arizona | Duckweed | *Cryptosporidium* | — | Oocysts/L | 2.3 | 0.2 | 90% | Karpiscak *et al.* (1996) |
| Tucson, Arizona | Mixed | *Giardia* | — | Cysts/L | 55 | 16 | 71% | Karpiscak *et al.* (1996) |
| Tucson, Arizona | Mixed | *Cryptosporidium* | — | Oocysts/L | 13 | 5 | 61% | Karpiscak *et al.* (1996) |
| Marrakech, Morocco | *Phragmites* | Nematodes | 1.20 | Eggs/L | 3.83 | 0.48 | 87% | Mandi *et al.* (1998) |
| Marrakech | *Phragmites* | *Ascaris, Trichuris* | 1.78 | Eggs/L | 3.83 | 0.51 | 87% | Mandi *et al.* (1998) |
| Marrakech | *Phragmites* | *Ascaris, Trichuris* | 2.38 | Eggs/L | 3.83 | 0.39 | 90% | Mandi *et al.* (1998) |
| Marrakech | *Phragmites* | *Ascaris, Trichuris* | 3.49 | Eggs/L | 3.83 | 0.02 | 99% | Mandi *et al.* (1998) |
| Marrakech | *Phragmites* | Cestodes | 1.20 | Eggs/L | 0.17 | 0.14 | 18% | Mandi *et al.* (1998) |
| Marrakech | *Phragmites* | *Hymenolepsi* | 1.78 | Eggs/L | 0.17 | 0.00 | 100% | Mandi *et al.* (1998) |
| Marrakech | *Phragmites* | *Taenia* | 2.38 | Eggs/L | 0.17 | 0.00 | 100% | Mandi *et al.* (1998) |
| Marrakech | *Phragmites* | *Taenia* | 3.49 | Eggs/L | 0.17 | 0.00 | 100% | Mandi *et al.* (1998) |
| Brazil | Pond series | *Ascaris, Trichuris* | — | Eggs/L | 993 | 0.00 | 100% | Stott *et al.* (2003) |

wetlands is enhanced under the following conditions:

- Longer nominal hydraulic residence time (τ) or lower hydraulic loading
- Finer bed materials (sand), but only to the extent that the fine bed media does not impair hydraulic performance
- Warmer water temperatures
- Shallower bed depths

### Effect of Vegetation

In general, the presence of plants in HSSF wetlands has a beneficial effect on pathogen reduction (Table 12.14). It is not clear whether this is due to the effects of the plants on system hydraulics or to other causes, such as the presence of antibiotic exudates in the rhizosphere (Vacca *et al.*, 2005). Further, a plant advantage is by no means a universal conclusion.

**TABLE 12.12**
**Virus Removal in FWS Wetlands**

| Site | Vegetation | Virus | Inlet (PFU/100 mL) | Outlet (PFU/100 mL) | Removal Efficiency | Reference |
|---|---|---|---|---|---|---|
| Gainesville, Florida | Cypress | polio and coxsackie | 0.021 | 0.003 | 85.7% | Wellings (1977) |
| Houghton Lake, Michigan | *Typha, Carex* | reovirus + polio type 1 | 313 | 3 | 99.0% | Unpublished data |
| Tucson, Arizona | *Lemna* | coliphage | 1,200 | 720 | 40.0% | Karpiscak *et al.* (1996) |
| Duplin, North Carolina | *Typha* | F-specific coliphage | 2,000 | 70 | 96.5% | Hill and Sobsey (2001) |
| Oxelösund, Sweden | *Phragmites* | coliphage | 6,700 | 127 | 98.1% | Stenström and Carlander (2001) |
| Tucson, Arizona | Mixed | coliphage | 16,700 | 851 | 94.9% | Gerba *et al.* (1999) |
| Waldo, Florida | Cypress 2 | bacteriophage | 30,000 | 11,000 | 63.3% | Scheuerman *et al.* (1989) |
| Waldo, Florida | Cypress 1 | bacteriophage | 42,000 | 6,100 | 85.5% | Scheuerman *et al.* (1989) |
| Duplin, North Carolina | *Typha* | Somatic coliphage | 80,000 | 800 | 99.0% | Hill and Sobsey (2001) |
| Hemet, California | *Scirpus* + Deep Zone | MS2, pulse load PFU | 1.2E + 12 | 4.8E + 09 | 99.6% | Chendorain *et al.* (1998) |
| Hemet, California | *Scirpus* | MS2, pulse load PFU | 1.8E + 14 | 3.8E + 12 | 97.9% | Chendorain *et al.* (1998) |
| Hemet, California | *Scirpus* | MS2, pulse load PFU | 1.8E + 14 | 1.2E + 13 | 93.1% | Chendorain *et al.* (1998) |
| Hemet, California | *Scirpus* + Deep Zone | MS2, pulse load PFU | 1.8E + 14 | 2.7E + 12 | 98.5% | Chendorain *et al.* (1998) |
| Arcata, California | *Typha, Scirpus* | FRNA phage MS2 | — | — | 91.5% | Gearheart *et al.* (1983) |

## TABLE 12.13
## Reduction of Fecal Coliforms for SSF Wetlands

| Type | System Name and Location | Reference | Media | Vegetation | HRT (d) | HLR (cm/d) | Inlet $\log_{10}$ (CFU/100 mL) | Outlet $\log_{10}$ (CFU/100 mL) | Reduction ($\log_{10}$) |
|---|---|---|---|---|---|---|---|---|---|
| HSSF | Anderson County, Kentucky | Thom *et al.* (1998) | Gravel | Mixed | 12 | 9.2 | 8.114 | 5.519 | 2.60 |
| HSSF | Craven County, North Carolina 1A | Myers (1993) | Gravel | *Typha* | 1.1 | 4.36 | 4.373 | 3.722 | 0.65 |
| HSSF | Dar Es Salaam, Tanzania | Kaseva (2004) | Gravel | *Phragmites* | 1.96 | 12.2 | 7.111 | 6.613 | 0.50 |
| HSSF | Florence, Italy | Masi *et al.* (2004) | Gravel | *Phragmites* | 3.00 | 17 | 6.710 | 3.772 | 2.94 |
| HSSF | Hamilton, New Zealand | Tanner *et al.* (1995a) | Gravel | *Schoenoplectus* | 2 | 7.25 | 5.276 | 1.653 | 3.62 |
| HSSF | Hamilton, New Zealand | Tanner *et al.* (1995a) | Gravel | *Schoenoplectus* | 3 | 4.25 | 5.276 | 1.591 | 3.69 |
| HSSF | Hamilton, New Zealand | Tanner *et al.* (1995a) | Gravel | *Schoenoplectus* | 5.5 | 2.64 | 5.276 | 1.176 | 4.10 |
| HSSF | Hamilton, New Zealand | Tanner *et al.* (1995a) | Gravel | *Schoenoplectus* | 7 | 2.09 | 5.276 | 0.944 | 4.33 |
| HSSF | Morgantown, West Virginia | Hench *et al.* (2003) | Gravel | Mixed | 7 | 1.5 | 8.000 | 5.200 | 2.80 |
| HSSF | Ohio, System 13 Cell 1 | Steer *et al.* (2005) | Gravel | Mixed | 5.0 | 3.68 | 4.270 | 3.590 | 0.68 |
| HSSF | Pachuca, Mexico | Rivera *et al.* (1997) | Gravel | *Phragmites* | 1.7 | 14.1 | 10.668 | 9.276 | 1.39 |
| HSSF | Pathumthani, Thailand | Sawaittayothin and Polprasert (2007) | Gravel | *Typha angustifolia* | 1 | 20.0 | 2.831 | 2.393 | 0.44 |
| HSSF | Rabat, Morocco | El Hafiane and El Hamouri (2004) | Gravel | *Arundo donax* | 0.8 | 13 | 3.643 | 3.000 | 0.64 |
| HSSF | Sahuarita, Arizona | Nokes *et al.* (2003) | Gravel | *Scirpus* spp. | 3.4 | 5.9 | 5.079 | 3.519 | 1.56 |
| HSSF | Morgantown, West Virginia | Hench *et al.* (2003) | Gravel | *Typha, Juncus,* and *Scirpus* | 7.0 | 1.5 | 8.000 | 5.200 | 2.80 |
| HSSF | Rogers Farm, Nebraska | Jillson *et al.* (2001) | Gravel | *Typha* and *Scirpus* | 7.5 | 2.4 | 6.000 | 4.000 | 2.00 |
| HSSF | Pima County, Arizona | Quiñónez-Díaz *et al.* (2001) | Gravel | *Scirpus* spp. | 12.5 | 4.8 | 7.875 | 3.431 | 4.44 |
| HSSF | Richmond, New South Wales | Bavor *et al.* (1988) | Gravel | None | 4.7 | 3.9 | 6.100 | 3.500 | 2.60 |
| HSSF | Richmond, New South Wales | Bavor *et al.* (1988) | Gravel | *Schoenoplectus* | 6.6 | 5.1 | 6.100 | 4.100 | 2.00 |
| HSSF | Richmond, New South Wales | Bavor *et al.* (1988) | Gravel | *Typha* | 5.9 | 4.6 | 6.100 | 3.900 | 2.20 |
| HSSF | Richmond, New South Wales | Bavor *et al.* (1988) | Gravel | Mix A | 7.8 | 4.6 | 6.100 | 2.700 | 3.40 |
| HSSF | Richmond, New South Wales | Bavor *et al.* (1988) | Gravel | Mix B | 9.2 | 3.9 | 6.100 | 2.400 | 3.70 |
| HSSF | Beery, Kansas | Mankin and Powell (2000) | Gravel | *Phragmites* | 13.0 | 1.5 | 5.602 | 4.000 | 1.60 |
| HSSF | Scott, Kentucky | Thom *et al.* (1998) | Gravel | *Schoenoplectus* | 64.7 | 0.2 | 6.380 | 3.959 | 2.42 |
| HSSF | NERCC 1A, Minnesota | Raw data | Gravel | *Typha* and *Scirpus* | 1.9 | 8.4 | 5.540 | 4.265 | 1.28 |
| HSSF | Grand Lake, Minnesota | Raw data | Gravel | *Typha* | 4.6 | 5.2 | 5.347 | 3.416 | 1.93 |
| **Mean** | | | | | | | **6.04** | **3.72** | **2.32** |
| **Median** | | | | | | | **6.05** | **3.66** | **2.31** |
| VF | Florence, Italy | Masi *et al.* (2004) | Sand | *Phragmites* | — | 17.00 | 3.77 | 3.26 | 0.52 |
| VF | Lauwersoog, The Netherlands | Meuleman *et al.* (2003) | Sand | *Phragmites* | — | 0.75 | 4.70 | 1.27 | 3.43 |
| VF | Aarhus, Denmark | Arias *et al.* (2003b) | Gravel | *Phragmites* | — | 52.00 | 7.59 | 5.78 | 1.81 |
| VF | Aarhus, Denmark | Arias *et al.* (2003b) | Gravel | *Phragmites* | — | 26.00 | 5.78 | 3.78 | 2.00 |
| VF | Hobbitstee, Netherlands | ECOFYT – raw data | — | — | — | 2.92 | 5.76 | 1.53 | 4.19 |
| VF | Spijkerman, Netherlands | ECOFYT – raw data | — | — | — | 2.19 | 6.03 | 3.70 | 3.32 |
| VF | Aurignac, France | Torrens *et al.* (2007) | — | — | — | 20–80 | 4.2 | 2.65 | 1.55 |
| **Mean** | | | | | | | **5.40** | **3.14** | **2.40** |
| **Median** | | | | | | | **5.76** | **3.26** | **2.00** |

**TABLE 12.14**
**Effect of the Presence of Vegetation on the Reduction of Fecal Coliforms in Side-by-Side Studies in HSSF Wetlands**

| System Name and Location | Reference | Data Period | Media | $\log_{10}$ Reduction | | Ratio |
|---|---|---|---|---|---|---|
| | | | | No Vegetation | Vegetation | |
| ***Phragmites*** | | | | | | |
| Audlem, United Kingdom | Rivera *et al.* (1995) | Summer 1991 | Soil | 1.10 | 2.52 | 2.291 |
| Audlem, United Kingdom | Rivera *et al.* (1995) | Winter 1991 | Soil | 0.60 | 1.30 | 2.167 |
| Audlem, United Kingdom | Rivera *et al.* (1995) | Summer 1991 | Gravel | 2.70 | 2.40 | 0.889 |
| Audlem, United Kingdom | Rivera *et al.* (1995) | Winter 1991 | Gravel | 3.00 | 0.89 | 0.297 |
| Dar Es Salaam, Tanzania | Kaseva (2004) | 2002 | Gravel | 0.37 | 0.50 | 1.345 |
| **Mean** | | | | | | **1.398** |
| ***Typha*** | | | | | | |
| Dar Es Salaam, Tanzania | Kaseva (2004) | 2002 | Gravel | 0.37 | 0.55 | 1.497 |
| Richmond, New South Wales | Bavor *et al.* (1988) | 1984–86 | Gravel | 2.60 | 2.20 | 0.846 |
| Morgantown, West Virginia | Hench *et al.* (2003) | 2 years | Gravel | 2.80 | 2.30 | 0.821 |
| **Mean** | | | | | | **1.055** |
| ***Schoenoplectus*** | | | | | | |
| Pima County, Arizona | Quiñónez-Díaz *et al.* (2001) | 1996–97 | Gravel | 3.62 | 4.44 | 1.228 |
| Richmond, New South Wales | Bavor *et al.* (1988) | 1984–86 | Gravel | 2.60 | 2.00 | 0.769 |
| Hamilton, New Zealand | Tanner *et al.* (1995a) | 1990–91 | Gravel | 0.48 | 3.62 | 7.594 |
| Hamilton, New Zealand | Tanner *et al.* (1995a) | 1990–91 | Gravel | 0.85 | 3.69 | 4.361 |
| Hamilton, New Zealand | Tanner *et al.* (1995a) | 1990–91 | Gravel | 1.07 | 4.10 | 3.824 |
| Hamilton, New Zealand | Tanner *et al.* (1995a) | 1990–91 | Gravel | 1.44 | 4.33 | 3.013 |
| **Mean** | | | | | | **3.465** |

### Effect of Depth

The year-long studies of Barrett *et al.* (2001) provide a side-by-side comparison of 30- and 60-cm depths of gravel with mixed plantings (Table 12.15). Both beds were supplied with the same flow rate, thus creating equal hydraulic loadings, but doubled detention time in the deep bed. There was a very strong effect, with the deeper gravel bed performing much more poorly. The calculated volumetric rate constants were about five times lower in the deep bed for most organisms, except for *Clostridium perfringens*, which was only 40% lower in the deep bed. This difference was so large that calculated areal rate constants were also lower in the deep bed, by about a factor of three. It is not known if these differences arise from system hydraulics or from other causes.

### Effect of Media Size and Uniformity

The data of García *et al.* (2003b) help to understand the potential importance of media size and uniformity. Four equal-sized side-by-side gravel beds were operated with equal but varied nominal detention times. Each bed had a different media (Table 12.16), with three particle sizes and three uniformities. As detention time was increased, the removal increased. However, there was no log-linear decrease, indicating that hydraulics were far from plug flow. When the data were reanalyzed with the *P-k-C** model (with $C^* = 0$), excellent fits were obtained for $P = 2.6$ for fecal coliforms and $P = 2.2$ for somatic coliphage. An increase in rate constant with decreasing particle size is apparent. However, there is a greater increase with decreasing media uniformity.

### Background Concentrations

There is no reason to expect a background concentration for organisms that originate from warm-blooded animals, such as fecal coliforms, in SSF systems. The opportunity for organisms to reach the water within the bed from external sources is very limited. Nonetheless, the outdoor environment of the SSF wetland is not sterile, and various bacteria may be found in natural systems, absent any effect of wastewater.

It is possible to gain some insight about background concentrations in three ways: from data on sterile water feeds, from operations at extremely long detention times, and from the possibility of an extrapolation limit on effluent organism counts as inlet counts are lowered, across multiple systems. Thurston *et al.* (2001) pursued the former method and introduced well water into a *Typha–Scirpus* HSSF wetland. The outlet averaged TC = 130 #/100 mL and fecal coliform = 22

**TABLE 12.15**
**Effect of Depth on the Performance of Shallow and Deep Gravel HSSF Wetlands for Pathogen Reduction at a North Carolina Home Site**

| | Shallow Bed (30 cm) | | | Deep Bed (60 cm) | | |
|---|---|---|---|---|---|---|
| | **HLR** | **HRT** | ***P*-value** | **HLR** | **HRT** | ***P*-value** |
| | 2.43 cm/d | 3.4 d | 6 | 2.43 cm/d | 7.4 d | 6 |
| | **$\log_{10}$ Reduction** | ***k* (m/yr)** | **$k_V$ ($d^{-1}$)** | **$\log_{10}$ Reduction** | ***k* (m/yr)** | **$k_V$ ($d^{-1}$)** |
| Total coliforms | 4.0 | 194 | 6.43 | 2.4 | 80 | 1.23 |
| Fecal coliforms | 4.2 | 213 | 7.08 | 2.4 | 80 | 1.23 |
| Enterococci | 2.4 | 80 | 2.67 | 1.5 | 41 | 0.63 |
| *Clostridium perfringens* | 1.6 | 45 | 1.50 | 2.0 | 61 | 0.94 |
| Somatic coliphage | 2.7 | 97 | 3.21 | 1.2 | 31 | 0.47 |
| F+ coliphage | 1.0 | 25 | 0.83 | 0.3 | 6 | 0.10 |

*Source:* Reductions reported by Barrett *et al.* (2001) *Water Science and Technology* 44(11–12): 177–182.

#/100 mL over an 18-month period of operation. Operating data from some HSSF wetlands indicate that pathogen decline may be to a nonzero background concentration ($C^*$). For instance, a HSSF wetland operated in Craven County, North Carolina, returned an effluent concentration of 62 CFU/100 mL, despite an extremely long nominal hydraulic retention time of 56 days (Myers, 1993). This is comparable to the (Thurston *et al.*, 2001) result. The Water Environment Research Foundation (WERF) database contains outlet fecal coliform data for a large number of HSSF wetlands. Out of 4,400 reported values, 15% are less than fecal coliform = 3 CFU/100 mL. On the intersystem plot of outlet fecal coliform versus inlet fecal coliform (Figure 12.12), there is no plateau at low inlet fecal coliform, but rather a steady decline in outlet fecal coliform. However, this intersystem data does not extend to very low inlet concentrations.

It appears that if there is a background fecal coliform concentration for HSSF systems, it is quite low, probably <10 per 100 mL. That level does not affect the performance of wetlands that receive $10^5$–$10^6$ per 100 mL. Accordingly, the analysis of data herein assigns $C^* = 0$.

The existing data sets for fecal coliform reductions in VF wetlands are much more limited, with the largest body of knowledge for VF wetlands designed on the pulse loading system, commonly used in Europe. Flow in these wetlands is a complex combination of saturated and unsaturated flow (see Chapter 2). As a result, factors such as the loading/resting frequency, hydraulic loading rate, dose interval, and bed media

**TABLE 12.16**
**Effect of Particle Size and Uniformity on the Reduction of Fecal Coliforms and Somatic Coliphage in Side-by-Side Studies in HSSF Wetlands**

| Wetland | Media (mm) | Vegetation | Nominal HRT (d) | Size (mm) | Uniformity Coefficient | Maximum $\text{Log}_{10}$ Reduction | *k* (m/yr) | $k_V$ ($d^{-1}$) |
|---|---|---|---|---|---|---|---|---|
| **Fecal Coliforms** | | | | | | | | |
| Bed A | Gravel, 5–1 | *Phragmites* | 0.5–3 | 9.5 | 2.0 | 2.21 | 459 | 4.85 |
| Bed C | Gravel, 5–20 | *Phragmites* | 0.5–3 | 12.7 | 2.7 | 2.11 | 350 | 4.38 |
| Bed B | Gravel, 2–25 | *Phragmites* | 0.5–3 | 17.0 | 2.0 | 1.63 | 312 | 3.29 |
| Bed D | Gravel, 2–13 | *Phragmites* | 0.5–3 | 9.5 | 8.1 | 2.77 | 602 | 9.82 |
| **Somatic Coliphage** | | | | | | | | |
| Bed A | Gravel, 5–13 | *Phragmites* | 1.0–5.0 | 9.5 | 2.0 | 1.60 | 380 | 4.02 |
| Bed C | Gravel, 5–20 | *Phragmites* | 1.0–5.0 | 12.7 | 2.7 | 1.20 | 148 | 1.85 |
| Bed B | Gravel, 2–25 | *Phragmites* | 1.0–5.0 | 17.0 | 2.0 | 1.10 | 137 | 1.45 |
| Bed D | Gravel, 2–13 | *Phragmites* | 1.0–5.0 | 9.5 | 8.1 | 2.51 | 530 | 8.65 |

*Note:* Rate constants fit with $P = 2.6$ (fecal coliform), 2.2 (SC), and $C^* = 0$. The $R^2 > 0.97$.

*Source:* Data from García *et al.* (2003b) *Water Research* 37: 2645–2653.

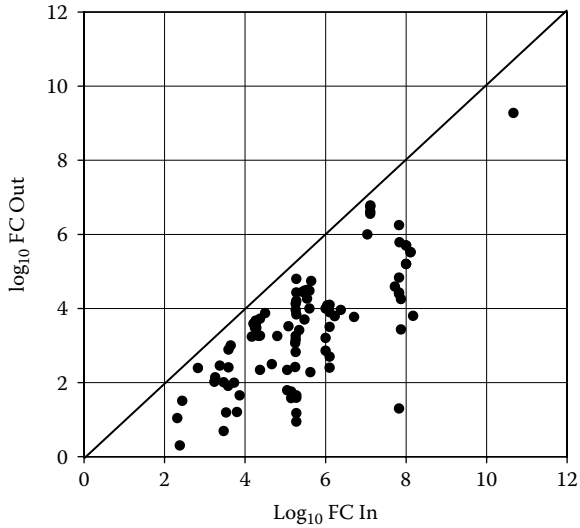



**FIGURE 12.12** Outlet fecal coliforms versus inlet fecal coliforms for HSSF wetlands. There are 97 data points for 88 different wetlands.

size all play an important role in the reduction of pathogens, including fecal coliforms. As a result of these interactions, it is difficult to generalize about the effectiveness of these systems in pathogen reductions. Table 12.13 lists example fecal coliform removals (input–output data).

For the six unsaturated downward flow systems listed in Table 12.13, the mean global removal was a 2.55 $\log_{10}$ reduction in fecal coliforms.

## First-Order Removal Models

### Longitudinal Profiles

Global first-order models forecast a decline in pathogens along the wetland flow path (provided the influent concentrations exceed the background concentrations for the system). This has been demonstrated for HSSF wetlands, as indicated in Figure 12.13.

As discussed previously for FWS wetlands, first-order models are traditionally used to account for pathogen inactivation, and such models are investigated here. There is adequate evidence that in the absence of artifacts of flow and mixing, exponential decays in the numbers of pathogenic organisms are to be expected in HSSF wetlands. For instance, Gersberg *et al.* (1987) studied inactivation of MS2 bacteriophage in dialysis bags located in the gravel matrix of a HSSF (bulrush) wetland at Santee, California. Log-linear decays were found, with over 3–5 $\log_{10}$ removal ranges (Figure 12.14). Although the study did not include the effect of gravel, it does indicate that water-phase inactivation is first order. The HSSF wetlands at Richmond, New South Wales, Australia, were tracer tested. When the data were analyzed, the gravel only and *Typha* in gravel wetlands were found to have *N*TIS = 25 and 21, respectively. These nearly plug flow systems may have resulted from the L:W = 25 used in their construction. The longitudinal profiles of fecal coliforms in these two wetlands are nearly log-linear (Figure 12.13).

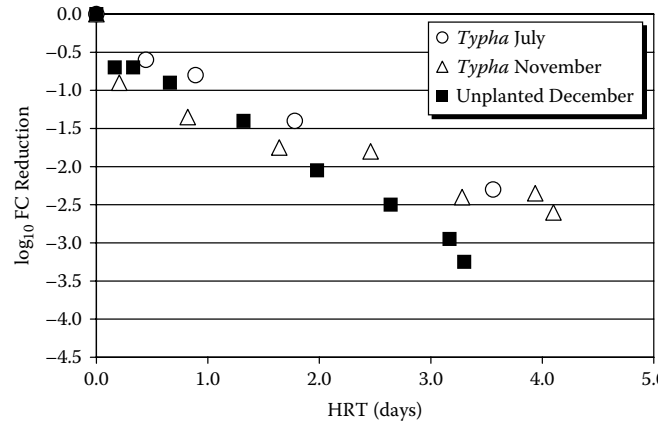



**FIGURE 12.13** Reduction of fecal coliforms at Richmond, New South Wales. Both wetlands were long and narrow (L:W = 25), and tracer test results indicated *N*TIS > 20. (Data from Bavor *et al.* (1988) *Treatment of secondary effluent*. Report to Sydney Water Board, Sydney, NSW, Australia.)

However, as noted in Chapters 2 and 6, flow in most HSSF systems is subject to greater nonplug flow effects. There is often enough nonideality to seriously affect removal behavior when reductions reach two or more $\log_{10}$ (99%).

Figure 12.5 is also valid for HSSF systems and demonstrates that exponential behavior is not to be expected even for fairly high *N*TIS values, e.g., *N*TIS = 8. García *et al.* (2003b) explored the effect of changing HRT for four HSSF wetlands for fecal coliform and somatic phage reduction. The expected deviation from exponential behavior can be found (see Figure 12.15).

In addition to the purely hydraulic effect of imperfect flow patterns, most pathogens are in fact mixtures of many species of organisms, and often occur in free form and in association with solids (TSS). Therefore, it is also expected that the mixture of organisms (e.g., fecal coliforms) will weather, as discussed in Chapter 6. The easier removals, which include

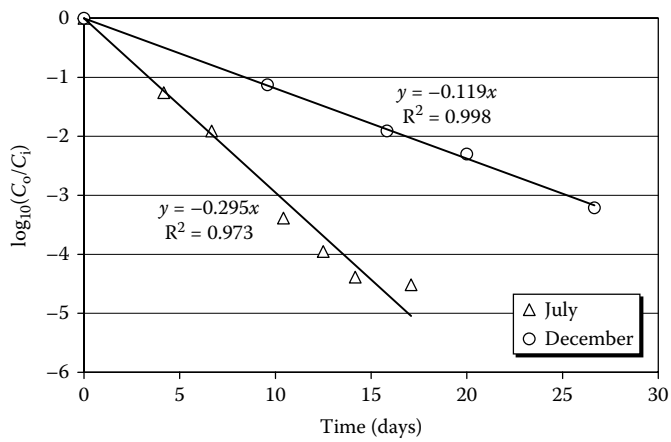



**FIGURE 12.14** The decay of bacteriophage MS2 in *in-situ* batch studies in the HSSF gravel bed environment at Santee, California. The values of $k_V$ are 0.27 and 0.68 $d^{-1}$ for December and July, respectively. (Data from Gersberg *et al.* (1989) In *Constructed Wetlands for Wastewater Treatment: Municipal, Industrial, and Agricultural.* Hammer (Ed.), Lewis Publishers, Chelsea, Michigan, pp. 431–445. Reprinted with permission.)

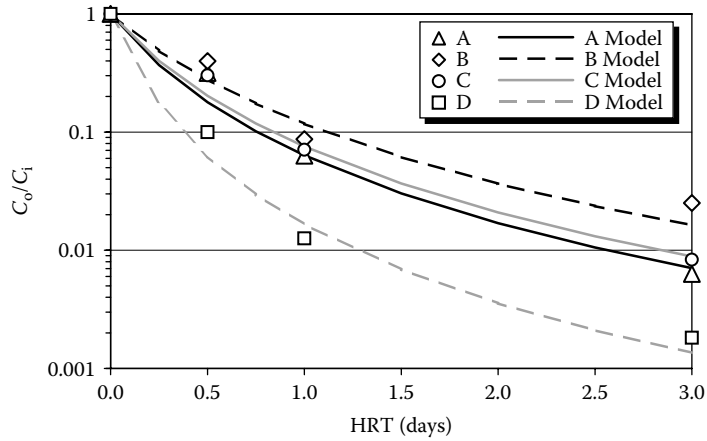



**FIGURE 12.15** *P*TIS model of the fecal coliform data from the four wetlands at Montcada i Reixac, Spain. The $R^2$ values range from 0.998 to 1.000. The *k*-values are 459, 312, 350, and 602 for A, B, C, and D, respectively, for *P*TIS = 2.6. The plug-flow model would be a straight line on this plot. (Data from García *et al.* (2003b) *Water Research* 37: 2645–2653.)

TSS pathogens, are likely to occur first, whereas die-off of specific organisms may occur later. The effect of weathering is the reduction of the *P*TIS value that is representative of pathogen removal.

Very different flow paths and conditions are typically present in HSSF and VF systems. HSSF wetlands use a saturated flow path, where the hydraulic residence time (τ) can be estimated (through nominal parameters) or measured (through tracer testing). In contrast, VF wetlands employ a variety of loading/resting schemes. In Europe, the most common mode of operation is a single-pass, pulse-loaded mode; in North America, recirculating systems with intermittent dosing are most common. The combined effect of these loading/resting modes is that movement of water through VF wetlands is a complex interaction of saturated and unsaturated flow mechanisms (see Chapter 2). As a result of these hydraulic differences, it is not possible to represent HSSF and VF pathogen removals using a common set of parameters.

## Rate Constants

The data from a number of HSSF wetlands has been analyzed to calculate first-order areal removal rate constants for use in the *P-k-C** model. Because tracer tests are typically unavailable for these wetlands, a value of $P = 6$ has been chosen. This is based on a median *N*TIS value of 8 (see Chapter 6), reduced to $P = 6$ to account for weathering phenomena. The value of $C^* = 0$ has been chosen. There is a wide range of *k*-values so calculated (see Tables 12.17–12.19). For fecal coliforms, the median $k = 103$ m/yr, corresponding to a $\log_{10}$ reduction of 1.82. For total coliforms, the median $k = 112$ m/yr, corresponding to a $\log_{10}$ reduction of 2.04. For various viruses, the median $k = 67$ m/yr, corresponding to a $\log_{10}$ reduction of 1.60. Importantly, the *k*-values are very sensitive to wetland hydraulics, and extrapolation to high removals is not acceptable for design purposes.

## Seasonal Trends

Some HSSF wetlands display seasonal effects for fecal coliform removal, with lower removal efficiencies during winter months at lower water temperatures. Data from Grand Lake,

**TABLE 12.17**
**Period of Record Reduction of Fecal Coliforms in HSSF Wetlands**

**Stipulations**

1. Period of record averages are used in calculations.
2. For *k*-value calculations, the following *P-k-C** parameters are selected:
   a. $C^* = 0.0$ #/100 mL
   b. $P = 6$ TIS
3. Ranges of variables:

| | HLR (cm/d) | $\log_{10}$ FC In (CFU/100 mL) | $\log_{10}$ FC Out (#/100 mL) |
|---|---|---|---|
| **Mean** | 7.8 | 5.50 | 3.52 |
| **Median** | 5.1 | 5.28 | 3.52 |
| **Maximum** | 51.8 | 10.67 | 9.28 |
| **Minimum** | 0.1 | 2.32 | 0.30 |

**Results (N = 130 wetlands/operating conditions).**

| Percentile | $\log_{10}$ Reduction (CFU/100 mL) | Rate Constant (m/yr) |
|---|---|---|
| 0.05 | 0.56 | 8 |
| 0.10 | 0.64 | 24 |
| 0.20 | 0.92 | 43 |
| 0.30 | 1.11 | 56 |
| 0.40 | 1.34 | 65 |
| 0.50 | 1.82 | 103 |
| 0.60 | 2.10 | 138 |
| 0.70 | 2.43 | 181 |
| 0.80 | 2.80 | 251 |
| 0.90 | 3.40 | 344 |
| 0.95 | 3.69 | 455 |

## TABLE 12.18
## Period of Record Reduction of Total Coliforms in HSSF Wetlands

**Stipulations**

1. Period of record averages are used in calculations.
2. For k-value calculations, the following *P-k-C** parameters are selected:
   a. $C^* = 0.0$ #/100 mL
   b. $P = 6$ TIS
3. Ranges of variables:

| | HLR (cm/d) | $\log_{10}$ FC In (CFU/100 mL) | $\log_{10}$ FC Out (#/100 mL) |
|---|---|---|---|
| **Mean** | 8.9 | 6.47 | 4.53 |
| **Median** | 5.1 | 6.58 | 4.21 |
| **Maximum** | 60.0 | 10.71 | 9.23 |
| **Minimum** | 0.6 | 3.09 | 1.83 |

**Results (N = 54 wetlands).**

| Percentile | $\log_{10}$ Reduction (CFU/100 mL) | Rate Constant (m/yr) |
|---|---|---|
| 0.05 | 0.33 | 21 |
| 0.10 | 0.51 | 28 |
| 0.20 | 1.01 | 43 |
| 0.30 | 1.34 | 76 |
| 0.40 | 1.62 | 91 |
| 0.50 | 2.04 | 112 |
| 0.60 | 2.19 | 134 |
| 0.70 | 2.43 | 201 |
| 0.80 | 2.54 | 322 |
| 0.90 | 3.56 | 510 |
| 0.95 | 3.99 | 800 |

## TABLE 12.19
## Period of Record Reduction of Viruses in HSSF Wetlands

**Stipulations**

1. A variety of different viruses and phages are represented.
2. Period of study averages are used in calculations.
3. For k-value calculations, the following *P-k-C** parameters are selected:
   a. $C^* = 0$
   b. $P = 6$ TIS
4. Ranges of variables:

| | HLR (cm/d) |
|---|---|
| **Mean** | 7.0 |
| **Median** | 4.6 |
| **Maximum** | 25.9 |
| **Minimum** | 0.6 |

**Results (N = 53 studies).**

| Percentile | $\log_{10}$ Reduction (CFU/100 mL) | Rate Constant (m/yr) |
|---|---|---|
| 0.05 | 0.14 | −1 |
| 0.10 | 0.45 | 11 |
| 0.20 | 0.72 | 21 |
| 0.30 | 1.07 | 34 |
| 0.40 | 1.20 | 46 |
| 0.50 | 1.60 | 67 |
| 0.60 | 1.85 | 117 |
| 0.70 | 2.34 | 129 |
| 0.80 | 2.92 | 225 |
| 0.90 | 3.99 | 365 |
| 0.95 | 4.00 | 477 |

Minnesota, show this pattern (Kadlec, 2004). Similar seasonal patterns (lower fecal coliform removal at lower water temperatures) have been observed for HSSF wetlands at Iselin, Pennsylvania (Watson *et al.*, 1987); Audlem, United Kingdom (Rivera *et al.*, 1995); and at NERCC, Minnesota (Axler *et al.*, 2001).

But there is no common theme for the variation of performance across the seasons. In some cases, there is a strong seasonal variation (Figure 12.16), whereas other HSSF wetlands display no such variation (Figure 12.17). The average amplitude of variation is 0.15 $\log_{10}$, and such trends account for only an average of 18% of the variability (Table 12.20).

### Temperature Coefficients

There are conflicting potential effects of temperature on pathogen reduction. Cold temperatures are inimical to the survival of organisms that originate in warm-blooded organisms, and thus human pathogens usually do not survive well outside the body. Survival times in waters are typically 10–30 days (Crites and Tchobanoglous, 1998). On the other hand, the activity of predatory organisms, such as protozoa, is lessened at cold temperatures, although direct lethality is unlikely (Davies-Colley *et al.*, 2005). Removal by adsorption is expected to be insensitive to temperature but slightly less effective under cold conditions. The very high temperature coefficients suggested for ponds and FWS wetlands ($\theta = 1.19$, Shilton and Mara, 2005; U.S. EPA, 2000a) are unlikely to prevail in the subsurface environment and do not apply.

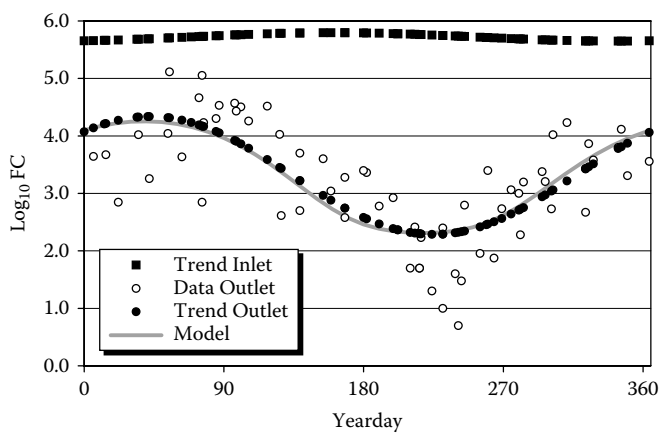



**FIGURE 12.16** Fecal coliform trends and model for the NERCC, Minnesota, HSSF wetland 1. The trends are cosine fits to the data. The model is the *P-k-C** with $P = \infty$, $k = 88$ m/yr, $C^* = 150$ per 100 mL, $\theta = 1.073$, $R^2 = 0.86$.

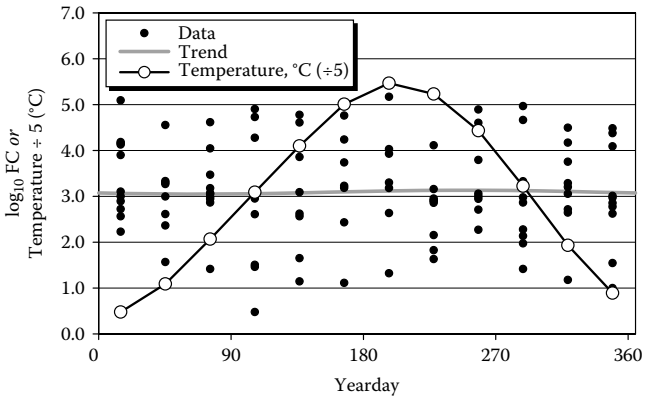



**FIGURE 12.17** Fecal coliform trend and data for the Tuckerman, Arkansas, HSSF wetland. The trend is a cosine fit to the data $R^2 = 0.001$. The 30-year normal mean monthly regional air temperature is also shown. (Data from WERF database (2006) *Small-Scale Constructed Wetland Treatment Systems Database (Project 01-CTS-5;* Final Report by Wallace and Knight, 2006). Compiled by J. Nivala and R. Clarke. Water Environment Research Foundation (WERF): Alexandria, Virginia.)

A number of HSSF wetlands have a median of $\theta = 1.002$ (Table 12.21). However, six systems display high values ranging from 1.043 to 1.073. The disparity in temperature effects is illustrated in Figures 12.16 and 12.17. There is no apparent relation that can explain the differences, because both high and low theta values occur across a variety of conditions.

### Variability

Stochastic variability in bacterial indicator organisms leaving HSSF wetlands is very large. Effluent fecal coliform concentrations are due to input loadings, decline of fecal coliform populations within the wetland, and potential reintroduction by wildlife. As a result, excursions around mean performance (Figures 12.16 and 12.17) should be expected and acknowledged in the design and regulation of HSSF wetland systems.

There is a very large degree of variation in organism numbers about seasonal trends, if any. Because of this, the $\log_{10}$ of the numerical counts is used to analyze variation. The $\log_{10}$ of the fecal coliform count at the 95% excursion level has a median of 1.55 times the trend value (Table 12.20). Suppose the trend outlet number density is 1,000 CFU/100 mL and thus the $\log_{10}$ FC = 3.00. At the 95% excursion level, the multiplier is 1.55 (Table 12.20), and the $\log_{10}$ FC = 4.65 to contain excursions at that frequency (one in twenty). Thus, the outlet number would be 44,668 CFU/100 mL, or over 40 times higher than the trend.

## 12.7 REMOVAL OF OTHER BACTERIA IN SSF WETLANDS

### Total Coliforms

The coliform group has been used as the standard for assessing fecal contamination of waters throughout the twentieth century (Gerba, 2000). However, many members of this broad group of organisms are not limited to fecal sources.

**TABLE 12.20**
**Annual Cyclic Trends and Trend Multipliers for the $\log_{10}$ of Concentration (CFU/100 mL) of Fecal Coliforms in the Effluent from HSSF Wetlands**

| System | Years | $\log_{10}$ Mean | $\log_{10}$ Amplitude | Yearday Maximum | $R^2$ | Excursion Frequency 50% | 80% | 95% | 99% |
|---|---|---|---|---|---|---|---|---|---|
| Hardin 1, Kentucky | 1 | 3.13 | 0.32 | 131 | 0.64 | 1.01 | 1.12 | 1.19 | 1.29 |
| Ola, Arkansas | 14 | 1.8 | 0.12 | 268 | 0.03 | 0.87 | 1.5 | 1.78 | 1.89 |
| Dierks, Arkansas | 12 | 2.24 | 0.09 | 237 | 0.01 | 0.72 | 1.79 | 1.92 | 1.98 |
| Tuckerman, Arkansas | 11 | 3.09 | 0.01 | 249 | 0 | 0.98 | 1.33 | 1.49 | 1.55 |
| Foreman, Arkansas | 14 | 1.74 | 0.15 | 295 | 0.05 | 0.88 | 1.35 | 1.72 | 1.95 |
| Gould, Arkansas | 14 | 2.92 | 0.02 | 319 | 0 | 0.99 | 1.33 | 1.46 | 1.55 |
| North Crossett, Arkansas | 14 | 2.34 | 0.04 | 178 | 0 | 1.11 | 1.65 | 1.78 | 1.88 |
| Gurndon, Arkansas | 14 | 1.88 | 0.12 | 204 | 0.04 | 0.93 | 1.32 | 1.55 | 1.74 |
| Grand Lake, Minnesota | 4 | 2.32 | 0.2 | 98 | 0.17 | 1.06 | 1.35 | 1.46 | 1.58 |
| North Yorkshire 1, England | 3 | 3.48 | 0.1 | 187 | 0.05 | 1.09 | 1.21 | 1.33 | 1.39 |
| Waipoua, New Zealand[a] | 4 | 2.53 | 0.09 | 183 | 0.06 | 1.07 | 1.22 | 1.25 | 1.36 |
| Benton 3, Kentucky | 1 | 1.88 | 0.37 | 174 | 0.3 | 1.02 | 1.26 | 1.39 | 1.46 |
| NERCC1, Minnesota | 4 | 3.31 | 0.31 | 43 | 0.47 | 1.05 | 1.17 | 1.27 | 1.32 |
| NERCC2, Minnesota | 4 | 3.45 | 0.23 | 72 | 0.39 | 1 | 1.21 | 1.25 | 1.31 |
| **Median** | **7.5** | **2.44** | **0.12** | **185** | **0.05** | **1.01** | **1.33** | **1.46** | **1.55** |
| **Mean** | **8.1** | **2.58** | **0.16** | **188** | **0.16** | **0.98** | **1.34** | **1.49** | **1.59** |

*Note:* Also shown are the multipliers that represent the frequency of excursions from the trend. For instance, the 95th percentile median $\log_{10}$ of concentration is 1.55 times higher than the trend. If the outlet trend value is 1,000 CFU per 100 mL, then one time out of twenty the outlet may reach $1{,}000^{1.55} = 44{,}700$ CFU per 100 mL. Trend multiplier is $(1 + \Psi)$; see Equation 6.61.

[a] Date adjusted 182 days.

**TABLE 12.21**
**Temperature Coefficients for Reduction of Pathogens in HSSF Wetlands**

| | Reference | Bed Designation | Temperature Range (°C) | Mean HLR (cm/d) | Mean $C_i$ (#/100 mL) | Mean $C_o$ (#/100 mL) | Theta |
|---|---|---|---|---|---|---|---|
| ***E. coli*** | | | | | | | |
| North Yorkshire 1, England | CWA database (2006) | 1 | 4–15 | 4.5 | 1.75E + 04 | 3.30E + 03 | 1.046 |
| **Total Coliforms** | | | | | | | |
| North Yorkshire 1, England | CWA database (2006) | 1 | 4–15 | 4.5 | 6.71E + 04 | 1.78E + 04 | 0.977 |
| **Fecal Coliforms** | | | | | | | |
| NERCC, Minnesota | Unpublished data | 1 | 1–16 | 1.4 | 3.47E + 05 | 4.69E + 03 | 1.073 |
| NERCC, Minnesota | Unpublished data | 2 | 1–16 | 1.4 | 3.47E + 05 | 7.44E + 03 | 1.073 |
| Grand Lake, Minnesota | Unpublished data | 1 | 1–17 | 1.0 | 2.22E + 05 | 2.60E + 03 | 1.043 |
| North Yorkshire 1, England | CWA database (2006) | 1 | 4–15 | 4.5 | 4.99E + 04 | 1.55E + 04 | 0.999 |
| Lincoln, Nebraska | Vanier and Dahab (1997) | 1 | 4–20 | 2.7 | 1.02E + 06 | 1.48E + 04 | 0.993 |
| Benton, Kentucky | TVA unpublished | 3 | 5–25 | 8.4 | 1.17E + 04 | 9.37E + 02 | 0.965 |
| Hardin, Kentucky | TVA unpublished | 1 | 6–27 | 9.7 | 2.36E + 05 | 4.34E + 03 | 0.952 |
| Waipoua, New Zealand | Unpublished data | 1 | 11–21 | 0.4 | 7.41E + 06 | 4.45E + 05 | 1.012 |
| Richmond, New South Wales | Bavor *et al.* (1988) | Gravel | 11–24 | 3.8 | 1.77E + 06 | 1.09E + 04 | 1.047 |
| Richmond, New South Wales | Bavor *et al.* (1988) | *Typha* | 11–24 | 4.6 | 1.77E + 06 | 1.68E + 04 | 1.002 |
| Richmond, New South Wales | Bavor *et al.* (1988) | *Schoenoplectus* | 11–24 | 5.1 | 1.77E + 06 | 2.24E + 04 | 1.054 |
| Richmond, New South Wales | Bavor *et al.* (1988) | Mixed A | 11–24 | 4.6 | 1.77E + 06 | 2.30E + 03 | 0.995 |
| Richmond, New South Wales | Bavor *et al.* (1988) | Mixed B | 11–24 | 3.8 | 1.77E + 06 | 2.44E + 03 | 0.987 |

| FC Percentiles | Theta |
|---|---|
| 0.05 | 0.960 |
| 0.10 | 0.969 |
| 0.20 | 0.989 |
| 0.30 | 0.994 |
| 0.40 | 0.998 |
| 0.50 | 1.002 |
| 0.60 | 1.018 |
| 0.70 | 1.045 |
| 0.80 | 1.051 |
| 0.90 | 1.069 |
| 0.95 | 1.073 |

Accordingly, fecal coliforms have become the preferred indicator, although total coliform reductions are widely reported in the literature. Examples of reduction of total coliforms in HSSF wetlands are reported in Table 12.22.

For a larger database of 54 HSSF wetlands/operating conditions, the mean global removal was a 2.04 $\log_{10}$ reduction in total coliforms. Removal mechanisms for total coliform bacteria are similar to those previously discussed for fecal coliforms. For nine studies of unsaturated downward VF systems, the mean global removal was a 2.02-log reduction in total coliforms (not shown).

### Miscellaneous Bacteria

A variety of other pathogenic bacteria have been assessed in SSF wetlands. These include *Escherichia coli*, *Clostridium perfringes*, Enterobacteriaceae, Enterococci, *Salmonella*, *Shigella*, *Yersinia,* and fecal streptococcus. The pathology of these organisms has been discussed for FWS wetlands.

Example data indicate that SSF wetlands are also effective in the reduction of these pathogenic organisms (Table 12.23).

Table 12.23 lists bacterial removals based on the period of record (POR) available from literature sources. For the 44 HSSF systems listed in Table 12.23, the mean global removal for these bacteria was a 2.21-log reduction. This is very similar to the reductions observed for fecal and total coliforms, presumably because the same removal mechanisms are operative. Broken down by species, these reductions are *Escherichia coli* (2.53-log), *Clostridium perfringes* (2.04-log), Enterobacteriaceae (1.59-log), Enterococci (1.40-log), *Salmonella* (2.47-log), *Shigella* (2.00-log), *Yersinia* (1.60-log), and fecal streptococci (2.43-log). It should be noted that the data in Table 12.23 do not represent a guarantee

## TABLE 12.22
## Reduction of Total Coliforms for HSSF Wetlands

| System Name and Location | Source | Media $d_{10}$ (mm) | Vegetation | Nominal HRT (estimated) (d) | Nominal HLR (estimated) (cm/d) | $\log_{10}$ Inlet (CFU/100 mL) | $\log_{10}$ Outlet (CFU/100 mL) | $\log_{10}$ Reduction |
|---|---|---|---|---|---|---|---|---|
| Abu Attwa, Egypt | Williams *et al.* (1995) | Gravel, 10–20 mm | *Phragmites* | 6.9 | 1.1 | 7.99 | 5.53 | 2.46 |
| Budds Farm, United Kingdom | Williams *et al.* (1995) | Gravel, 10–20 mm | *Phragmites* | 9.6 | 0.8 | 6.45 | 3.97 | 2.48 |
| Dar Es Salaam, Tanzania | Kaseva (2004) | Gravel, 8–25 mm | None | 1.9 | 13.0 | 7.40 | 7.16 | 0.23 |
| Dar Es Salaam, Tanzania | Kaseva (2004) | Gravel, 8–25 mm | *Phragmites mauritianus* | 2.0 | 12.2 | 7.40 | 7.04 | 0.36 |
| Dar Es Salaam, Tanzania | Kaseva (2004) | Gravel, 8–25 mm | *Typha latifolia* | 2.0 | 12.1 | 7.40 | 7.02 | 0.38 |
| Florence, Italy | Masi *et al.* (2004) | Gravel, 5–10 mm | *Phragmites* | 3.0 | 17.0 | 6.88 | 4.07 | 2.81 |
| Nucice, Czech Republic | Vymazal *et al.* (2001) | Gravel, 8–16 mm | Mixed | 11.0 | — | 4.69 | 2.51 | 2.18 |
| Pachuca, Mexico | Rivera *et al.* (1997) | Gravel | *Phragmites* | 1.7 | 14.1 | 10.71 | 9.23 | 1.49 |
| Pathumthani, Thailand | Sawaittayothin and Polprasert (2007) | Gravel + Sand | *Typha angustifolia* | 1.0 | 20.0 | 3.22 | 2.66 | 0.56 |
| Pathumthani, Thailand | Sawaittayothin and Polprasert (2007) | Gravel + Sand | *Typha angustifolia* | 3.0 | 6.7 | 3.09 | 2.09 | 1.00 |
| Pathumthani, Thailand | Sawaittayothin and Polprasert (2007) | Gravel + Sand | *Typha angustifolia* | 5.0 | 4.0 | 3.12 | 1.88 | 1.25 |
| Pathumthani, Thailand | Sawaittayothin and Polprasert (2007) | Gravel + Sand | *Typha angustifolia* | 8.0 | 2.5 | 4.30 | 1.83 | 2.46 |
| San Michele de Ganzaria, Italy | Cirelli *et al.* (2006) | Gravel, 8–10 mm | *Phragmites* | 2.6 | 7.7 | 6.00 | 4.48 | 1.52 |
| Santee, California | Gersberg *et al.* (1989) | Gravel | *Schoenoplectus* | 1.6 | 18.0 | 4.96 | 2.83 | 2.13 |
| Santee, California | Gersberg *et al.* (1987) | Gravel | *Schoenoplectus* | 5.8 | 5.0 | 7.83 | 5.76 | 2.07 |
| Wickenburg, Arizona | Nokes *et al.* (2003) | Gravel | None | 3.3 | 3.6 | 6.46 | 4.45 | 2.02 |
| Sahuarita, Arizona | Nokes *et al.* (2003) | Gravel | *Scirpus* spp. | 3.4 | 5.9 | 5.60 | 3.89 | 1.72 |
| Tubac, Arizona | Nokes *et al.* (2003) | Gravel | *Typha* and *Scirpus* | 4.0 | 3.0 | 5.73 | 3.23 | 2.50 |
| Pima County, Arizona | Quiñónez-Díaz *et al.* (2001) | Gravel bottom | *Scirpus* spp. | 12.5 | 4.8 | 8.45 | 4.04 | 4.41 |
| Pima County, Arizona | Thurston *et al.* (2001) | Gravel | *Typha* and *Scirpus* | 16.7 | 3.8 | 4.23 | 2.04 | 2.19 |
| Langenreichenbach, Germany | Vacca *et al.* (2005) | Sand | *Phragmites* | 8.0 | 6.0 | 8.20 | 5.70 | 2.50 |
| Wiedersburg, Germany | Ulrich *et al.* (2005) | Sand | *Phragmites* | — | — | 4.70 | 3.08 | 1.62 |
| Richmond, New South Wales | Bavor *et al.* (1988) | Gravel | None | 4.7 | 3.9 | 6.58 | 4.04 | 2.54 |
| Richmond, New South Wales | Bavor *et al.* (1988) | Gravel | *Schoenoplectus* | 6.6 | 5.1 | 6.58 | 5.08 | 1.50 |
| Richmond, New South Wales | Bavor *et al.* (1988) | Gravel | *Typha* | 5.9 | 4.6 | 6.58 | 4.35 | 2.23 |
| Richmond, New South Wales | Bavor *et al.* (1988) | Gravel | Mix A | 7.8 | 4.6 | 6.58 | 2.60 | 3.98 |
| Richmond, New South Wales | Bavor *et al.* (1988) | Gravel | Mix B | 9.2 | 3.9 | 6.58 | 2.65 | 3.93 |
| Pretoria 1, South Africa | Batchelor *et al.* (1990) | Gravel | *Phragmites* | — | 19.5 | 8.18 | 7.02 | 1.16 |
| Pretoria 3, South Africa | Batchelor *et al.* (1990) | Topsoil | *Phragmites* | — | 8.7 | 8.18 | 2.31 | 5.87 |
| **Mean** | | | | | | **6.35** | **4.23** | **2.12** |
| **Median** | | | | | | **6.58** | **4.04** | **2.13** |

of treatment performance, because the data on which the table is based is extremely limited.

Table 12.24 lists bacterial removals for 20 studies/operating periods in VF systems. The mean global removal for these bacteria was a 2.63-log reduction. This is very similar to the reductions observed for fecal and total coliforms, presumably because the same removal mechanisms are operative. Broken down by species, these reductions are: *Escherichia coli* (2.66-log), *Clostridium perfringes* (2.80-log), Enterococci (2.30-log), *Salmonella* (3.84-log), and fecal streptococci (2.10-log). It should be noted that data in Table 12.24, similar to Table 12.23, do not represent a guarantee of treatment performance, because the data on which the table is based is extremely limited.

## 12.8 PARASITE AND VIRUS REMOVAL IN SSF WETLANDS

### Parasites

Parasites include protozoa and helminthes. In the United States, two protozoa of special interest are *Giardia lamblia* and *Cryptosporidium parvum*. Worldwide, *Ascaris lumbricoides* is the most prevalent parasitic infection. The pathology of these organisms has already been discussed for FWS wetlands.

HSSF wetlands are effective in reducing the number of eggs or oocysts of these organisms (Table 12.25). This is presumably due to a combination of settling, filtration, interception, and predation. Removal percentages reported in the

**TABLE 12.23**
**Reduction of Various Bacteria for Gravel HSSF Wetlands**

| Organism | System Name and Location | Reference | Vegetation | Nominal HRT (estimated) (d) | Nominal HLR (cm/d) | $\log_{10}$ Reduction |
|---|---|---|---|---|---|---|
| *Clostridium perfringens* | North Carolina home (0.3 m bed depth) | Barrett *et al.* (2001) | Mixed | 3.4 | 2.43 | 1.60 |
| *Clostridium perfringens* | North Carolina home (0.6 m bed depth) | Barrett *et al.* (2001) | Mixed | 7.4 | 2.43 | 2.00 |
| *Clostridium perfringens* | North Carolina ranger station | Barrett *et al.* (2001) | Mixed | 3 | 3.12 | 2.40 |
| Enterobacteriaceae | Chmelna, Czech Republic | Ottová *et al.* (1997) | *Phalaris* | 1.7 | 11.0 | 1.28 |
| Enterobacteriaceae | Doksy, Czech Republic | Ottová *et al.* (1997) | *Glyceria* | 10.7 | 2.2 | 1.52 |
| Enterobacteriaceae | Kolodeje, Czech Republic | Ottová *et al.* (1997) | *Phragmites* | 4.2 | 4 | 1.70 |
| Enterobacteriaceae | Ondrejov, Czech Republic | Ottová *et al.* (1997) | *Phragmites* | 4.5 | 6.6 | 2.30 |
| Enterobacteriaceae | Onsov, Czech Republic | Ottová *et al.* (1997) | *Phragmites* | 4.7 | 4.5 | 1.17 |
| Enterococci | Alabama, United States (four systems) | Barrett *et al.* (2001) | Mixed | 2.25 | 5.15 | 0.80 |
| Enterococci | Morgantown, West Virginia | Hench *et al.* (2003) | Mixed | 7 | 1.5 | 2.30 |
| Enterococci | North Carolina home (0.3 m bed depth) | Barrett *et al.* (2001) | Mixed | 2.25 | 5.15 | 2.40 |
| Enterococci | North Carolina home (0.6 m bed depth) | Barrett *et al.* (2001) | Mixed | 2.25 | 5.15 | 1.50 |
| Enterococci | North Carolina ranger station | Barrett *et al.* (2001) | Mixed | 2.25 | 5.15 | 0.30 |
| Enterococci | North Carolina ranger station | Barrett *et al.* (2001) | Mixed | 2.25 | 5.15 | 0.60 |
| *Escherichia coli* | Audlem, United Kingdom | Decamp and Warren (2000) | *Phragmites* | 2.1 | 9.2 | 2.94 |
| *Escherichia coli* | Finham, United Kingdom | Green *et al.* (1997) | *Phragmites* | 0.25 | 60 | 1.31 |
| *Escherichia coli* | Finham, United Kingdom | Ottová *et al.* (1997) | *Phragmites* | 0.5 | 30 | 2.25 |
| *Escherichia coli* | Finham, United Kingdom | Ottová *et al.* (1997) | *Phragmites* | 1 | 15 | 2.43 |
| *Escherichia coli* | Finham, United Kingdom | Ottová *et al.* (1997) | *Phragmites* | 2 | 7.5 | 3.10 |
| *Escherichia coli* | Finham, United Kingdom | Ottová *et al.* (1997) | *Phragmites* | 5 | 3 | 2.87 |
| *Escherichia coli* | Florence, Italy | Masi *et al.* (2004) | *Phragmites* | 3.00 | 17 | 3.14 |
| *Escherichia coli* | San Michele de Ganzaria, Italy | Cirelli *et al.* (2006) | *Phragmites* | 2.6 | 7.7 | 1.70 |
| Fecal streptococci | Florence, Italy | Masi *et al.* (2004) | *Phragmites* | 3.00 | 17 | 3.17 |
| Fecal streptococci | Nucice, Czech Republic | Vymazal *et al.* (2001) | Mixed | 11 | — | 2.67 |
| *Salmonella* | Budds Farm, United Kingdom | Williams *et al.* (1995) | *Phragmites* | 9.6 | 0.79 | 2.12 |
| *Salmonella* | Budds Farm, United Kingdom | Williams *et al.* (1995) | *Phragmites* | 13 | 0.58 | 2.43 |
| *Salmonella* | Morgantown, West Virginia | Hench *et al.* (2003) | Mixed | 7 | 1.5 | 1.90 |
| *Salmonella* | San Michele de Ganzaria, Italy | Cirelli *et al.* (2006) | *Phragmites* | 2.6 | 7.7 | 4.00 |
| *Salmonella choleraesuis* | NERCC, Minnesota | Axler *et al.* (2001) | Mixed | 9 | 2.00 | 2.40 |
| *Salmonella choleraesuis* | NERCC, Minnesota | Axler *et al.* (2001) | Mixed | 9 | 2.00 | 1.30 |
| *Salmonella choleraesuis* | NERCC, Minnesota | Axler *et al.* (2001) | Mixed | 14 | 1.29 | 5.20 |
| *Salmonella choleraesuis* | NERCC, Minnesota | Axler *et al.* (2001) | Mixed | 14 | 1.29 | 1.40 |
| *Shigella* | Morgantown, West Virginia | Hench *et al.* (2003) | Mixed | 7 | 1.5 | 2.30 |
| *Vibrio cholerae* | Abu Attwa, Egypt | Williams *et al.* (1995) | *Phragmites* | 6.9 | 1.1 | 1.54 |
| *Yersinia* | Morgantown, West Virginia | Hench *et al.* (2003) | Mixed | 7 | 1.5 | 1.70 |

## TABLE 12.24
## Reduction of Various Bacteria for Gravel VF Wetlands

| System Name and Location | Reference | Media $d_{10}$ (mm) | Vegetation | HLR (m/d) | Organism | Inlet (CFU/ 100 mL) | Outlet (CFU/ 100 mL) | $\log_{10}$ Reduction |
|---|---|---|---|---|---|---|---|---|
| Aarhus, Denmark | Arias *et al.* (2003b) | Gravel 0–2 mm | *Phragmites* | 0.52 | Fecal streptococci | 5.3E + 05 | 1.0E + 05 | 0.71 |
| Aarhus, Denmark | Arias *et al.* (2003b) | Gravel 0–2 mm | *Phragmites* | 0.71 | Fecal streptococci | 2.7E + 05 | 2.6E + 04 | 1.02 |
| Aarhus, Denmark | Arias *et al.* (2003b) | Gravel 0–2 mm | *Phragmites* | 1.37 | Fecal streptococci | 1.6E + 05 | 1.3E + 04 | 1.09 |
| Aarhus, Denmark | Arias *et al.* (2003b) | Gravel 0–2 mm | *Phragmites* | 1.34 | Fecal streptococci | 1.6E + 05 | 2.3E + 04 | 0.84 |
| Leipzig, Germany | Wand *et al.* (2006) | Sand 0.25–3.15 mm | *Phragmites* | 0.23 | Fecal streptococci | 3.2E + 04 | n.d. | 4.00 |
| Leipzig, Germany | Wand *et al.* (2006) | Sand 0.25–3.15 mm | None | 0.23 | Fecal streptococci | 3.0E + 04 | 1.0E + 02 | 2.48 |
| Leipzig, Germany | Wand *et al.* (2006) | Sand 0.25–3.15 mm | *Juncus effusus* | 0.23 | Fecal streptococci | 2.0E + 04 | n.d. | 4.00 |
| Leipzig, Germany | Wand *et al.* (2006) | Sand 0.25–3.15 mm | None | 0.23 | Fecal streptococci | 2.4E + 04 | n.d. | 4.00 |
| Florence, Italy | Masi *et al.* (2004) | Sand 0–4 mm | *Phragmites* | 0.17 | Fecal streptococci | 5.35E + 02 | 8.6E + 01 | 0.79 |
| Leipzig, Germany | Wand *et al.* (2006) | Sand 0.25–3.15 mm | *Phragmites* | 0.23 | *Salmonellae* | 1.4E + 05 | 3.0E + 01 | 3.67 |
| Leipzig, Germany | Wand *et al.* (2006) | Sand 0.25–3.15 mm | None | 0.23 | *Salmonellae* | 1.5E + 05 | 3.0E + 01 | 3.70 |
| Leipzig, Germany | Wand *et al.* (2006) | Sand 0.25–3.15 mm | *Juncus effusus* | 0.23 | *Salmonellae* | 1.6E + 04 | n.d. | 4.00 |
| Leipzig, Germany | Wand *et al.* (2006) | Sand 0.25–3.15 mm | None | 0.23 | *Salmonellae* | 2.4E + 04 | n.d. | 4.00 |
| NC School 1, (0.3 m bed depth) | Barrett *et al.* (2001) | Sand–Gravel–Brick | *Juncus effusus* | 0.1 | Enterococci | — | — | 2.40 |
| NC School 2, (0.3 m bed depth) | Barrett *et al.* (2001) | Sand–Gravel–Brick | *Juncus effusus* | 0.1 | Enterococci | — | — | 2.20 |
| NC School 1, (0.3 m bed depth) | Barrett *et al.* (2001) | Sand–Gravel–Brick | *Juncus effusus* | 0.1 | *Clostridium perfringens* | — | — | 2.90 |
| NC School 2, (0.3 m bed depth) | Barrett *et al.* (2001) | Sand–Gravel–Brick | *Juncus effusus* | 0.1 | *Clostridium perfringens* | — | — | 2.70 |
| Aurignac, France | Torrens *et al.* (2007) | River sand (0.25 mm) | *Phragmites* | 0.2–0.8 | *Escherichia coli* | 7.4E + 02 | — | 1.61 |
| Aurignac, France | Torrens *et al.* (2007) | River sand (0.25 mm) | None | 0.2–0.8 | *Escherichia coli* | 9.7E + 02 | — | 1.49 |
| Aurignac, France | Torrens *et al.* (2007) | Crushed sand (0.19 mm) | None | 0.2–0.8 | *Escherichia coli* | 8.1E + 02 | — | 1.58 |
| Florence, Italy | Masi *et al.* (2004) | Sand 0–4 mm | *Phragmites* | 0.17 | *Escherichia coli* | 3.72E + 03 | 2.3E + 02 | 1.20 |
| Langenreichenbach, Germany | Baeder-Bederski *et al.* (2005) | LECA + Sand | *Phragmites* | 0.04 | *Escherichia coli* | — | — | 2.40 |
| Langenreichenbach, Germany | Baeder-Bederski *et al.* (2005) | Sand 0–2 mm | *Phragmites* | 0.04 | *Escherichia coli* | — | — | 3.60 |
| Lauwersoog, The Netherlands | Meuleman *et al.* (2003) | Sand | *Phragmites* | 0.4 | *Escherichia coli* | 5.0E + 04 | 1.9E + 01 | 3.43 |

*Note:* n.d. = non detect

**TABLE 12.25**
**Reduction of Various Parasites for Gravel HSSF Wetlands**

| Organism | System Name and Location | Reference | Media $d_{10}$ (mm) | Vegetation | Assay | Nominal HRT (Estimated) (d) | Nominal HLR (cm/d) | Percentage Reduction |
|---|---|---|---|---|---|---|---|---|
| *Ascaris lumbrioides* | Unam, Mexico City | Rivera *et al.* (1995) | Soil | *Phragmites* | Eggs/g of sludge | — | — | 83 |
| *Ascaris lumbrioides* | Unam, Mexico City | Rivera *et al.* (1995) | Soil | None | Eggs/g of sludge | — | — | 33 |
| *Ascaris lumbrioides* | Unam, Mexico City | Rivera *et al.* (1995) | Gravel | *Phragmites* | Eggs/g of sludge | — | — | 100 |
| *Ascaris lumbrioides* | Unam, Mexico City | Rivera *et al.* (1995) | Soil | *Phragmites* | Eggs/g of sludge | — | — | 67 |
| *Ascaris lumbrioides* | Unam, Mexico City | Rivera *et al.* (1995) | Soil | None | Eggs/g of sludge | — | — | 33 |
| *Ascaris lumbrioides* | Unam, Mexico City | Rivera *et al.* (1995) | Gravel | *Phragmites* | Eggs/g of sludge | — | — | 100 |
| *Ascaris* | Ismailia, Egypt | Stott *et al.* (2003) | Gravel | Mixed | Eggs/L | 0.2 | — | 99 |
| *Ascaris* | Ismailia, Egypt | Stott *et al.* (2003) | Gravel | Mixed | Eggs/L | 0.4 | — | 100 |
| *Ascaris* | Ismailia, Egypt | Stott *et al.* (1999) | Gravel | Mixed | Eggs | 0.2 | 7.2 | 100 |
| *Hymenolepis* | Ismailia, Egypt | Stott *et al.* (1999) | Gravel | Mixed | Eggs | 0.2 | 7.2 | 100 |
| *Toxocara* | Ismailia, Egypt | Stott *et al.* (1999) | Gravel | Mixed | Eggs | 0.2 | 7.2 | 100 |
| Helminth | San Michele de Ganzaria, Italy | Cirelli *et al.* (2006) | Gravel, 8–10 mm | *Phragmites* | Eggs/L | 2.6 | 7.7 | 100 |
| Giardia | Wickenburg, Arizona | Nokes *et al.* (2003) | Gravel | None | Cysts | 3.3 | 3.6 | 99 |
| Giardia | Sahuarita, Arizona | Nokes *et al.* (2003) | Gravel | *Scirpus* spp. | Cysts | 3.4 | 5.9 | 98 |
| Giardia | Pima County, Arizona | Quiñónez-Díaz *et al.* (2001) | Gravel bottom | *Scirpus* spp. | Cysts/L | 12.5 | 4.8 | 100 |
| Giardia | Pima County, Arizona | Quiñónez-Díaz *et al.* (2001) | Gravel bottom | None | Cysts/L | 12.5 | 4.8 | 100 |
| Giardia | Pima County, Arizona | Thurston *et al.* (2001) | Gravel | *Typha* and *Scirpus* | Cysts/100 L | 16.7 | 3.8 | 95 |
| Cryptosporidium | Wickenburg, Arizona | Nokes *et al.* (2003) | Gravel | None | Oocysts | 3.3 | 3.6 | 86 |
| Cryptosporidium | Sahuarita, Arizona | Nokes *et al.* (2003) | Gravel | *Scirpus* spp. | Oocysts | 3.4 | 5.9 | 39 |
| Cryptosporidium | Pima County, Arizona | Quiñónez-Díaz *et al.* (2001) | Gravel bottom | *Scirpus* spp. | Oocysts/L | 12.5 | 4.8 | 100 |
| Cryptosporidium | Pima County, Arizona | Quiñónez-Díaz *et al.* (2001) | Gravel bottom | None | Oocysts/L | 12.5 | 4.8 | 100 |
| Cryptosporidium | Pima County, Arizona | Thurston *et al.* (2001) | Gravel | *Typha* and *Scirpus* | Oocysts/100 L | 16.7 | 3.8 | 79 |

## TABLE 12.26
## Reduction of Various Viruses for Gravel HSSF Wetlands

| Organism | System Name and Location | Reference | Data Record | Vegetation | Nominal HRT (estimated) (days) | Nominal HLR (cm/d) | $Log_{10}$ Reduction |
|---|---|---|---|---|---|---|---|
| Coliphage | Abu Attwa, Egypt | Williams *et al.* (1995) | 1988–1990 | *Phragmites* | 6.9 | 1.1 | 2.50 |
| Coliphage | Budds Farm, United Kingdom | Williams *et al.* (1995) | 1988–1991 | *Phragmites* | 9.6 | 0.79 | 4.00 |
| Coliphage | Budds Farm, United Kingdom | Williams *et al.* (1995) | 1988–1991 | *Phragmites* | 13 | 0.58 | 4.00 |
| Coliphage | Morgantown, West Virginia | Hench *et al.* (2003) | 2 years | Mixed | 7 | 1.5 | 1.70 |
| Coliphage | Sahuarita, Arizona | Nokes *et al.* (2003) | April–October | *Scirpus* spp. | 3.4 | 5.9 | 0.65 |
| Coliphage | Tubac, Arizona | Nokes *et al.* (2003) | April–October | *Typha* and *Scirpus* | 4.0 | 3.0 | 1.84 |
| Coliphage | Pima County, Arizona | Quiñónez-Díaz *et al.* (2001) | April 1996–February 1997 | *Scirpus* spp. | 12.5 | 4.8 | 4.46 |
| Coliphage | Pima County, Arizona | Thurston *et al.* (2001) | February 1995–August 1996 | *Typha* and *Scirpus* | 16.7 | 3.8 | 1.73 |
| Coliphage | Morgantown, West Virginia | Hench *et al.* (2003) | 2 years | *Typha, Juncus,* and *Scirpus* | 7.0 | 1.5 | 1.70 |
| Coliphage | Pretoria 1, South Africa | Batchelor *et al.* (1990) | 1989–90 | *Phragmites* | — | 15.8 | 0.82 |
| Enteric viruses | Pima County, Arizona | Quiñónez-Díaz *et al.* (2001) | April 1996–February 1997 | *Scirpus* spp. | 12.5 | 4.8 | 3.94 |
| Enterovirus | Budds Farm, United Kingdom | Williams *et al.* (1995) | 1988–1991 | *Phragmites* | 9.6 | 0.79 | 4.00 |
| Enterovirus | Budds Farm, United Kingdom | Williams *et al.* (1995) | 1988–1991 | *Phragmites* | 13 | 0.58 | 4.00 |
| F+ coliphage | Alabama (four systems) | Barrett *et al.* (2001) | September 1997–July 1998 | Mixed | 2.25 | 5.15 | 0.43 |
| F+ coliphage | NC home (0.3 m bed depth) | Barrett *et al.* (2001) | September 1997–July 1998 | Mixed | 3.4 | 2.43 | 1.00 |
| F+ coliphage | NC home (0.6 m bed depth) | Barrett *et al.* (2001) | September 1997–July 1998 | Mixed | 7.4 | 2.43 | 0.30 |
| F+ coliphage | North Carolina ranger station | Barrett *et al.* (2001) | September–July 1998 | Mixed | 3 | 3.12 | –0.10 |
| FRNA phage MS2 | Santee, California | Gersberg *et al.* (1987) | — | Bulrush 1W | 6.1 | 5.0 | 3.07 |
| FRNA phage MS2 | Santee, California | Gersberg *et al.* (1987) | — | Bulrush 2Wa | 6.1 | 5.0 | 2.65 |
| FRNA phage MS2 | Santee, California | Gersberg *et al.* (1987) | — | Bulrush 2Wb | 6.1 | 5.0 | 3.49 |
| F-specific phages | Santee, California | Gersberg *et al.* (1987) | — | Bulrush | 6.1 | 5.0 | 1.98 |
| Polio type 1 | Santee, California | Gersberg *et al.* (1987) | — | Bulrush 1W | 6.1 | 5.0 | 4.38 |
| Polio type 1 | Santee, California | Gersberg *et al.* (1987) | — | Bulrush 2W | 6.1 | 5.0 | 3.87 |
| Somatic coliphage | Alabama (three systems) | Barrett *et al.* (2001) | September 1997–July 1998 | Mixed | 2.25 | 5.15 | 1.20 |
| Somatic coliphage | Barcelona, Spain (bed A) | García *et al.* (2003b) | November–May 2001 | *Phragmites* | 1 | 25.9 | 0.53 |
| Somatic coliphage | Barcelona, Spain (bed A) | García *et al.* (2003b) | November–May 2001 | *Phragmites* | 3 | 25.9 | 1.22 |
| Somatic coliphage | Barcelona, Spain (bed A) | García *et al.* (2003b) | November–May 2001 | *Phragmites* | 5 | 25.9 | 1.60 |
| Somatic coliphage | Barcelona, Spain (bed B) | García *et al.* (2003b) | November–May 2001 | *Phragmites* | 1 | 25.9 | 0.56 |
| Somatic coliphage | Barcelona, Spain (bed B) | García *et al.* (2003b) | November–May 2001 | *Phragmites* | 3 | 25.9 | 0.99 |
| Somatic coliphage | Barcelona, Spain (bed B) | García *et al.* (2003b) | November–May 2001 | *Phragmites* | 5 | 25.9 | 1.10 |
| Somatic coliphage | Barcelona, Spain (bed C) | García *et al.* (2003b) | November–May 2001 | *Phragmites* | 1 | 21.9 | 0.57 |
| Somatic coliphage | Barcelona, Spain (bed C) | García *et al.* (2003b) | November–May 2001 | *Phragmites* | 3 | 7.3 | 1.05 |
| Somatic coliphage | Barcelona, Spain (bed C) | García *et al.* (2003b) | November–May 2001 | *Phragmites* | 5 | 4.38 | 1.20 |
| Somatic coliphage | Barcelona, Spain (bed D) | García *et al.* (2003b) | November–May 2001 | *Phragmites* | 1 | 16.8 | 0.81 |
| Somatic coliphage | Barcelona, Spain (bed D) | García *et al.* (2003b) | November–May 2001 | *Phragmites* | 3 | 5.6 | 1.89 |
| Somatic coliphage | Barcelona, Spain (bed D) | García *et al.* (2003b) | November–May 2001 | *Phragmites* | 5 | 3.36 | 2.51 |
| Somatic coliphage | NERCC, Minnesota | Axler *et al.* (2001) | Summer (May–October) 1996–2000 | Mixed | 14 | 1.14 | 1.70 |
| Somatic coliphage | NERCC, Minnesota | Axler *et al.* (2001) | Winter (November–April) 1996–2000 | Mixed | 14 | 1.14 | 1.20 |
| Somatic coliphage | NC home (0.3 m bed depth) | Barrett *et al.* (2001) | September 1997–July 1998 | Mixed | 3.4 | 2.43 | 2.70 |
| Somatic coliphage | NC home (0.6 m bed depth) | Barrett *et al.* (2001) | September 1997–July 1998 | Mixed | 7.4 | 2.43 | 1.20 |
| Somatic coliphage | North Carolina ranger station | Barrett *et al.* (2001) | September 1997–July 1998 | Mixed | 3 | 3.12 | –0.20 |

literature range from 79 to 100% for water-phase contamination (Thurston *et al*., 2001; Stott *et al*., 2003; Cirelli *et al*., 2006).

At the time of this writing, there is not sufficient data to assess the performance of VF wetland systems for parasite removal. Presumably, these systems (especially pulse-loaded designs that employ unsaturated flow) would remove parasite cysts or eggs effectively. This is based on the observation that VF systems typically remove bacteria and viruses (which have a much smaller particle size—see Figure 12.1) as effectively as HSSF wetlands.

### Viruses

Because the potential infective dose of viruses can be so low, and viruses are hardier in natural environments than bacterial pathogens, there is considerable concern about their fate and transport in wetland systems. However, because viruses are much more costly and difficult to monitor analytically, they are less studied in the literature than bacterial pathogens. Reduction of various viral organisms in HSSF wetlands is summarized in Table 12.26.

Data from the 32 systems listed in Table 12.26 indicate that the global reduction of viruses is on the order of 1.47-log. For different virus classifications, this is broken down as coliphage (2.52-log), somatic coliphage (1.14-log), F+ coliphage (0.41-log), MS-2 (1.78-log), and enterovirus (4.00-log). It should be noted that the data presented in Table 12.26 is limited to available research and does not imply a guarantee of treatment performance.

Compared to FWS or HSSF wetlands, considerably less is known about the fate and transport of viral organisms in VF wetlands. Barrett *et al*., (2001) indicate that pulse-loaded unsaturated flow VF wetlands can reduce viral organisms on the order of 2-log. This can be subdivided into somatic coliphage (2.70-log) and F+ coliphage (1.30-log). A similar study treating pond effluents by Torrens *et al*. (2007) also produced somatic coliphage (1.18-log) and F+ coliphage reductions (0.64-log).

## SUMMARY

Pathogenic organisms are a common component of domestic wastewater. Their populations are highly variable depending on the health of the human population contributing to the waste. Wetland treatment systems, especially those that have long residence times (greater than about ten days), provide some disinfection. The extent of removal is strongly dependent on the hydraulic efficiency of the wetland. Vegetated wetlands appear to be slightly more effective for pathogen removal than facultative ponds and other natural treatment systems that have less physical contact between pathogens and solid (biofilm) surfaces.

Because of the generally low density of actual wastewater pathogens and the difficulty associated with their enumeration, indicator organisms such as fecal coliforms, *E. coli*, and MS-2 bacteriophage are used to monitor the probable pathogenic activity of wastewater discharged to the environment. Empirical evidence is available that demonstrates that a significant die-off of indicator organisms and pathogenic species occurs in wetland treatment systems. It is also evident that wetlands, especially FWS systems, contain numerous animals that contribute base populations of these same indicator organisms to surface water quality. These naturally occurring indicator organisms must be recognized during regulatory permitting and review of data from wetland treatment systems.

Available information indicates that wetlands are at least as effective as other natural treatment technologies in reducing viral pathogen populations. In their review of the fate of pathogens in wetland treatment systems, Krishnan and Smith (1987) concluded that “while data exist to indicate the potential for public health problems arising from wetland discharges, no incidences of disease resulting directly from such discharges have been identified.” That conclusion appears to have withstood the test of time and remains true some 20 years later.

# 13 Organic Chemicals

Organic compounds are major constituents of biochemical oxygen demand (BOD) or chemical oxygen demand (COD) in municipal wastewater, as discussed in Chapter 8. However, there is an increasing number of treatment wetlands that target specific individual or groups of organic chemicals. These chemicals pose a new and somewhat more difficult set of problems because of their possible toxicity to plants and the limitations of aerobic and anaerobic degradation. The major routes for removal of hydrocarbons from wetland waters include: volatilization, photochemical oxidation, sedimentation, sorption, biological (microbial) degradation, and plant uptake. Three types of microbial processes can contribute: fermentation, aerobic respiration, and anaerobic respiration. The general principles and chemistry of processes affecting carbon compounds are discussed in Chapter 8; see Equations 8.10 through 8.18.

Wetlands manufacture and contain a wide spectrum of organic compounds. These compounds range from small molecules such as methane to humic acids of very high molecular weight. Many wetland soils are organic in nature and possess an affinity for introduced organics, via sorption and other binding mechanisms. Aliphatic hydrocarbons, both straight-chain and branched, are present as natural waxes. As a result, trace (background) amounts of some hydrocarbons are present in all wetlands, whether constructed or natural.

## 13.1 PETROLEUM HYDROCARBONS

There is considerable information on the use of treatment wetlands in the petroleum industry. Some of the general principles and available data have been summarized in a 1998 industry report (Knight *et al*., 1997; Knight *et al*., 1999). However, that compilation did not focus on specific hydrocarbon classes, such as BTEX and its constituent components (benzene, toluene, ethyl benzene, and xylenes). Two other hydrocarbon designations of interest include Gasoline Range Organics (GRO) and Diesel Range Organics (DRO). There is some overlap and ambiguity in these designations. Typically, GRO consists of hydrocarbons with 6–9 carbon atoms, while DRO contains 10–40 carbon atoms (Chapple *et al*., 2002). Total Petroleum Hydrocarbons (TPH) is a measure of the sum of paraffinic and aromatic constituents.

### BTEX

#### Biodegradation

Biodegradation of BTEX in a wetland environment is complicated by the existence of biofilms on submerged plant parts, plant litter, and gravel. Although small in terms of mass per unit volume, these biofilms are very active in biodegradation, and consequently serve as important sinks for organics. In this aspect, treatment wetlands resemble conventional attached-growth treatment processes.

Chang *et al*. (2001) established that all BTEX constituents degrade rapidly (half-lives of one to two days) when inoculated with a microbial consortium conditioned with toluene. Their study showed that benzene, toluene, and ethylbenzene were directly consumed as carbon sources, while xylene was removed by co-metabolism.

Moore *et al*. (2002) measured both aerobic and anaerobic biodegradation of BTEX in peat. Aerobic degradation was tracked via oxygen consumption and carbon dioxide production, and averaged 56 mg/kg·d. Anaerobic degradation was inferred from the consumption of other electron acceptors, including nitrate, sulfate, iron, manganese, and methane. About one third of the BTEX loss could have been attributed to anaerobic degradation.

#### Volatilization

BTEX constituents are volatile and may be easily lost from water, especially shallow water bodies, such as FWS wetlands. Lee *et al*. (2004a) have reported that first-order loss rate constants for BTEX constituents exhibit a fixed ratio to the loss rate constant for benzene, independent of the water-mixing regime (Table 13.1). Further, the presence of more than one BTEX compound, or of surfactants, had only minor effects on this ratio.

Of direct interest are the estimates, based on wetland data, of Keefe *et al*. (2004a) for an FWS wetland in Arizona. Values were determined of $K_w$ = 0.015 m/hr (130 m/yr) for toluene. This is a relatively high rate, compared to other pollutants treated in FWS wetlands.

Moore *et al*. (2002) evaluated volatilization losses from field vapor collection in a peatland contaminated with BTEX. Losses averaged 2,500 mg/m$^2$·d over all seasons. This high rate may have been in part due to the existence of a Light Non-Aqueous Phase Layer (LNAPL) in the peat.

#### Plant Uptake

Willows (*Salix babylonica*) were shown to materially contribute to the removal of benzene from water under hydroponic conditions (Corseuil and Moreno, 2001). Volatilization was effectively suppressed in the experiments, as demonstrated by control mesocosms. About 80% reduction in benzene was found with an HRT of one day. Corseuil and Moreno suggested that benzene was initially sorbed onto root biomass, followed by plant uptake and biological degradation.

**TABLE 13.1**
**Volatility of BTEX and Chlorinated Benzenes Compared to Benzene**

| Compound | Volatilization Ratio |
|---|---|
| Benzene | 1.000 |
| Toluene | 0.966 |
| Ethylbenzene | 0.922 |
| *m*-Xylene | 0.930 |
| 1,3 Dichlorobenzene | 0.882 |
| 1,3,5 Trichlorobenzene | 0.569 |
| 1,2,3,4-Tetrachlorobenzene | 0.530 |
| Pentachlorobenzene | 0.493 |

*Note:* The volatilization ratio is the ratio of first-order loss rate coefficients.

*Source:* Data from Lee *et al.* (2004a) *Water Research* 38(2): 365–374.

## Sorption

BTEX components are expected to partition strongly to organic wetland substrates, although a definitive wetland study is lacking. Wetland sediments typically have high organic content, and therefore sorption may be an important first step in overall removal. Benzene sorption onto peat was found to be strong, with a linear sorption coefficient of $K_{OC}$ = 5.6 L/kg (Moore *et al*., 2002), whereas binding to clay was much weaker, with $K_{OC}$ = 0.12 L/kg.

## Wetland System Studies

In all existing petrochemical plant applications, wetlands have been accompanied by pretreatment. However, other applications, such as those involving remediation, do not involve pretreatment. Hydrocarbon contamination of groundwater includes closed and operating sanitary landfills, army ammunition plants, and former oil refinery sites. These facilities were often the recipient of solvents and other organics, which in the course of time have contaminated nearby waters. Concentrations of petrochemicals in landfill leachate are typically lower than those associated with refineries and terminals.

### *Gulf Strachan, Rocky Mountain House, Alberta*

HSSF wetlands were tested for the ability to reduce hydrocarbons, notably BTEX and TPH (Moore *et al*., 2000a). The wetlands were planted with *Phragmites australis* and *Typha latifolia*. Reductions of 40–60% were achieved within 14 days of detention time with no aeration, while aeration produced 100% removal in the same detention time. Influent BTEX concentrations ranged from 4.5–12.1 mg/L, and influent flows ranged from 7–33 L/min.

At the same site, Moore *et al*. (2002) reported on the natural attenuation of BTEX in a natural peatland, which received both aqueous and nonaqueous phase hydrocarbons for more than 15 years. No BTEX was detected leaving the peatland. Companion studies elucidated some of the candidate removal mechanisms, including sorption, aerobic degradation, volatilization, and anaerobic degradation. Aerobic degradation, which was stimulated by air injection, was the dominant removal mechanism.

### *Former Oil Refinery—United Kingdom*

HSSF wetlands planted with *Phragmites* were tested for the ability to reduce hydrocarbons, notably DRO (Chapple *et al*., 2002). Reductions of 40–64% were achieved in less than one day's detention time. Gravel-based beds performed better ($k$ = 270 m/yr) than soil-based beds ($k$ = 137 m/yr) for DRO removal.

### *Mobil Oil AG Terminal at Bremen, Germany*

The information here is taken from Altman *et al*. (1989), which summarizes three years of research results at Bremen. Highly contaminated wastewater (COD up to 14,000 mg/L) was brought through an API separator, a parallel plate separator, and a percolating reactor to a two-train pilot wetland system patterned after the concepts of Seidel (1966; 1973) at the Max Planck Institute. Each train had five subsurface flow wetlands in series, each being 2.5 m wide, 4.0 m long, and 0.8 m deep. Beds 1 and 2 were vertical flow, with passive aeration to the underdrains. Beds 3, 4, and 5 were horizontal flow. Each bed was filled with a stratified media, from the bottom: 5 cm of 8/16 mm (pea) gravel, 20 cm of 36/72 mm stone, 25 cm of 8/16 mm gravel, 10 cm of sharp sand, and topped with 2 cm of organic soil. Water is intermittently dosed to the surface, where it spreads and infiltrates, ultimately reaching the (10 cm) under-drains in the second layer. The passive air is admitted to the first layer via perforated plastic pipe. Details of this type of system are given in Vymazal *et al*. (1998). The plants that proved best for Bremen were cattails (*Typha angustifolia*) and bulrushes (*Scirpus lacustris*).

Hydrocarbon removal performance of this system was primarily measured as total hydrocarbon, but BTEX and its constituents were also measured less frequently. The influent BTEX in August was 3.8 mg/L (1.5 mg/L benzene), and the outflow contained no detectable BTEX, with all constituents being below 0.01 mg/L. The flow of 5 $m^3$/d corresponded to about 5 cm/d hydraulic loading, or about five days' detention. Although these results are encouraging, they do not answer the question of how much of the five-bed system was needed to achieve a given reduction.

However, inferences about BTEX removal may be made from the removal of total hydrocarbon (TH). Data were taken at the outlet from each bed of the trains, and therefore a removal model may be calibrated. A first-order areal TH model yields $k$ = 66 m/yr, with no evidence of temperature dependence. This is consistent with the benzene removal rate coefficient, which cannot be less than 90 m/yr for the data in the preceding paragraph.

### *Williams Pipeline, Watertown, South Dakota*

An aerated HSSF wetland was operated to reduce BTEX from petroleum contact waters. Complete removal of

hydrocarbons was reported (Wallace, 2002), despite high influent concentrations ($CBOD_5$ > 16,000 mg/L, $NH_4$-N > 200 mg/L, and BTEX > 1 mg/L), but this facility operated at an extremely low hydraulic loading (1 mm/d), and consequently had no water discharge.

### *ESSO, Chilliwack, British Columbia*

A cardlock facility produced stormwaters contaminated by diesel fuel. FWS wetlands were found successfully reducing water-phase diesel range organics to permit levels (Nix, 1995). Removal rate constants were about 10 m/yr.

### *Marathon-Pitchfork, Wyoming Produced Water Project*

The Colorado School of Mines tested a combined pilot system for hydrocarbon removal (Caswell *et al*., 1992). Overland flow gravel beds (shallow) were loaded at 20–80 cm/d, and reduced benzene from 15 to 2 µg/L. Further removal in SSF wetlands reduced benzene to below detection.

### *Isanti-Chisago, Minnesota Leachate Treatment System*

The Isanti-Chisago Sanitary Landfill, an unlined municipal solid waste facility located near Cambridge, Minnesota, was closed in 1992 (Loer *et al*., 1999; Kadlec, 2003c). Leaching of soluble wastes had contaminated the surficial and increasingly deeper aquifers with toxic organic compounds and heavy metals. Extraction wells permit pumping of leachate to the top surface of the landfill mound, about eight meters above the surrounding landscape. Volatile Organic Compounds (VOCs), including BTEX, are largely removed by cascading the water down the side of the landfill into a sedimentation basin which serves to settle and store iron precipitates. The next component of the treatment train is a 0.6-ha FWS wetland. During five seasons of treatment, operating results indicated that the system efficiency ranges from 85–100% for VOCs and 98% for iron.

Only low levels of BTEX enter the system: (benzene 3.7 µg/L; toluene 0.6 µg/L; ethylbenzene 1.1 µg/L, xylene 1.8 µg/L). The cascade and settling basin do all the BTEX removal, although the FWS wetland does polish out any remaining traces of BTEX. Benzene, toluene, ethyl benzene, and xylenes are all individually below detection (0.1 µg/L) in the system outflow.

### *Saginaw Township, Michigan, Leachate Treatment System*

Saginaw Charter Township's Center Road Landfill was closed in the early 1980s (Kadlec, 2003c). Finger drains were installed in the seepage zones, which connect to a collection pipe. Leachate is then pumped to an aerator, which provides some ammonia and BTEX stripping. The water is then held in a sedimentation basin, to promote removal of solids. Water is discharged periodically to one of two intermittent sand/gravel filters, which provide filtration and nitrification. Underdrains then convey the water to two parallel free water surface wetlands. BTEX and its constituents were monitored for five years after startup. A mean total BTEX of 39 µg/L was removed, with only one detection in weekly samples (5.9 µg/L).

### *Phoenix, Arizona Wastewater Polishing*

A demonstration wetland project was studied for VOC removal (Keefe *et al*., 2004a). The wetland was a 1.4-ha free water surface system of depth about 60 cm. The detention time was 3.9 days. An 80% reduction of toluene from inlet to outlet was measured. Keefe *et al*. (2004a) attributed the reduction to volatilization, but concluded that theoretical predictions were only good for order-of-magnitude estimation.

### *Alcoa, Tennessee Groundwater Remediation*

A pilot wetland project was initiated and operated for a period of over one year, in a moderate north temperate continental climate. DRO and GRO were among the targeted substances (Gessner *et al*., 2005). GRO entered at monthly average values of 0.04–0.37 mg/L, and never exceeded the detection limit of 0.01 mg/L at the outlet of the FWS wetland. DRO entered at monthly average values of 0.29–1.08 mg/L, and exited at 0.11–0.44 mg/L. GRO removal rate constants could not have been less than 100 m/yr. In contrast, the DRO removal rate constants were 12 m/yr annual average. The first wetlands at the site were SSF wetlands, which were abandoned in favor of FWS because of continual operational difficulties.

### *Casper, Wyoming, Groundwater Remediation*

A pilot scale subsurface vertical flow wetland system was constructed at the former British Petroleum Refinery in Casper, Wyoming, in order to determine benzene, toluene, ethylbenzene, and xylene (BTEX) degradation rates in a cold-climate application (Ferro *et al*., 2002). The pilot system, consisting of four cells, each dosed at a nominal flow rate of 5.4 $m^3$/d, was operated between August and December 2002. The pilot tested the effects of wetland mulch and aeration on system performance. Areal rate constants ($k_A$) were calculated based on an assumed three tanks in series (3TIS). The presence of both wetland sod and aeration improved treatment performance. Mean $k_A$ values were 244 m/yr for cells without sod or aeration, and improved to 356 m/yr for cells with sod and aeration (Table 13.2).

Based on the results of the pilot system, a full-scale wetland system (capable of operating at 6,000 $m^3$/d) was started up in May 2003 (Wallace and Kadlec, 2005). The full-scale system achieved permit compliance within one week of start-up.

### *University of Edinburgh, Scotland*

Bench scale vertical flow wetlands were operated to demonstrate benzene removal (Eke and Scholz, 2006). The systems were operated in a fill-and-drain batch mode, cycling twice per week. Influent concentrations of 1,000 mg/L benzene were removed to 37–87 mg/L in the outdoor environment. The presence of wetland media (gravel), fertilizer, and warm temperatures were noted in improving treatment.

**TABLE 13.2**
**Casper, Wyoming, Mean Pilot System Areal Rate Constants $k_A$, in m/yr**

| | Aeration | | No Aeration | | Aeration |
|---|---|---|---|---|---|
| Compound | Wetland Mulch | No Mulch | Wetland Mulch | No Mulch | Full Scale |
| Benzene | 518 (237) | 456 (414) | 317 (273) | 226 (164) | 240 |
| BTEX | 356 (218) | 311 (136) | 257 (151) | 244 (151) | 350 |
| TPH | 1,058 (1,141) | 965 (722) | 725 (558) | 579 (506) | 325 |

*Note:* Based on 3TIS; standard deviation in parenthesis.

*Source:* Data from Wallace and Kadlec (2005) *Water Science and Technology* 51(9): 165–171.

### Alkanes

Omari *et al.* (2001) determined removal efficiencies in both vegetated and unvegetated horizontal-flow gravel beds for straight-chain alkanes of 10 to 26 carbon atoms. Removals were typically above 80% for a vegetated top-fed wetland mesocosm, and above 70% for a top-fed unvegetated mesocosm, in eight hours' detention. Removals were somewhat less for the lighter hydrocarbons, up to C15, but did not differ for the heavier alkanes. Omari *et al.* concluded that constructed wetlands planted with *Typha latifolia* were capable of treating oil-polluted water.

Similarly, Hoffman (2003) found no pronounced effects of chain length on alkane removals in willow mesocosms, over the range C10–C32. Willows enhanced the removal of TPH, with the effluent alkane concentrations in vegetated mesocosms about half of that in unvegetated mesocosms. Mechanisms of removal were not elucidated by these studies.

Salmon *et al.* (1998) evaluated removal of total hydrocarbons in FWS mesocosms in France. The bed was planted with *Typha latifolia,* and during the experiment, natural development of *Lemna minor* occurred. Constructed wetlands removed 90% of total hydrocarbons.

Descriptions of other petroleum industry treatment wetland projects may be found in Knight *et al.* (1997). These include the earliest such project at Mandan, North Dakota (Litchfield and Schatz, 1989; Litchfield, 1990; Litchfield, 1993), as well as well-studied projects such as that at Bulwer Island, Australia (Simi and Mitchell, 1999; Simi, 2000).

### Polycyclic Aromatic Hydrocarbons

Polycyclic aromatic hydrocarbons (PAHs) are fused ring aromatic compounds formed during the incomplete combustion of almost any organic material, and are ubiquitous in the environment (Figure 13.1). Some of them are considered as dangerous substances as a function of their toxic and mutagenic or carcinogenic potentialities. The presence of PAHs in contaminated soils and sediments may pose a risk to the environment and human health. PAHs are hydrophobic compounds, whose persistence within ecosystems is chiefly due to their low aqueous solubility. These materials are not volatile, and no loss to the atmosphere is anticipated under wetland conditions.

Creosote consists of a mixture of organic compounds, dominated by PAHs, many of which are individually designated as hazardous wastes. Wood may be treated with creosote to enhance its resistance to decomposition. For instance, railroad ties are typically treated with 128 kg/m$^3$. Many of the industrial sites that are or were used to treat ties are now

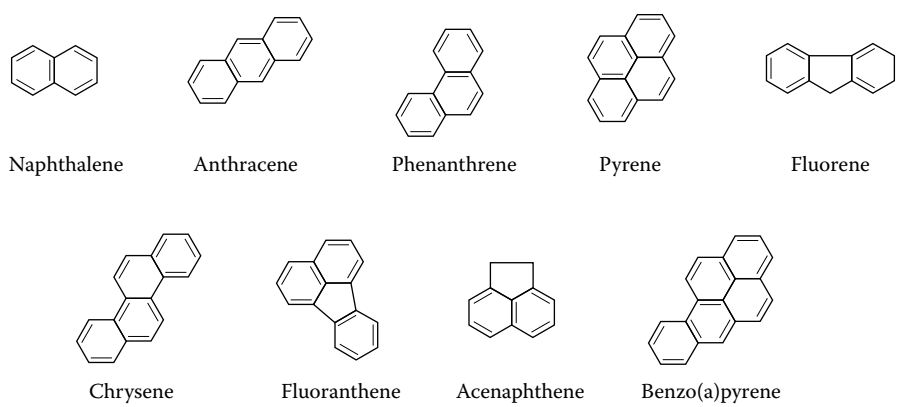



**FIGURE 13.1** Example polynuclear aromatic hydrocarbons.

classified as hazardous waste sites, due to spillage and the associated soil contamination. Conversely, there is little or no risk to aquatic and wetland environments from ties as they are normally placed (Brooks, 2000b), nor from bridge or dock pilings (Brooks, 2000a).

## Biodegradation

Aromatics follow a pattern, with polyaromatic hydrocarbons (PAHs) degrading more slowly than benzene; those with more than three rings may not support microbial growth (Zander, 1980). The aerobic microbial degradation of PAHs having two and three rings is well documented. A large number of bacteria that metabolize PAHs have been isolated (*Alcaligenes denitrificans*, *Rhodococcus* spp., *Pseudomonas* spp., *Mycobacterium* spp.; Giraud *et al*., 2001). A variety of bacteria can degrade certain PAHs completely to $CO_2$ and metabolic intermediates. As the number of fused rings increases, the degree of degradation decreases (Cookson, 1995, as referenced in Walsh, 1999). Degradation of PAHs by anaerobic organisms has not been very successful. However, some degradation has been achieved under denitrifying, sulfate reducing, and methanogenic conditions. Naphthalene and anthracene have been found to be slightly degraded anaerobically under denitrifying conditions (Walsh, 1999).

PAH-degrading bacteria were isolated from contaminated estuarine sediment and salt marsh rhizosphere by enrichment using naphthalene, phenanthrene, or biphenyl as the sole source of carbon and energy (Daane *et al*., 2001). Identification of the isolates assigned them to three main bacterial groups: gram-negative pseudomonads; gram-positive, non-spore-forming nocardioforms; and the gram-positive, spore-forming group, *Paenibacillus* (Table 13.3). This study indicated that the rhizosphere of salt marsh plants contains a diverse population of PAH-degrading bacteria, and the use of plant-associated microorganisms has the potential for bioremediation of contaminated sediments. Contaminated sediment was obtained from Newtown Creek in the New York Harbor, Brooklyn, New York. Chemical analysis showed the dredged sediment to contain 2 to 7 ppm of the PAHs naphthalene, anthracene, and phenanthrene. An induction period of 15 days was observed for pyrene, after which degradation was complete in 10 to 15 days. It should be noted that both the pure culture and microbial slurry experiments were performed under highly oxygenated conditions and that the limited diffusion of oxygen into organic-rich sediments has been found to restrict PAH biodegradation in the natural environment (DeLaune *et al*., 1981).

## Sorption

Peat soils adsorb PAH compounds quite effectively. These organics partition very strongly to carbonaceous soils (IT Corporation, 1997; Pardue and Shin, 2000), and are not readily desorbed (Pardue and Shin, 2000; Shin and Pardue, 2002). It is therefore expected that PAHs would be sorbed and stored in wetland peats. Partition coefficients in the range of $10^3$–$10^4$ L/kg were found for phenanthrene (Shin and Pardue, 2002).

## Plant Uptake

Polycyclic aromatic hydrocarbons are not taken up by wetland plants to any significant extent. Studies at Duluth, Minnesota, have shown very low concentrations in aboveground plant parts, and only trace amounts associated with roots (IT Corporation, 1997). PAHs were found in dogwood (*Cornus stolonifera*) and cattail (*Typha* spp.) shoots at the Duluth site. Willows (*Salix* spp.) and bulrushes (*Scirpus* spp.) have been reported to access tightly bound phenanthrene (Gomez-Hermosillo *et al*., 2000; Gomez and Pardue, 2002). Most of the PAH uptake was to the roots, and was associated

**TABLE 13.3**
**Abilities of Several Isolates to Utilize a Variety of PAHs**

| | % PAH Remaining | | | | |
|---|---|---|---|---|---|
| **Isolate** | **Naphthalene** | **Biphenyl** | **Fluorene** | **Phenanthrene** | **Pyrene** |
| **Pseudomonads** | | | | | |
| PR-N7 | 0 | 85 | 100 | 85 | 90 |
| PR-N9 | 0 | 69 | 78 | 71 | 90 |
| PR-N10 | 40 | 90 | 97 | 89 | 100 |
| PS-P2 | 44 | 51 | 79 | 28 | 100 |
| **Gram-Positive Non-spore Formers** | | | | | |
| PR-N15 | 0 | 84 | 80 | 68 | 100 |
| PR-P3 | 0 | 84 | 42 | 1 | 100 |
| **Spore former** | | | | | |
| PR-P1 | 0 | 0 | 29 | 8 | 100 |

*Source:* Data from Daane *et al*. (2001) *Applied and Environmental Microbiology* 67(6): 2683–2691.

with sorption. Because PAHs are not required for plant metabolism and growth, up take is dependent on concentrations and supplies to the pore water in the root zone.

### Wetland System Studies

Testing of wetlands for PAH removal has presented mixed results. Some tests show minimal removal, whereas others show excellent reductions. Anecdotal monitoring of a cypress swamp (forested peatland) receiving landfill leachate in Florida indicated no PAH removal after about 15 years (Schwartz *et al.*, 1999). However, other landfill leachate constructed wetland studies have shown no detectable PAHs in the effluent, including naphthalene at a Minnesota site; and naphthalene, fluorene, and phenanthrene at a Michigan site (Kadlec, 2003c). Boving (2002) found virtually no retention or removal of ten frequently detected PAHs in a stormwater system comprised of ponds and wetlands. Boving showed that heavier PAHs were present in urban freeway runoff at concentrations in excess of U.S. EPA benchmarks for chronic toxicity. In particular, benzo(*a*)pyrene was present at over 1.0 µg/L.

Subsurface flow constructed wetlands were used to treat coke plant wastewaters at Sollac, Fos sur Mer, France (Jardinier *et al.*, 2001). Removal of 45% was achieved in 11 days' detention. Jardinier *et al.* concluded that reedbeds were a valid method to remove PAHs.

In a German study, naphthalene was removed using hydroponic cultures of *Carex gracilis* and *Juncus effusus* and using sand-bed reactors planted with *Carex gracilis* and *Juncus effusus*, respectively, under batch and flow through conditions (Wand *et al.*, 2002). Concentrations of about 30 mg/L naphthalene were efficiently eliminated over periods of up to six months. Vegetated cultures were found to achieve a better removal rate than systems without vegetation. In the systems investigated, naphthalene was thought to be mainly degraded by bacteria in the rhizosphere.

The behavior of the three-ring PAH phenanthrene was investigated in a VF wetland system in Munich, Germany (Machate *et al.*, 1997), with overall removals of more than 99% at a detention time of 6.5 days. Intermediate degradation involved formation of 1-hydroxy-2-naphthoic acid as a bacterial metabolite, which subsequently was also removed in the wetland. Phenanthrene-degrading bacteria were enumerated, and found to be highest in the inlet zone of the system ($10^4$ per mL) in comparison to the outlet end (fewer than 10 per mL). Feed concentrations of 0.385 mg/L phenanthrene were reduced to less than 0.003 mg/L. The lava rock substrate had only a small partition coefficient, in the range 0.1 to 1.8 L/kg.

An experimental subsurface flow constructed wetland was developed in Curienne (Savoie-France), and operated with a feed of wastewater dosed with fluoranthene (Giraud *et al.*, 2001). Two beds were operated in series, with a total detention time of three days. The inlet fluoranthene concentration was set at 6,660 mg/L, and no PAHs were detected in the wetland outflows. A total of 40 fungal species (24 genera) were isolated and identified from samples (gravel and sediments) from the test wetland and a control wetland. Fluoranthene was degraded efficiently by 33 species whereas only two species were able to remove anthracene by over 70%.

Salmon *et al.* (1998) evaluated removal of total hydrocarbons in FWS mesocosms in France. The bed was planted with *Typha latifolia*, and during the experiment, natural development of *Lemna minor* occurred. Constructed wetlands removed 90% of total hydrocarbons.

## 13.2 CHLORINATED HYDROCARBONS

### Chlorinated Benzenes

Chlorobenzenes were and are used in the production of phenol, aniline, and DDT. Mono-, di-, and trichlorobenzenes are used as solvents (Grayson, 1985). Various benzene hexachloride isomers are used as broad-spectrum insecticides, including Lindane™. Mono-, di-, and trichlorobenzenes are resistant to photo-oxidation, and to both aerobic and anaerobic degradation in purely aquatic environments, with estimated half-lives of up to 6 to 24 months (Howard *et al.*, 1991). However, wetland environments are quite different, because of the close interactions with plants and soils. The major routes for removal of chlorobenzenes from wetland waters are: biological (microbial) degradation, sorption, volatilization, and plant uptake.

### Biodegradation

In general, chlorinated hydrocarbons may be dechlorinated under anaerobic conditions, and the responsible microbial consortia have been identified (van Eekert and Schraa, 2001). Reductive dechlorination of chlorobenzenes occurs via an anaerobic sequential pathway in wetland soils and sediments. Jackson and Pardue (2000) established that the dichlorobenzene (DCB) formed monochlorobenzene (MCB), which subsequently mineralized:

$$\mathrm{DCB} \rightarrow \mathrm{MCB} \rightarrow \mathrm{CO_2} + \mathrm{H_2O} + \mathrm{Cl^-} \qquad (13.1)$$

Their study showed that MCB produced reached half of the initial DCB concentration in 30 days, and was accompanied by the formation of methane. In separate experiments, Jackson and Pardue (2000) recovered 13% of $^{14}$C-MCB as $^{14}CO_2$ in a surface sediment modulated decomposition.

### Sorption

Chlorobenzenes sorb strongly to both organic and inorganic wetland substrates (Pardue *et al.*, 1993). Shin and Pardue (2000; 2002) showed that there are both reversible and irreversible portions of the overall sorption. For several sediment and soil samples, the reversible part ranged $1.88 \leq \log_{10}K_{OC} \leq 2.88$, while the irreversible part ranged $3.75 \leq \log_{10}K_{OC} \leq 5.55$. Thus, partitioning to organics is very strong, and dominated by irreversible binding.

Suspended particulate matter forms a mobile substrate for partitioned chlorobenzenes (Shugui *et al.*, 1994).

The FWS wetland biogeochemical cycle creates and processes very large quantities of total suspended solids (TSS), and thus can play an important role in organic chemical cycling and removal in these systems. The wetland environment is further complicated by the presence of large molecules of humic substances, which comprise a good share of dissolved organic carbon. Organic solutes can partition to these dissolved humic substances, thus creating two soluble forms with different chemical characteristics.

### Volatilization

The chlorobenzenes are volatile, and therefore may be easily lost from water, especially shallow water bodies, such as free water surface wetlands. Of direct interest are the estimates, based on wetland data, of Keefe *et al.* (2004a) for a FWS wetland in Arizona. Values were determined of $K_w = 0.011$ m/hr (100 m/yr) for chlorobenzene, and $K_w = 0.008$ m/hr (70 m/yr) for 1,4 dichlorobenzene. These are relatively high rates, compared to other pollutants treated in FWS wetlands.

### Plant Uptake

Plants are capable of taking up organic chemicals (Trapp and Matthies, 1995), and processing them in a number of ways. They may be carried through the plant into the atmosphere via the transpiration flux, metabolized, or accumulated in plant tissues. For example, Leppich *et al.* (2000) found up to 100 mg/kg of various chlorobenzenes in black willow (*Salix nigra*) bark, and up to 25 mg/kg dw in leaves. The willows were growing in a contaminated swamp site, with large concentrations of di-, tri-, penta-, and hexa-chlorobenzenes. Leppich *et al.* (2000) concluded that partitioning of these organics to the plants formed an important part of the site model.

### Overall Removal Coefficients

Despite the investigations detailed above, there is no reported treatment wetland that has specifically been designed to target chlorobenzenes. Therefore, results from several treatment technologies are examined here to gain some insight as to the anticipated rates of removal to be expected in wetlands.

#### *Wetlands*

Keefe *et al.* (2004a) calculated removal rate coefficients from a FWS dataset to be 135 m/yr for chlorobenzene, and 67 m/yr for 1,4 dichlorobenzene. The wetland was a 1.4-ha free water surface system of depth about two feet. The detention time was 3.9 days. These results correspond to a 67% reduction of 1,4 dichlorobenzene from inlet to outlet. Keefe *et al.* (2004a) attributed the reduction to volatilization, but concluded that theoretical predictions were only good for order-of-magnitude estimation.

Braeckevelt *et al.* (2006) studied monochlorobenzene reduction in a pilot-scale horizontal subsurface flow wetland constructed of local soil materials. Contaminated groundwater containing up to 22 mg/L of monocholorobenzene was added to the system at a hydraulic loading rate of 2.3 cm/d. Chorobenzene reductions of up to 77.1% were observed in the system. Monochlorobenzene reductions were highest in the upper soil layer, possibly due to volatilization, and decreased to 37.1% at the bottom of the wetland bed (0.5 m). Enrichment of $^{13}C$ and low dissolved oxygen concentrations suggest that reductive dehalogenation under anaerobic conditions was the dominant removal mechanism.

## Chlorinated Ethenes

Perchloroethylene (PCE) and trichloroethylene (TCE) are solvents that saw extensive use for metal cleaning and other applications in previous decades. At many locations, these materials found their way into groundwater, creating hazardous waste conditions, because of concern due to their carcinogenic properties. Wetlands provide environments for several mechanisms of removal of chlorinated aliphatic organics, including sorption, volatilization, reductive dechlorination, direct biological oxidation, co-metabolism, and plant uptake and metabolism (Pardue *et al.*, 1993; Parkin, 1999; Pardue *et al.*, 2000).

Kassenga (2002) conducted continuous vertical flow column and wetland microcosm studies to investigate the attenuation potential of chlorinated volatile organic compounds. Calibrated simulations indicated that removal of TCE in constructed wetland columns was controlled by biodegradation whereas both sorption and biodegradation were important natural wetland columns. Kassenga *et al.* (2003) evaluated the removal of *cis*-1,2-dichlorethene (*cis*-1,2-DCE) in upflow wetland mesocosms planted with *Scirpus americanus*. The results confirmed that biodegradation was occurring in the system, and sand, peat, and Bion soil mixture had greater degradation rate than the sand and peat mixture.

Lorah *et al.* (1997; 2002) observed complete removal of TCE and daughter products as contaminated water moved upward through peat to the surface. Reducing conditions were present in the peat, and both methanogenic and iron-reducing zones were identified. This important study provided the impetus to examine the future role of natural and constructed wetlands in the remediation of TCE.

A former TCE recycling plant is now the site of a city park in New Brighton, Minnesota. A TCE plume in a surficial aquifer discharges into the wetland of an adjacent lake (Bankston *et al.*, 2002). Transect studies showed TCE in monitoring wells just upgradient from the wetland, and to a lesser degree in the fringe of the wetland. The anaerobic degradation product of TCE, *cis*-1,2-DCE, was detected in the aquifer and the wetland. It was hypothesized that the indigenous cattail (*Typha latifolia*) assisted in phytoremediation. Accordingly, microcosm studies were performed to determine the fate of the removed TCE. The recovery of $^{14}C$ totaled 94.1%, of which 46.8% was volatilized, most likely as [$^{14}C$] TCE because it was added to the microcosms by surface application. Plant tissue contained 38.2% of the $^{14}C$; 5.3% was present as [$^{14}C$] $CO_2$, and 3.7% was recovered from the

**TABLE 13.4**
**Performance of the Schilling Farm Constructed FWS Wetlands**

| | Annual | Winter | Spring | Summer | Autumn |
|---|---|---|---|---|---|
| HLR (cm/d) | 1.00 | 1.14 | 1.39 | 0.85 | 0.60 |
| T (°C) | 11.9 | 1.0 | 12.4 | 23.6 | 12.7 |
| TCE $C_i$ (μg/L) | 726 | 726 | 726 | 726 | 726 |
| TCE $C_o$ (μg/L) | 37 | 87 | 25 | 18 | 21 |
| cis-1,2-DCE $C_i$ (μg/L) | 472 | 472 | 472 | 472 | 472 |
| cis-1,2-DCE $C_o$ (μg/L) | 44 | 65 | 21 | 46 | 46 |
| VC $C_o$ (μg/L) | 5.8 | 5.4 | 1.6 | 8.5 | 7.6 |

*Note:* Inlet concentrations are average undiluted plume values.

soil and water. This data suggested that natural attenuation is a potential bioremediative strategy for TCE-contaminated wetlands.

Given that natural wetlands contribute to reduction of TCE, the next logical step is the reconstruction of former wetlands that are positioned in the flow path of a contamination plume. Richard *et al.* (2002; 2003) filled and planted a dredged channel that was conveying TCE from a contaminated site to a lake. In this case, an aquatic feature was converted to wetland to aid in treatment. This Minnesota project was completed in 2000.

In other cases, terrestrial landforms may be converted to constructed treatment wetlands to provide reductions in TCE. The Schilling Farm Project at Hillsdale, Michigan, intercepts a TCE plume for treatment in a constructed FWS wetland, built in a former corn field. The wetland system is made up of four treatment cells in parallel (see Figures 4.21 and 4.26). The two large central cells were sited to intercept the plume, and the two small flanking cells were added to ensure the full capture of that plume. Groundwater moves down a slight incline into a 4-m deep capture trench, designed to intercept approximately the top 2 m of the underground flow. This trench is filled with coarse rock to eliminate safety hazards and control rodent problems. The water flows upward and out across the FWS wetland cells, carrying TCE and DCE into the wetlands. During passage through the wetland, TCE undergoes reductive dechlorination to DCE and then to vinyl chloride (VC) in anoxic zones, and these are further degraded to carbon dioxide and water in aerobic zones. There is also volatilization of VC, and to a lesser extent DCE and TCE. Water is collected in rock-filled trenches, approximately 1 m deep, at the downstream end of all four cells. These four outflows are metered, and merged to form a single project outflow for compliance monitoring. Additionally, another independent rock-filled trench is positioned across the entire downstream end of the four cells, which is drained via perforated pipe into the compliance outflow. The purpose of this trench is to capture waters that may pass totally underneath the treatment wetland.

Control of this wetland system is solely by means of setting water levels in each of the four cells, by use of the weir settings in the outlet structures. It is totally passive, with no pumps, and runs year-round in a cold climate. Therefore, water level control must compensate for ice formation. The correct operating strategy is the subject of ongoing investigations. Herbivory and short-circuiting were created by muskrats (*Ondatra zibethicus*), necessitating removal of both the animals and their habitat, by filling trenches with large rock, and by fencing the wetland.

Table 13.4 shows the performance results for the system for 88 of the 95 months of operation, October 1998 through September 2005. Figure 13.2 shows performance for October 2003 through September 2005, respectively. The first seven months were a dormant period for vegetation planted in autumn 1998, which remained sparse for that period. Effluent standards for TCE (limit 150 μg/L) were met in 84 of 88 months after startup. Exceedances for VC (limit 13 μg/L) tend to occur in late summer, when wetland surface flow contributes little to the outflow.

In summary, these projects all utilize the characteristics of wetlands to reduce TCE. Research work continues at the time of this writing at Wright–Patterson Air Force Base in Ohio (Entingh, 2002; Blalock, 2003). The addition of recycle in 2007 has eliminated VC from the outflow.

## 13.3 ORGANIC CHEMICALS

### Explosives

It has been estimated that approximately 100 military bases and explosives manufacturing facilities have soil and/or groundwater contaminated with munitions (Medina *et al.*, 2000). Studies have shown that plants are capable of transforming 2,4,6-trinitrotoluene (TNT) without microbial contribution, but very little accumulation of TNT has been found in plant material. Therefore, plant-enhanced degradation, or phytoremediation, of TNT by aquatic macrophytes has been proposed as a promising groundwater treatment process.

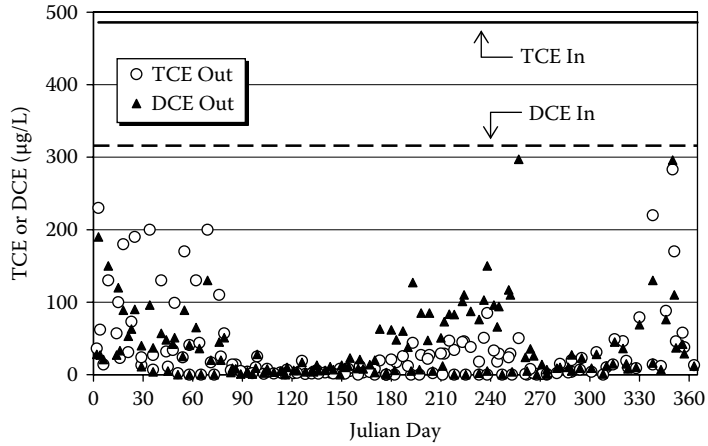



**FIGURE 13.2** Annual pattern of treatment of TCE and DCE in the Schilling Farm wetland system. The inlet concentrations have been adjusted for dilution with clean groundwater entering the flank cells. Winter conditions provide lesser treatment. There is a second period of lesser treatment in late summer and autumn, which is occasioned by very low surface flows and dominance of underground flows.

TNT is a reactive molecule that biotransforms readily under both aerobic and anaerobic conditions to give aminodinitrotoluenes (Brannon and Myers, 1997; Hawari *et al*., 2000; Xiang, 2001; Esteve-Nunez *et al*., 2001). The resulting amines biotransform to give several other products, including azo, azoxy, acetyl, and phenolic derivatives, leaving the aromatic ring intact. Little or no mineralization is encountered during bacterial or wetland treatment. The nonaromatic cyclic nitramine explosives hexahydro-1,3,5-trinitro-1,3,5-triazine (RDX) and octahydro-1,3,5,7-tetranitro-1,3,5,7-tetrazocine (HMX) lack the electronic stability enjoyed by TNT or its transformed products. Therefore, an enzymatic change on one of the N-$NO_2$ or C-H bonds of the cyclic nitramine could lead to a ring cleavage, and subsequent mineralization.

Medina *et al*. (2000) performed a series of batch-scale experiments to assess the engineering kinetics of phytodegradation of TNT under a variety of operational conditions. Parrotfeather (*Myriophyllum aquaticum*) was hydroponically grown in laboratory microcosms. TNT was degraded according to a near-first order model in vegetated microcosms, but not in unvegetated systems (Figure 13.3). Measurements at varying plant densities indicated that rate constants increased with increasing plant abundance. Removal rate constants also increased with increasing temperature from 2 to 34°C, leveled off between 34 and 43°C, and at 54°C, no activity was found. This pattern is also found for enzyme kinetics, in which rates increase until the enzyme denatures. Plants appeared healthy up to 34°C, but wilted at 43°C. Plants incubated at 54°C were dead by the end of the experiment. The modified Arrhenius temperature coefficient was 1.093 for water temperatures between 2 and 30°C. Hughes *et al*. (1997) found that Eurasian water milfoil (*Myriophyllum spicatum*) degraded TNT to aminonitrotoluenes, whereas unvegetated controls did not.

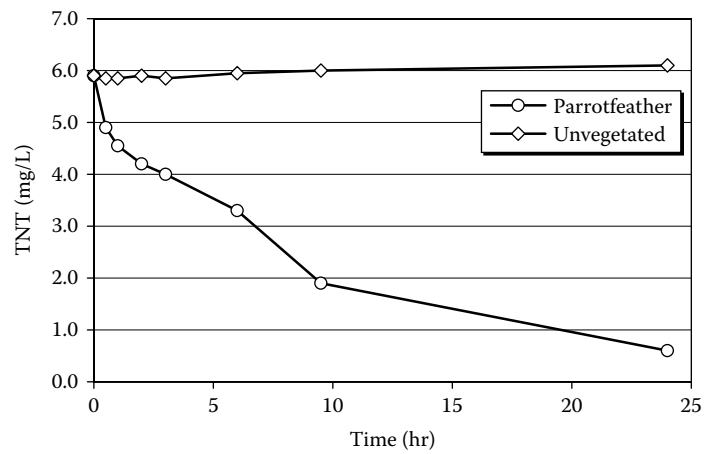



**FIGURE 13.3** Disappearance of TNT in batch microcosms. The vegetated system points represent the mean of five replicates. The half-life of TNT was about seven hours. (Data from Medina *et al*. (2000) *Water Research* 34(10): 2713–2722.)

**TABLE 13.5**
**Batch Kinetics of TNT Reduction for Several Submersed Species with Data Including Two Different Plant Densities**

| Scientific Name | Common Name | $k_A$ (m/yr) |
|---|---|---|
| *Myriophyllum aquaticum* | Parrotfeather | 39 |
| *Myriophyllum spicatum* | Milfoil | 48 |
| *Egeria densa* | Egeria | 30 |
| *Eiodea canadensis* | Elodea | 40 |
| *Vallisneria amerilcana* | Vallisneria | 50 |
| *Potamogeton crispus* | Curly leaf pondweed | 48 |
| *Potamogeton pectinatus* | Sago pondweed | 40 |
| *Heteranthera dubia* | Star-grass | 45 |
| *Eleocharis parvula* | Spikerush | 49 |
| *Chara vulgaris* | Stonewort | 49 |

*Source:* Data from Best *et al.* (1997a) *Screening aquatic and wetland plant species for phytoremediation of explosives-contaminated groundwater from the Iowa Army Ammunition Plant.* Technical Report EL-97-2, U.S. Army Corps of Engineers Waterways Experiment Station: Vicksburg, Mississippi.

Zoh and Horne (2000) also performed a series of batch-scale experiments. Utilizing straw, cattail, and bulrush litter in water, they found no degradation without litter, and first-order behavior in the presence of litter. Sorption and $^{14}C$ studies indicated that removal was due to initial sorption followed by degradation to aminonitrotoluenes. Volumetric rate coefficients ($k_V$) translate to areal rates ($k_A$) on the order of 25 m/yr for TNT, regardless of litter type. A modified Arrhenius temperature coefficient ($\theta$) of 1.17 fit the observed difference in rates at 10 and 20°C.

The ability of ten species of submerged aquatic to phytoremediate explosives-contaminated groundwater was investigated by Best *et al.* (1997a). Phase I of this project provided for laboratory-scale plant screenings to evaluate locally adapted aquatic and wetland species for their differential ability to diminish levels of TNT and RDX. These were evaluated under hydroponic batch conditions. Analysis of the data according to a batch first-order areal model shows remarkable similarity among species (Table 13.5).

Best *et al.* (Best *et al.*, 1999a,b) reported that per unit of mass, uptake of TNT was higher in submerged (*Elodea canadensis, Potamogeton pectinatus, Heteranthera dubia, Myriophyllum aquaticum*) rather than emergent species (*Acorus calamus, Phalaris arundinacea, Scirpus cyperinus*) and biotransformation of TNT had occurred in all plant treatments after a seven-day incubation in 1.6 to 3.4 mg/L TN. TNT declined less with substrates, and least in controls without plants. Mineralization to $CO_2$ was very low, and evolution into C-volatile organics negligible. RDX disappeared less rapidly than TNT from groundwater.

Cattails (*Typha angustifolia*) in FWS mesocosms were used to test treatment of mono-, di-, and trinitrotoluene mixtures at the Volunteer Army Ammunition Plant, Tennessee (Best *et al.*, 2000; 2001). Rate coefficients (Table 13.6) ranged from 16 to 45 m/yr. The potential contribution of photodegradation was determined by shielding nonplanted mesocosms from UV in sunlight. Radiation accounted for 30% of TNT, 60% of DNT, and 10% of NT removal in the absence of plants.

**TABLE 13.6**
**Continuous-Flow Kinetics of Nitrotoluenes Reduction for *Typha angustifolia* Mesocosms**

| Contaminant | Inlet (mg/L) | Outlet (mg/L) | Percent Removal (season) | Areal Rate Constant (m/yr) |
|---|---|---|---|---|
| Trinitrotoluene | 2.73 | 0.30 | 79% | 31 |
| 2,4-dinitrotoluene | 16.66 | 6.00 | 58% | 16 |
| 2,6-dinitrotoluene | 5.17 | 1.40 | 61% | 17 |
| 2 nitrotoluene | 42.64 | 2.20 | — | 45 |

*Source:* Data from Best *et al.* (2000) *Explosives removal from groundwater at the Volunteer Army Ammunition Plant, Tennessee, in small-scale wetland modules*. Means and Hinchee (Eds.), Wetlands and Remediation: An International Conference; Battelle Press: Columbus, Ohio, pp. 365–374; and Best *et al.* (2001) *Water Science and Technology* 44(11–12): 515–521.

*Note:* Season refers to June through October.

However, planted mesocosms had higher removal rates than unshielded unplanted systems (Best *et al*., 2001).

The U.S. Army Corps of Engineers constructed treatment wetlands with the purpose of containing and treating residual TNT (2,4,6-trinitrotoluene) and RDX (hexahydro-1,3,5-trinitro-1,3,5-triazine) contamination at the Iowa Army Ammunition Plant (Thompson *et al*., 2003). Screening studies established the relative capabilities of various plants to reduce explosives found at the site (Best *et al*., 1997a,b). Demonstration studies were then conducted to establish vegetative uptake, fate, and toxicity of TNT and RDX. Results from two years of testing indicated that TNT concentrations were undetectable in surface water or plant tissues. Concentrations of RDX in wetland surface waters were below the U.S. EPA human health advisory level of 2 µg/L and were undetectable in plant tissues during the summer months.

Based upon these results, and other wetland design considerations, testing then proceeded to pilot scale. Behrends *et al*. (1997) reported that a FWS wetland system of two cells, each 9.4 × 24 meters, performed poorly for TNT and RDX removal at Milan, Tennessee. This was attributed to an inability to maintain the vegetation, parrotfeather (*Myriophyllum aquaticum*), in the face of tadpole herbivory and hailstorms. In contrast, a two-cell SSF system (473 $m^2$ total) performed well for TNT, RDX, and HDX. Cold weather limited the gravel-based system's ability to reduce the total nitrobodies below the 50 ppb demonstration goal (Sikora *et al*., 1997).

Analysis indicated that the full-scale system could be economically resized to overcome the winter performance deficit by increasing the retention time from 10–14.5 days (ESTCP, 1999).

Xiang (2001) tested VF wetland mesocosms, in series, the first two planted with *Typha* spp., and the last three with *Scirpus lacustris*. Virtually all of the TNT was removed in the first two cells, at a combined detention time of 2.4 days. In radiolabelled experiments, less than 0.1% of the radioactivity added to the system as TNT was present in the metabolically produced $^{14}CO_2$. Consequently, mineralization was not an important pathway. Small amounts of mono- and dinitrotoluenes were detected, indicating some utilization of a pathway that removed $NO_2$ groups. Sorption experiments showed negligible binding to the media. Planted cascades removed 80% of the TNT, while unplanted removed only 23%. It was noted that photodegradation might have contributed to the removals in the planted tank. Only small amounts of TNT and aminodinitrotoluene were found in plants in the first tank, about 1.0 mg/kg of TNT in shoots, and 0.25 mg/kg (dry weight unless otherwise noted) in roots. Slightly greater amounts of aminodinitrotoluene were found, 1.8 mg/kg in stems and 0.8 mg/kg in roots. Xiang (2001) found enhanced removal with the addition of a carbon source (molasses or sucrose).

## De-icing Compounds

The waste generated by aircraft de-icing operations represents a threat to the environment arising from the high biological oxygen demand of commonly used propylene and ethylene glycols (Castro *et al*., 2004). Other candidates for de-icing include urea, which can also pose a pollution threat to nearby receiving waters (Thoren *et al*., 2003). In temperate climates, pollutant loadings are seasonal, and generally occur from November to April.

Both glycol and urea are biodegradable. Microbial metabolism of glycol within a wetland system utilizes the material as it would other forms of $BOD_5$ (see Chapters 8 and 13). Degradation of urea follows patterns established for other species of nitrogenous compounds (see Chapter 9). However, pollutant loads occur at cold temperatures, which minimize some biological processes, notably those for nitrogen reduction. The threat of bird–aircraft strikes indicates the use of SSF wetlands in many cases.

The applicability of treatment wetlands for reducing the concentration of ethylene glycol is being tested or implemented at several airports, including Pearson Airport in Toronto, Edmonton International Airport in Alberta, Heathrow Airport in London, and Westover Air Reserve Base in Massachusetts. Constructed wetlands are also in use for nitrogen de-icer control at Kalmar Airport, southeast Sweden. Pilot wetland testing has occurred at the Airborne Express airport in Wilmington, Ohio (Naval Facilities Engineering Service Center and Wetland Solutions, Inc., 2004b); Zurich, Switzerland (Richter *et al*., 2004); and Schonefeld Airport in Berlin, Germany (Abydoz Environmental, Inc., 2005).

In 2000, the Greater Toronto Airports Authority (GTAA) constructed the first full-scale vertical flow treatment wetland in Canada for reduction of ethylene glycol in stormwater (Flindall *et al*., 2001). The facility consists of a two-cell treatment wetland preceded by a sedimentation forebay. The wetland facility collects water discharged from a drainage area of approximately 382 hectares, comprised of terminal aprons, taxiways, and runways where aircraft receive glycol de-icing compounds. The facility provides 24 to 48 hours of retention to attenuate storm flows and provide removal of suspended sediment. Cell 1 of the wetland is a 0.42-ha VF system, and Cell 2 is a 1.38-ha FWS system, and is primarily intended for storage of runoff rather than treatment. Both treatment cells were planted with *Phragmites australis*. A key component of the facility is the sediment forebay, which provides temporary storage of flows entering the facility. The forebay allows removal of some of the sediment load in the runoff, which is necessary to prevent blockage of the vertical flow reed bed. Additionally, forebay storage allows intermittent dosing of the vertical flow reed bed, which is an important design feature of the vertical flow system.

Urea (chemical formula: $NH_2$-CO-$NH_2$) is used as a de-icing agent at Kalmar Airport, southeast Sweden (Thoren *et al*., 2003). To reduce stream transport of nitrogen from airport, agricultural, and other diffuse sources to the Baltic Sea, a FWS wetland system was constructed in 1996. The 18-ha wetland consists of four ponds (1.5 m deep), which are dominated (75%) by the submerged plant *Elodea canadensis*, separated by shallower water (0.5 m deep) dominated by common reed (*Phragmites australis*). The theoretical water retention times for the wetland are 5.3 and 2.4 days during

average and high water discharges, respectively. Annual wetland retention of TN varied in the range of 6 to 36% during 1998 to 2001. During airport de-icing, January through March 2001, 660 kg urea-N out of 2,600 kg applied urea-N reached the wetland, and approximately 40% of the incoming urea-N was eliminated in the wetland system at air temperatures around 0°C.

The Edmonton Airport in Canada has a subsurface wetlands treatment facility to treat stormwater contaminated with ethylene and propylene glycol from the de-icing activities that occur at the airport (Higgins and Maclean, 2002; Edmonton Airports, 2003). The HSSF facility is comprised of six trains of two cells each. The 4.5-ha wetlands started operating in 2001. Cells are vegetated with cattails (*Typha* spp.), with the exception of one cell being planted with reed canary grass (*Phalaris arundinacea*). The new system integrates the stormwater collection system serving the apron area by channeling the fluids into a 300,000 cubic meter detention facility, prior to the wetlands. Treated water flows out of the area to surface receiving waters. The cost to construct the wetlands was CDN $2 million.

A 0.24-ha HSSF wetland system was installed at the Westover Air Reserve Base in Chicopee, Massachusetts to demonstrate the treatment of stormwaters from on-site de-icing operations (Karrh *et al.*, 2002; ESTCP, 2004; Naval Facilities Engineering Service Center and Wetland Solutions Inc., 2004a). The apparent hydraulic residence time in the bed was estimated at 1.3 to 1.9 days. Event mean flow through the wetland was 278 $m^3$/d for an average hydraulic loading rate of 11.4 cm/d. Peak inflow BOD concentrations ranged from 974 to 15,098 mg/L during ten de-icing events in 2002, and these were reduced by more than 50% in 5 of the 10 events. Event flow-weighted mean concentrations for BOD were 1,183 and 937 mg/L at the wetland inflow and outflow for an estimated mass reduction of 21%. Overall averages for the season are given in Table 13.7. The estimated mass removal rate was 286 kg/ha·d. The system was overloaded by a factor of four due to extreme conditions during the winter of testing.

London's Heathrow airport began pilot testing of wetlands for glycol removal in 1994 (Chong *et al.*, 1999; Revitt *et al.*, 2001). Surface and subsurface flow systems and floating rafts were operated, with detention times of less than one day. Modest reductions (approximately 30%) of various glycols were achieved (Worrall *et al.*, 2002). Subsequently, a 1.2-ha floating raft wetland and a 2.08-ha HSSF were built, and planted with *Phragmites*. The design detention times were both just over one day (Revitt *et al.*, 2001; Richter *et al.*, 2003). Evaluation of performance is in progress, with a five-year data acquisition period (Richter *et al.*, 2003; Richter *et al.*, 2004). The system responds to pulse events by delaying and reducing the incoming COD peak (Figure 13.4).

The Buffalo–Niagara International Airport (BNIA) has studied the use of aerated vertical flow wetlands for glycol removal. A pilot-scale system was constructed to determine necessary design parameters (Higgins *et al.*, 2006a). A volumetric rate coefficient of $k_v = 3.3\ d^{-1}$ was observed for reduction of glycol-produced COD. This rate coefficient was developed based on a hydraulic flow model of two tanks in series (2TIS) and an operating temperature of 4°C.

## Phenols

Phenol is a compound of considerable industrial importance and a frequent pollutant in industrial effluents. It also has been studied for over four decades in the wetland treatment context. Early work at the Max Planck Institute in Krefeld, Germany, focused on SSF microcosms with bulrushes (*Schoenoplectus* (*Scirpus*) *lacustris*) and resulted in a series of publications on removals and effects on plant physiology (for example, Seidel, 1966). That data showed degradation rates of 5 to 20 g/$m^3$·d, and in some instances, an induction period of one to three days

**TABLE 13.7**
**Summary of Results from the 2002–2003 De-icing Season at Westover Air Reserve Base**

| Constituent | Inlet | Outlet | Percent Reduction |
|---|---|---|---|
| BOD (mg/L) | 2,226 | 2,094 | 6% |
| COD (mg/L) | 1,883 | 1,335 | 29% |
| MeBT (mg/L) | 0.68 | 0.72 | –6% |
| DO (%) | 52.2 | 47.7 | –9% |
| pH (S.U.) | 7.58 | 9.54 | — |
| Redox (mV) | 391 | 172 | –56% |
| Turbidity (NTU) | 5.22 | 4.61 | 12% |

*Note:* MeBT is methyl-1H-benzotriazole, an additive to the de-icing fluid.

*Source.* Data from ESTCP (2004) *Enhanced biological attenuation of aircraft deicing fluid runoff using constructed wetlands.* Environmental Safety Technology Certification Program (ESTCP) Cost and Performance Report CP-0007, U.S. Department of Defense: Arlington, Virginia.

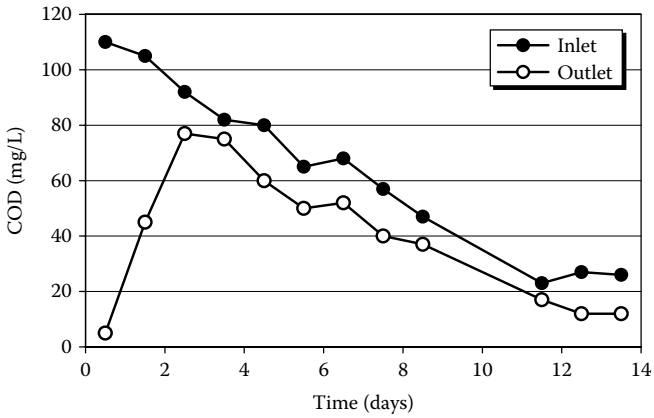



**FIGURE 13.4** Time series of COD into and out of the Heathrow reedbeds. (Data from Richter *et al.* (2004) *Treatment performance of Heathrow constructed wetlands*. Liénard and Burnett (Eds.). Proceedings of the 9th International Conference on Wetland Systems for Water Pollution Control, 26–30 September 2004; Association Scientifique et Technique pour l'Eau et l'Environnement (ASTEE), Cemagref, and IWA: Avignon, France, pp. 125–132.)

before reduction commenced. The Seidel data may be modeled by a zero-order reaction. The zero-order rate constant for the Krefeld data was $k_0 = 81 \pm 18$ mg/d in summer, and 34 ± 10 mg/d in winter. The corresponding modified Arrhenius temperature coefficient is $\theta = 1.083$.

Wolverton and McDonald (1981) evaluated removal of phenol in experimental mesocosms filled with 16 cm of 2.5–7.5 cm diameter railroad rocks covered with 5 cm deep of 0.25–1.3 cm diameter pea gravel. The mesocosms were planted with *Phragmites australis* and *Typha latifolia*. After 24 hr the concentration of phenol dropped down from 104 to 7 mg/L in *Phragmites* unit whereas in *Typha* unit the phenol concentration dropped down from 101 to 17 mg/L during this time. After 48 hours the respective phenol concentrations were 4 and 7 mg/L. Phenol concentration in unplanted control unit dropped from 100 to 40 and 28 mg/L after 24 and 48 hours, respectively.

Microcosm research on FWS wetlands with cattails produced similar results (Kadlec and Srinivasan, 1995). The average value of the zero-order rate constant was 76 ± 33 mg/d (loss = 0.55 g/m$^2$·d) for vegetated microcosms, and 108 ± 26 mg/d (loss = 0.78 g/m$^2$·d) for unvegetated microcosms. The sizes of these microcosms were comparable to those at Krefeld. There was little selective loss of phenol from microcosms containing only water and the phenol dose. This study showed no effect of water depth, but a slight effect of soil type, with more organic content promoting phenol removal. There were higher evaporative losses in summer, with phenol being lost with water at the bulk water concentration. There was not significant evaporative concentration or selective stripping of phenol.

Polprasert *et al.* (1994; 1996) operated both FWS microcosms and a FWS pilot wetland in Bangkok, Thailand, for phenol reduction at high strength. Nearly complete removal of phenol was measured for loading rates up to 300 kg/ha·d. Some phenol was volatilized; some was found in the roots of the cattails. Removal followed a first-order model, with rate constants on the order of 100 m/yr. About one third of the removal was reported due to volatilization, and the remaining two thirds due to biodegradation, plant uptake, and sorption.

Abira *et al.* (2005) studied duplicate pilot scale subsurface flow wetlands for phenol reduction, with two cells each of unvegetated gravel, *Cyperus papyrus, Phragmites mauritanius,* and *Typha domingensis*. Removals in the 15-month study ranged from 50–75%, with higher reductions for ten days detention compared to five days. Unvegetated cells performed nearly as well as vegetated cells. The paper mill wastewater had high COD (300–600 mg/L), but only about 1 to 3 mg/L of phenol.

The Jinling Petrochemical Company, Beijing, China, reported reductions in several parameters, including phenol, in water hyacinth wetlands (Tang and Lu, 1993). Two wetlands, 25 m × 150 m × 1 m deep, received wastewater at detention times ranging from 2 to 12 days. One channel was 80% covered by *Eichhornia crassipes*, the other was unvegetated. The calculated rate constants for phenol removal were 150 m/yr for the vegetated system, and 76 m/yr for the unvegetated system.

Dong and Lin (1994a) report data and models for a research facility polishing secondarily treated petrochemical wastewater in six wetlands and six ponds comprising a 1.5-ha test unit at Yanshan Petroleum, in Niukouyu, China. This research facility tested a variety of plants and soil substrates over the seasons. Phenol was reduced by 28–37%, with higher values in warm months. Dong and Lin (1994b) then described a full-scale facility, built in 1990, consisting of 50 ha of wetlands and ponds treating 100,000 m$^3$/d. Average inflow and outflow phenol concentrations were 27 mg/L and 10 mg/L, respectively, with a mean removal efficiency of 63%. The full-scale facility had a phenol rate constant removal rate coefficient of 100 m/yr.

The treatment system at Amoco Mandan, North Dakota, is comprised of an API separator, lagoon, and a series-parallel set of FWS wetlands. Approximately 5,700 m$^3$/d of process water is directed to an API separator for primary treatment, and then passed through the oxidation lagoon for secondary treatment. Process wastewater and stormwater are then directed through an 0.8-km earthen canal to 6 of the 11 cascading ponds (16.6 ha) before eventual discharge from Dam 4 to the river. The remaining five ponds (19.1 ha) are reserved for wildlife. (Litchfield and Schatz, 1989; Litchfield, 1990; Litchfield, 1993). Removal of phenol amounted to 94% with an average outflow concentration of 5 mg/L.

Microcosms and pilot scales display removal rates that are greater than those observed in some full-scale field situations. For instance, the Mandan, North Dakota, wetlands do not reduce phenols to zero, as they should, because of a detention time on the order of months (Litchfield and Schatz, 1989; Litchfield, 1990; Litchfield, 1993). Further, the loss rates for the Jinling project are also much smaller than would be predicted from the microcosm data (Tang and Lu, 1993).

Both of these systems receive phenol at concentrations much lower, by a factor of more than 100, than the lab studies. It is clear that microcosms are not adequate for determination of design parameters for phenol. For instance, the areal mixing efficiency of the wetland may be an important determinant of trace amounts of phenol in the effluents from pilot and full-scale wetlands.

The Listowel studies included the monitoring of phenols in five wetlands for four years (Herskowitz, 1986). These substances were present in the aeration cell and in the lagoon that provided the wetland influent, at concentrations on the order of 10 µg/L. Winter levels were higher than summer, and the wetlands did not provide removal at these low concentration levels.

### Surfactants

Linear alkylbenzene sulfonate (LAS) is a widely used synthetic surfactant for domestic detergents. This group of substances is amenable to phytoremediation using both terrestrial and wetland plants (Schwitzguebel *et al.*, 2001). Wetland plant detritus is an important source of carbon and energy and offers a large surface area for sorption of dissolved organics. To examine the biodegradation of surfactants by detrital microorganisms, Federle and Ventullo (1990) obtained submerged oak leaves from a laundromat wastewater pond, which were incubated in water amended with $^{14}$C-labeled LAS. Sorption and evolution of $^{14}CO_2$ were followed with time. LAS that was sorbed to the detritus was mineralized without a lag, with a half-life of 12.6 days. This study showed that detritus represents a significant site of surfactant removal in detritus-rich systems.

Barber *et al.* (2001) performed synoptic sampling of several existing treatment wetlands, including analysis for LAS (Table 13.8). Results were mixed, with removal occurring in some, but not all, systems.

Adsorption and degradation were found to both be operative in the wetland and in companion laboratory research (Inaba, 1992). LAS was effectively reduced in a FWS *Typha–Phragmites* wetland. The 474-m$^2$ system treated gray water from a 100-inhabitant community. The calculated plug flow volumetric removal coefficients ($k_V$) for the removal of these surfactants were in the range of 30 to 80 m/yr, depending upon chain length and isomeric form, with greater removal rates for the longer chain lengths. Seasonal dependence was strong, with a temperature factor of $\theta = 1.087 \pm 0.005$ averaged over all forms of the detergent. The inlet concentration totaled about 5 mg/L, and was reduced by over 90% in summer and over 40% in winter at an average hydraulic loading of approximately 5 cm/d. The surfactant partitioned strongly to wetland TSS, with a partition coefficient of 30 L/kg for chain length 11, isomer 5. Values for other forms of the surfactant were higher, ranging up to 520 L/kg for chain length 13, isomer 2. Sorption was not diminished at cold temperatures. Therefore, the sorption capacity and TSS trapping efficiency of the wetland partially offset the diminished degradation during winter.

LAS was investigated in SSF pilot subsurface flow constructed wetlands comprised of eight beds of 55 m$^2$ with different aspect ratios, two water depths and two medium sizes (Huang *et al.*, 2004). The *Phragmites* wetlands treated urban wastewater. Under the most favorable conditions, LAS was biodegraded up to 71%. Shallower water depth (27 cm) was considerably more efficient than deeper (47 cm) for LAS removal. Water temperature has a significant positive effect on the LAS reduction. Biodegradation of LAS was found to occur under both sulfate- and nitrate-reducing conditions. C13 LAS homologues were generally removed in higher extent than the shorter alkyl chain counterparts, thus reinforcing the findings of Inaba (1992).

Del Bubba *et al.* (2000) used a pilot HSSF system planted with *Phragmites australis* to study LAS removal in Florence, Italy. With HLR of 3.7 cm/d the inflow LAS concentrations in unfiltered samples of 278 mg/L and 18.8 mg/L were reduced to 0.18 mg/L and 0.06 mg/L. This resulted in respective removal efficiencies of 99.9% and 99.7%. The authors also pointed out that the biodegradation of LAS homologues increased with increasing carbon atom number: C14 > C13 > C12 > C11.

Kantawanichkul and Wara-Aswapati (2005) reported that LAS inflow concentration of 40 mg/L was reduced by 89% in an experimental HSSF constructed wetland planted with *Typha angustifolia* in Thailand. The highest LAS removal

**TABLE 13.8**
**Results of Synoptic Sampling of Linear Alkylbenzene Sulfonate (LAS) at Several Treatment Wetland Sites**

| Site | Wetland Type | Source Water | Area (ha) | $\tau_n$ (days) | $C_i$ (µg/L) | $C_o$ (µg/L) |
|---|---|---|---|---|---|---|
| Hemet | FWS | Municipal | 10 | 14 | 6.2 | 4.0 |
| Arcata treatment | FWS | Lagoon | 6 | 1 | 0.9 | < 0.1 |
| Arcata enhancement | FWS | Wetland | 12 | 20 | <0.1 | 1.9 |
| Halsey | HSSF | Pulp mill | 0.13 | 2–10 | 10.0 | 11.2 |
| Halsey | FWS | Pulp mill | 0.13 | 2–10 | 10.0 | 3.4 |
| Halsey | FWS | Pulp mill | 0.13 | 2–10 | 10.0 | 7.9 |
| Corvallis | FWS | Dairy | 0.02 | 7 | 92.6 | 1.9 |

*Source:* Data from Barber *et al.* (2001) *Environmental Science and Technology* 35(24): 4805–4816.

was achieved at HLR 15 cm/d while at HLR of 10 cm/d and 20 cm/d the respective removals amounted to 87% and 77%.

Billore *et al.* (2002) operated a 300-$m^2$ SSF wetland, vegetated with *Phragmites karka*, and studied LAS reduction over a 12-month period. Three days' nominal detention provided 43, 61, 76, and 88% reduction of C10, C11, C12, and C13 LAS, respectively, again reinforcing the findings of Inaba (1992).

LAS was monitored for one year at three SSF *Phragmites* reedbed sites in the United Kingdom (Thomas *et al.*, 2003). Hydraulic loadings were 133 cm/d at Brynsiencyn, 14.5 cm/d at Clutton, and 77 cm/d at Rosset; and had average reductions of 57% at Clutton and 55% at the other two sites. The longer alkyl chain homologues were removed to a greater extent.

### Miscellaneous Hydrocarbons

The estrogenic hormones 17-estradiol (E2) and 17-ethinyl estradiol (EE2) have been detected in municipal wastewater effluent and surface waters at concentrations sufficient to cause feminization of male fish (Gray and Sedlak, 2004). These hormones were added to an engineered treatment wetland, and 36% of E2 and 41% of EE2 were removed in 3.5 days' nominal retention time. The observed attenuation was attributed to sorption, followed by biotransformation. Sorption was indicated by the retardation of the hormones relative to a conservative (lithium) tracer. Biotransformation was indicated by elevated concentrations of the E2 metabolite, estrone.

A full-scale FWS treatment wetland has been in operation since 1995 at Quimigal S.A. in Estarreja, Portugal, for reducing organics in the effluent from a sulfanilic acid plant (Haberl *et al.*, 2003) (Figure 13.5).

This wastewater contains nitrobenzene, aniline, and sulfanilic acid. Four vertical flow wetland beds totaling one hectare are operated at a hydraulic loading rate of 2.4 cm/d. Removals were 6 g/$m^2$·d (99%) for aniline, 2 g/$m^2$·d (98%) for nitrobenzene, and 4 g/$m^2$·d (99%) for aniline. *Phragmites* has survived well in this industrial wastewater (Figure 13.5).

*Lemna minor* batch mesocosms displayed half-lives of one to three days for a variety of chlorinated phenols (Tront, 2004; Tront *et al.*, 2004). Rapid sorption (less than five minutes) was observed, followed by first-order removal. There was no clear pattern associated with chemical structure. The microcosm research at Krefeld, Germany, found a surprisingly large amount of pentachlorophenol was absorbed in wetland systems (Seidel, 1976). This result was somewhat unexpected, as pentachlorophenol is used as a wood preservative and does not biodegrade readily.

Polar organics are also degraded in constructed wetlands. Oil sands tailings water contains toxic constituents naphthenic and naphthoic acids (Wood *et al.*, 1995). Cattail mesocosms were used to examine the removal of naphthoic acid in a Kadlec and Srinivasan (1995) project. A lag time was observed before removal started, as observed in the Krefeld experiments with phenol (Seidel, 1976). However, that lag phase was followed by a steady but slow decline of this hydrophilic hydrocarbon. The zero-order rate constant is smaller than for phenol, $k_0 = 0.044$ g/$m^2$·d.

Three polar organic solvents, acetone, tetrahydrofuran (THF), and 1-butanol, were tested in experimental microcosm subsurface constructed vegetated with *Juncus effusus*, *Carex lurida*, *Iris pseudacorus*, *Pondeteria cordata*, and unplanted controls (Grove and Stein, 2005). Ninety percent removal of 1-butanol typically took less than three days, acetone required from 5 to 14, and THF required at least ten days, but was frequently not achieved in 14 days. Removal was typically better in planted replicates, and was slower in winter. Herrera-Melian *et al.* (2004) found essentially complete removal of methanol and formaldehyde in planted and unplanted gravel mesocosms in less than one day. Small SSF wetlands were examined for potential use in reduction of benzoic acid (Zachritz *et al.*, 1996). Planted (*Schoenoplectus* (*Scirpus*) *validus*) and unplanted rock bed mesocosms were fed 5 to 50 mg/L benzoic acid at hydraulic loading rates of 4.9 and 9.6 cm/d. Removals were 30 to 96% in single- and double-tank systems.

Phthalate acid esters (PAEs) are important at many industrial products, and occupy 80–85% of the global plasticizer market (Eljertsson *et al.*, 1997). Studies have shown that PAE contamination in waters, sediments, soils, and biota (Staple *et al.*, 1997) may be toxic to mammals, and aquatic organisms (Adams *et al.*, 1995; Foster *et al.*, 2000; Patyna and Cooper, 2000). In addition, the high bioaccumulation rate and mutagenic action posed a potential threat to human health (Zhou *et al.*, 2005b). Di-*n*-butyl phthalate (DBP) is the most frequently identified PAE in diverse environmnetal symplex, and it has been listed as a priority polutant by the U.S. EPA.

Zhou *et al.* (2005b) evaluated the fate of di-*n*-butyl phthalate in a downflow/upflow constructed wetland in China. The results indicated that on the tenth day, 35% of the DBP in the surface soil and 62% of the DBP in the subsurface soil remained. After 30 days, 95.7% DBP was degraded in the surface soil and 64.2% DBP was degraded in the subsurface soil. The degradation fit a first-order model and the kinetic equations gave rate constants for surface and subsurface soils of 0.50 and 0.17 $d^{-1}$ and the half-life of DBP in the respective soils was 1.4 and 4.0 days.

## 13.4 PESTICIDES

The list of pesticides is very long, and therefore wetland fate and transport studies lag far behind their introduction and use. A distinction may be drawn between the persistent chemicals used prior to the 1950s and the more degradable substances used since that time. Possible retention factors are adsorption to soil particles and organic matter, sedimentation of particles, photodegradation, plant uptake, and biodegradation. The U.S. Department of Agriculture, Agricultural Research Service publishes a compendium of chemical and physical properties of 334 widely used pesticides. Information included in the database focuses on 16 of the most important properties that affect pesticide transport and degradation characteristics (U.S. Department of Agriculture, 2001).



**FIGURE 13.5** Reed bed treating sulfanilic acid at Quimigal S.A. in Estarreja, Portugal.

Many of the "old" pesticides, such as DDT, are very persistent in the environment. These substances partition strongly to particulate matter. It is doubtful that wetlands can provide any very effective mechanism for degradation, but wetlands can act as a trap for the particulates that carry most of the load. Very little data is available for this class of compounds, but Winter (as referenced in Gilges, 1991) reports some success treating PCBs and Lindane™ in SSF systems.

Studies at the Des Plaines, Illinois, treatment wetlands indicated the presence of some of the old "hard" pesticides. The river and wetland sediment samples all contained quantifiable levels of PCBs and some of them also had one or more of the chlorinated pesticides or metabolites DDT, DDE, and dieldrin. All identified species were found in the low micrograms-per-kilogram (parts per billion) range. PCBs were present at the 20–25 µg/kg level in river-borne sediments throughout most of the year, and were found in the sediments of wetland EW3 at less than 10 µg/kg. DDT, together with its decomposition product DDE, were present at approximately 2 µg/kg. Wetland cell EW3 sediments contained similar low levels of DDT and DDE. Dieldrin ranged from 0.1 to 3.0 µg/kg in river sediments and 0.5 µg/kg in wetland sediments.

Modern pesticides degrade more readily, and wetlands have been found to generally reduce levels of many of these compounds. Some of these may also volatilize (Suntio *et al*., 1988). However, the many types of wetlands and modes of operation (continuous flow, event flow, batch) compound with a very long list of pesticides to create very numerous combinations. Examples of studies that partially define treatment expectations are shown in Table 13.9. Perhaps the most commonly used agricultural herbicide is atrazine, and it has therefore been the subject of many treatment wetland studies.

### Atrazine

Atrazine (2-chloro-4-(ethylamino)-6-(isopropylamino)-s-triazine) is a triazine herbicide used to control broadleaf weeds in corn, sorghum, sugarcane, turfgrass, fruits, and vegetables (Kao *et al*., 2001b). Agricultural practices produce runoff with atrazine at concentrations which sometimes peak in excess of the federal drinking water standard (Alvord and Kadlec, 1995; Anderson *et al*., 2002). The atrazine–wetland interaction is very complex, including removal from the area by convection in the water, loss of chemical identity by hydrolysis to hydroxytriazine and dealkylation, and sorption on wetland sediments and litter. Atrazine degrades to trihydroxytriazine in ultraviolet light (Hequet *et al*., 2001), but this process is not rapid in natural waters (Konstantinou *et al*., 2001). Degradation half-lives in various natural waters range from 26 to 65 days (Table 13.10). In the absence of light, aqueous degradation rates are much slower, half-lives greater than 1,000 days, even with added carbon sources (Chung *et al*., 1996; Ghosh and Philip, 2003).

The studies of Seybold *et al.* (2001) illustrate that wetland soils and water, by themselves, are not efficient reducers of atrazine. Soil samples for the study were collected from a tidal freshwater wetland site along the James River in Prince George County, Virginia. The soil was a Levy silt loam with an organic matter content of 14.1%. The half-life was 38 days for atrazine in the anaerobic sediments, and 86 days in the overlying water. Metabolites detected in the aqueous phase above the soil were hydroxyatrazine, deethylatrazine, and deisopropylatrazine. The hydroxyatrazine accumulated in the sediments, and was resistant to further degradation.

Anderson *et al.* (2002) studied the mineralization of atrazine in sediments from a constructed wetland in central Ohio. Atrazine mineralization potential was measured by $^{14}CO_2$ evolution from [U-ring-14C]-atrazine in biometers. The constructed wetland showed 70 to 80% mineralization of atrazine within one month (Figure 13.6). The highest levels of mineralization were localized to the top 5 cm of the wetland sediment.

Weaver *et al.* (2004) pointed out that atrazine dissipated in saturated and flooded soils with a half-time of approximately 23 days, but only 10% of atrazine was mineralized to $CO_2$. The major route of atrazine dissipation was incorporation of atrazine residues into methanol-nonextractable (soil-bound) components, with minimal extractable metabolite accumulation. A mixed-mode extractant (potassium phosphate:aceto-nitrile) recovered greater amounts of $^{14}C$-residues from atrazine-treated soils, suggesting that hydrolysis of atrazine to hydroxylated metabolites was a major component of the bound residues.

Atrazine transport, sorption, and identity loss were studied at the Des Plaines River constructed wetland site, and in accompanying laboratory work. Sorption was effective for soils and sediments, but the more organic materials, such as litter, showed a stronger affinity for atrazine than the mineral base soils of the wetland cells at Des Plaines, Illinois (Alvord and Kadlec, 1995; 1996). Atrazine was found to degrade on those sediments according to a first-order rate law, with a half-life of 40 to 90 days. However, degradation was faster on cattail litter, precluding a rate measurement. The estimated half-life on cattail detritus was on the order of five days.

**TABLE 13.9**
**Removal of Pesticides in Treatment Wetlands**

| Pesticide | Wetland Type | Size | Flow Regime | Vegetation | HRT (days) | Reduction (%) | Reference |
|---|---|---|---|---|---|---|---|
| Aldrin | FWS | Field | Continuous/ Event | Freshwater marsh | 2.1 | 86 | Lopez-Flores *et al.* (2003) |
| Azinphos-methyl | FWS | Field | Event | *Typha capensis* | 0. 5–2.9 | 100 | Schulz and Peall (2001) |
| Chlorothalonil | FWS | Mesocosm | Batch | *Scirpus cyperinus* | 0.5 | 98 | Sherrard *et al.* (2004) |
| Chlorpyrifos | FWS | Field | Event | *Typha capensis* | 0.5–2.9 | 100 | Schulz and Peall (2000) |
| Chlorpyrifos | FWS | Mesocosm | Batch | *Elodea densa* | 2.5 | 40 | Karen *et al.* (1998) |
| Chlorpyrifos | FWS | Mesocosm | Batch | *Scirpus cyperinus* | 3 | 98 | Sherrard *et al.* (2004) |
| Chlorpyrifos | FWS | Field | Dosed | — | 15–30 | 95 | Darby (1995) |
| Cinosulfuron | FWS | Field | Batch | *Oryza sativa* | 21 | 81 | Ferrero *et al.* (2001) |
| DDT | FWS | Field | Continuous/ Event | Freshwater marsh | 2.1 | 100 | Lopez-Flores *et al.* (2003) |
| Diazinon | FWS | Field | Continuous | *Scirpus* spp. | 3.2 | 22 | City of Phoenix unpublished data |
| Diazinon | FWS | Field | Continuous | *Scirpus* spp. | 6.3 | 12 | City of Phoenix unpublished data |
| Diazinon | FWS | Field | Continuous | *Scirpus* spp. | 6.9 | 60 | City of Phoenix unpublished data |
| Diazinon | FWS | Field | Continuous | *Scirpus* spp. | 6.8 | 28 | City of Phoenix unpublished data |
| Endosulfan | FWS | Field | Event | *Typha capensis* | 0.5–2.9 | 100 | Schulz and Peall (2000) |
| Fenpropimorph | FWS | Field | Event | *Sparganium erectum* | 0.25 | 23 | Braskerud and Haarstad (2003) |
| Lindane | FWS | Field | Continuous/ Event | Freshwater marsh | 2.1 | –19 | Lopez-Flores *et al.* (2003) |
| Linuron | FWS | Field | Event | *Sparganium erectum* | 0.25 | 17 | Braskerud and Haarstad (2003) |
| Metalaxyl | FWS | Field | Event | *Sparganium erectum* | 0.25 | 15 | Braskerud and Haarstad (2003) |
| Metamitron | FWS | Field | Event | *Sparganium erectum* | 0.25 | 33 | Braskerud and Haarstad (2003) |
| Methyl Parathion | FWS | Mesocosm | Batch | *Juncus effusus* | 4 | 100 | Schulz *et al.* (2003c) |
| Metolachlor | SSF | Mesocosm | Dosed | *Scirpus validus* | 2–19 | 34–97 | George *et al.* (2003) Stearman *et al.* (2004) |
| Metolachlor | Pond | Mesocosm | Dosed | *Scirpus validus* | 12–20 | 50 | Mazanti *et al.* (2003) |
| Metribuzin | FWS | Field | Event | *Sparganium erectum* | 0.25 | 30 | Braskerud and Haarstad (2003) |
| Permethrin | FWS | Field | Continuous/ Event | Freshwater marsh | 2.1 | 4 | Lopez-Flores *et al.* (2003) |
| Propachlor | FWS | Field | Event | *Sparganium erectum* | 0.25 | 41 | Braskerud and Haarstad (2003) |
| Propiconazole | FWS | Field | Event | *Sparganium erectum* | 0.25 | 19 | Braskerud and Haarstad (2003) |
| Simazine | SSF | Mesocosm | Dosed | *Scirpus validus* | 2–19 | 51–96 | George *et al.* (2003) Stearman *et al.* (2004) |
| Simazine | FWS | Field | Continuous/ Event | Freshwater marsh | 2.1 | 100 | Lopez-Flores *et al.* (2003) |
| Simazine | FWS | Microcosm | Batch | *Myriophyllum aquaticum* | 7 | 51 | Wilson *et al.* (2001) |

**TABLE 13.10**
**Photolysis Half-Lives of Triazine Herbicides in Natural Waters (in days)**

| | Lake Water | River Water | Groundwater | Distilled Water |
|---|---|---|---|---|
| *Atrazine* | 53 | 43 | 26 | 35 |
| *Propazine* | 65 | 53 | 29 | 33 |
| *Prometryne* | 51 | 52 | 28 | 32 |

*Note:* Dark blanks have been deducted.

*Source:* Data from Konstantinou *et al.* (2001) *Journal of Environmental Quality* 30(1): 121–130.

Outflows from the Des Plaines wetland cells contained reduced amounts of atrazine compared to the river water inputs. During 1991, atrazine peaked in the river due to two rain events. Only about 25% of the incoming atrazine was removed in wetland cell EW3, but 95% was removed in wetland cell EW4, due to longer detention time. Either time averaged or dynamic models, both embodying a nonideal mixing model, fit the data for wetland EW3 (Figure 13.7). A removal rate coefficient ($k$) of approximately 14 m/yr calibrated the models to the data.

McKinlay and Kasperek (1999) tested two-stage, recirculating VF mesocosms. Clubrush (*Schoenoplectus* (*Scirpus*) *lacustris*), cattails (*Typha latifolia*), iris (*Iris pseudocorus*), and *Phragmites australis* provided complete removal of atrazine, as did sterile unplanted systems, but rates were slower in the sterile systems. These results confirm those of Stearman *et al.* (2004) for a SSF system treating simazine, another triazine herbicide (Table 13.11). There were not large differences among plant species. Acclimatization was observed in serial dosing, with later atrazine doses removed at higher rates. The fastest removals showed a half-life of about five days.

Moore *et al.* (2000b) dosed atrazine into 0.1-ha FWS demonstration wetlands at Oxford, Mississippi (Rodgers and Dunn, 1992). Between one third and two thirds of the atrazine was removed, with observed half-lives of 16 to 48 days. Atrazine was not detected in plant or sediment samples.

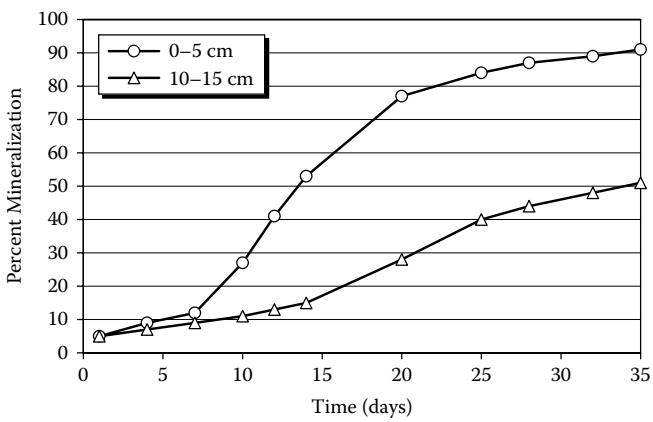



**FIGURE 13.6** The mineralization of atrazine to $CO_2$ in treatment wetland sediments. (Data from Anderson *et al.* (2002) *Water Research* 36(19): 4785–4794.)

Detenbeck *et al.* (1996) performed comprehensive studies on FWS wetlands, for the purpose of determining ecological effects of atrazine on receiving wetlands. Experiments were carried out in the downstream wetland portions of four of the eight outdoor experimental wetland/stream mesocosms located at the U.S. EPA Ecological Research Station in Monticello, Minnesota. Dominant emergent plants consist of cattail (*Typha* spp.), reed canary grass (*Phalaris arundinacea*), and wild rice (*Zizania aquatica*). Dominant submersed vegetation consists of waterweed (*Elodea canadensis*), pondweed (*Potamogeton* spp.), and coontail (*Ceratophyllum demersum*), and floating-leaved vegetation dominated by duckweed (*Lemna minor*). Half-lives of 8 to 14 days were found. Sorption measurements gave isotherms for sediments that agreed with those found by Alvord and Kadlec (1995), but sorption alone predicted half-lives of 36 days. Therefore, other mechanisms were involved in the overall reductions. Ecosystem effects included repression of periphyton, and early senescence of the wild rice.

Runes *et al.* (2001) evaluated the microbial degradation of atrazine in sediments taken from a constructed wetland treating irrigation runoff from a container nursery near Portland, Oregon. To stimulate atrazine-degrading ability in wetland sediments, dried shoot material from cattail was incorporated, and sediment soil was bioaugmented with an atrazine spill-site soil known to contain atrazine-mineralizing microbial populations. The population density of atrazine-degrading microorganisms in unbioaugmented sediment was increased from about $10^2$/g to $10^4$/g by bioaugmentation (1:100 w/w) and increased by another 60-fold ($6 \times 10^5$/g) after incubation with 10 μg/g of atrazine. A high population of atrazine degraders (approximately $10^6$/g) and enhanced rates of atrazine mineralization also developed in bioaugmented sediment after incubation in flooded mesocosms planted with *Typha latifolia* and supplemented with atrazine (3.2 mg/L, 1 μg/g sediment). In the absence of atrazine, neither the population of atrazine degraders nor the atrazine mineralizing potential of bioaugmented sediment increased, regardless of the presence or absence of cattails.

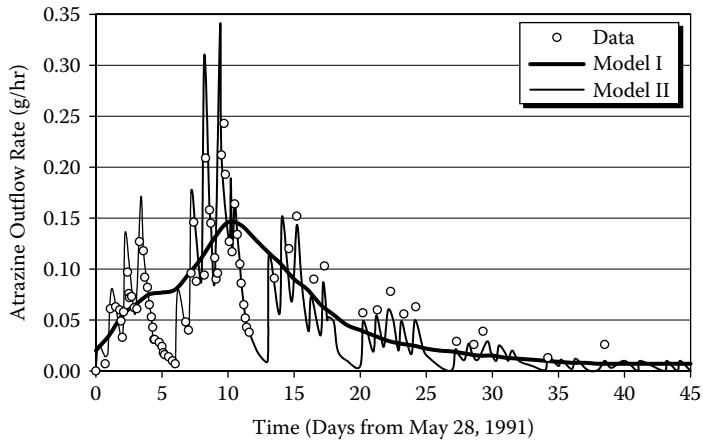



**FIGURE 13.7** Des Plaines wetland EW3 atrazine mass outflow as a function of time. Model I is a steady-flow, tracer-based RTD calculation. Model II is a dynamic simulation which fits the hourly changes in flows of water and atrazine. Both models use the same postulated first-order homogeneous reaction rate constant ($k$ = 14.4 m/yr). (Data from Alvord and Kadlec (1996) *Ecological Modelling* 90(1): 97–107.)

Runes *et al.* (2003) then conducted field experiments in a constructed wetland consisting of five sequential cells, each approximately 3 m × 40 m filled with a silt loam, and planted with *Typha latifolia* at greater than 75% coverage. Water came from a container nursery facility, and was delivered episodically, and dosed with atrazine. Flow events were ranked by intensity (low: less than 250 L/min; medium: 250 to 500 L/min; high: greater than 500 L/min) and frequency of the events (low: fewer than 6 events per week, medium: 6 to 9 events per week, high: greater than 9 events per week). Atrazine treatment efficiency at the outlet of the constructed wetland during a seven-day period ranged from 76 to 82% in 1998 (experiments 1 through 3) and 83 to 84% in 1999 (Table 13.12), but zero treatment was found in one monitoring period. For all experiments, deethylatrazine and deisopropylatrazine accounted for 13 to 21% of the initial application. Hydroxyatrazine was rarely detected in the water.

**TABLE 13.11**
**Percentage Simazine Removed in HSSF Wetland Cells**

| HRT (days) | Vegetated (%) | HRT (days) | Unvegetated (%) |
|---|---|---|---|
| 2.3 | 65 | 2.1 | 64 |
| 3.5 | 67 | 3.2 | 64 |
| 5.1 | 78 | 4.1 | 57 |
| 7.9 | 81 | 6.4 | 65 |
| 13.3 | 96 | 8.3 | 78 |
| 20.6 | 93 | 8.3 | 64 |
| 20.7 | 95 | 12.9 | 67 |

*Source:* Data from George *et al.* (2003) *Water Environment Research* 75(2): 101–112.

## 13.5 CYANIDE

Cyanide has an active cycle in nature, where it is present in green plants as part of the methionine cycle (ethylene production), and is therefore prevalent in autumn foliage and ripening fruit (Neuhauser and Swallow, 2005). If not handled properly, black and bing cherries (*Prunus serotina* and *Prunus avium*) and cassava can cause toxicity in humans. There are clear risks associated with cyanide release into the environment because of toxicity. The most toxic species are the free cyanides (HCN and $CN^-$), which are infrequent in uncontaminated soils and groundwater. Iron cyanide is less toxic, and often predominates in environmental samples. The U.S. Environmental Protection Agency and various state agencies regulate cyanide in groundwater and surface water differently. Agencies either choose to regulate the total or the free cyanide form. Because of the potential for photolytic production of free cyanide from iron cyanide, regulatory agencies may consider total cyanide to be indirectly toxic. U.S. EPA criteria for free cyanide include a drinking water standard of 200 µg/L, a freshwater acute criterion of 22 µg/L, and a freshwater chronic criterion of 5.2 µg/L.

Cyanide compounds are classified into those that are readily converted into hydrogen cyanide under acidic conditions, or into CN under basic conditions, and those that are strongly complexed. This Weak Acid-Dissociable cyanide (WAD) is distinguished from strong complexes formed with iron and cadmium, as examples. Total cyanide is the sum of both forms (Kavanaugh *et al.*, 2003).

Cyanide is a pollutant of concern in several situations. Gold mine wastewaters may contain cyanide as the result of the extraction process. Manufactured gas plants historically left residues rich in cyanide and other constituents, which may subsequently leach to natural water bodies. Aluminum processing produces spent pot liners, which, when landfilled, cause similar leaching of cyanide.

**TABLE 13.12**
**Percentage Atrazine Removed in FWS Wetland Cells under Event-Driven Operation**

| Runoff Frequency | Runoff Intensity | Cell 1 | Cell 3 | Cell 5 |
|---|---|---|---|---|
| High | Medium | 3 | 80 | 82 |
| High | Medium | 90 | 69 | 76 |
| Medium | High | 99 | 94 | 80 |
| High | High | 27 | 76 | 83 |
| Low | Low | 55 | 48 | 84 |

*Source:* Data from Runes *et al.* (2003) *Water Research* 37: 539–550.

Both aerobic and anaerobic microorganisms are able to destroy cyanide (Hinchee *et al.*, 1995; Goncalves *et al.*, 1998; Alexander, 1999). Further, plants such as willows are able to ingest cyanide, and metabolically degrade it (Larsen *et al.*, 2004). As a consequence, a natural treatment system that provides maximum opportunity for contact between cyanide-contaminated water and plants and microbes would have strong promise for removal and degradation of cyanide. Treatment wetlands are leading candidates to accomplish these goals.

The processing of cyanide in wetlands is known to involve microbial degradation, volatilization, plant uptake, and complexation with iron (Young, 2003). Additionally, iron cyanide decomposes photolytically to free cyanide (Kim *et al.*, 1998). The aerobic and nutrient-rich environment promotes the growth of the microbial population capable of uptake, conversion, sorption, and/or precipitation of thiocyanate, cyanide, ammonia, nitrate, sulfate, and metals. The microbial removal involves production of ammonia (Goncalves *et al.*, 1998; Akcil, 2003):

$$M(CN)_2 + 4H_2O + O_2 \rightarrow M^{2+} + 2HCO_3^- + 2NH_3 \quad (13.2)$$

$$SCN^- + 3H_2O + 2O_2 \rightarrow SO_4^{2-} + NH_4^+ + HCO_3^- + H^+ \quad (13.3)$$

where $M$ represents a metal species.

Various *Pseudomonas* species are responsible for oxidation of cyanide and thiocyanate. Some of the organisms known to oxidize cyanide include species of the genera *Actinomyces*, *Alcaligenes, Arthrobacter, Bacillus, Micrococcus, Neisseria, Paracoccus*, *Pseudomonas*, and *Thiobacillus* (Akcil and Mudder, 2003).

However, plant uptake is also potentially active (Ebbs *et al.*, 2003; Larsen *et al.*, 2004; Samiotakis and Ebbs, 2004; Larsen *et al.*, 2005). Larsen *et al.* (2005) found Michaelis-Menten kinetics with half-lives of less than a day (batch, with leaves or roots). Samiotakis and Ebbs (2004) tracked the transport and metabolism of two cyanide compounds, potassium cyanide and potassium ferrocyanide, by willow (*Salix eriocephala*), using a hydroponic system that preserved cyanide speciation and solubility. The cyanide compounds were labeled with $^{15}N$ to quantify transport. These analyses revealed that while little free cyanide was detected in the aerial tissues of plants exposed to either of these two cyanide compounds, significant enrichments in $^{15}N$ were observed, suggesting transport and subsequent metabolism of free cyanide as well as ferrocyanide. Yu *et al.* (2005) found temperature to be a factor for uptake by willow (*Salix babylonica*), from which Arrhenius factors of about 1.07 may be calculated. Disappearance rates in solution, using detached leaves, were characterized by half-lives of about eight hours.

These encouraging process studies have led to several treatment wetland projects targeting cyanide reduction. Garcia (2003) found more than 90% reduction in wetland mesocosms (*Typha*). Alvarez *et al.* (2006a; 2006b) experimented with wetland-based passive systems at laboratory and field scale at the site of a gold mine in northern Spain. The pilot-scale system was comprised of aeration cascades and aerobic and anaerobic cells. Results suggest that this technology is able to treat wastewaters with about 25% reduction for dissolved cyanide for nominal detention of six days. The waste strengths in this study were quite high, with total cyanide of about 200 mg/L and dissolved cyanide of 75 mg/L. The removal of WAD cyanide was 99%, from 15 mg/L to 0.14 mg/L. Vegetated cells (*Typha latifolia*) performed only marginally better than gravel control cells.

A large-scale trial was conducted at the Rustler's Roost mine near Darwin, Australia. A 3.2-ha site was configured as ponds and wetlands, covered with peat, and planted with *Typha orientalis* (Halford, 1999). A light hydraulic loading (<1 cm/d) was applied during the dry season of 1996 (July–September). The incoming water had total cyanide of about 2,000 µg/L, and the system discharge was 40–400 µg/L. Weak acid dissociable cyanide (WAD) was reduced from about 900 to 200 µg/L.

Wetland remediation of groundwaters contaminated with cyanide from landfills associated with the aluminum industry has been shown to be effective (Gessner *et al.*, 2005). A pilot project at Alcoa, Tennessee, consisted of two lined FWS cells operated in series. Softstem bulrushes (*Schoenoplectus* (*Scirpus*) *tabernaemontani*) were planted in Cell 1 (around 5 m × 15 m), and cattails (*Typha latifolia*) were planted in Cell

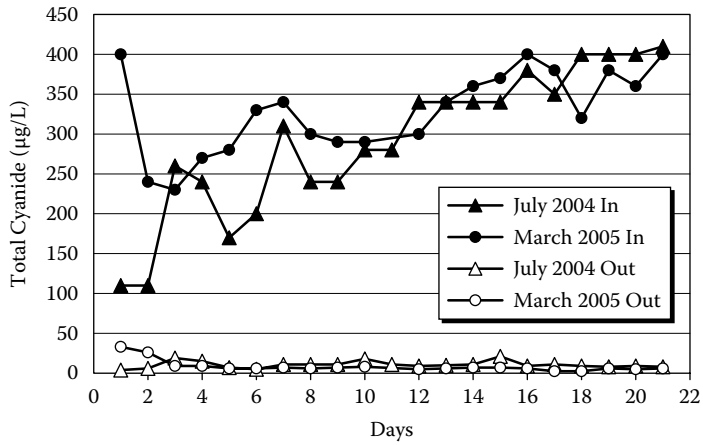



**FIGURE 13.8** Input–output data for two intensive sampling periods for total cyanide at the Duck Spring FWS treatment wetland at Alcoa, Tennessee. (Unpublished data from Alcoa Aluminum.)

2 (around 7 m × 12 m). Inflow concentrations typically ranged from 0.03 mg/L to 0.05 mg/L, while all outflow concentrations except one were below the lowest reporting level (LRL) of 0.01 mg/L. Total cyanide showed a statistically significant ($p < 0.00005$, N = 25) decrease in mean concentration (88%) between the treatment wetland inflow and outflow. By the time water reaches the midpoint of the treatment wetland, total cyanide average concentrations are at, or below, the LRL. A second pilot project was operated during 2004–2006, also in Alcoa, Tennessee. This 2,300-$m^2$ system was one lined FWS cell that received about 300 µg/L of total cyanide at 130 $m^3$/d, resulting in a nominal detention time of about three days. This system achieved well over 90% reduction in total cyanide (Figure 13.8).

The early successes of wetland treatment have caused optimism for continued application of wetlands to reduce cyanide. The PIRAMID summary research report stated that "the safe, passive destruction of residual cyanide leaching from gold mine tailings has been achieved using compost-based wetland-type passive systems" (PIRAMID Consortium, 2003b). This report goes on to suggest that WAD removal in such systems can be expected to average around 10.6 mg/$m^3$·d, which equates to an area-adjusted removal rate of 4.2 g/$m^2$·d for a substrate 0.5 m in depth. However, this recommendation is ten times higher than the results of Alvarez *et al.* (2006b), who found 0.49 g/$m^2$·d for WAD cyanide.

## SUMMARY

Wastewaters containing a wide variety of hydrocarbon constituents are presently being treated in wetland systems, and performance has been generally good. Rate constants are typically high, especially for volatile materials. The surfactant work has collectively shown that individual compounds may behave quite differently, depending upon isomeric form and upon small changes in molecular weight. This is somewhat daunting, because it means pre-project investigations for specialty chemicals are a necessity; rate constants may not be inferred from data on related compounds. The available data on phenol show that microcosms are only qualitative predictors of the performance of full-scale wetland systems. This means that pilot-scale tests are a necessity for generating design data.

However, there is some removal of nearly every organic compound tested in wetland treatment systems, whether full scale, pilot scale, or microcosm. This includes organics from very important generators: pesticides, petroleum and petrochemicals, food wastes, and remediation. In each of these categories, there are multiple studies that have initiated the design database. Current knowledge shows great promise for wetlands technology for organics control, especially in situations where passive systems can be fit into the landscape and accumulated organics will not pose a threat to wetlands biota.

# 14 Event-Driven Wetlands

The treatment of stormwaters of various origins is of growing concern as attempts to rectify point-source pollution reach maturity. Runoff from urban, agricultural, and industrial sources comprises a sizeable fraction of the total pollution load to receiving waters in many locations. Often, these sources are relatively dilute compared to primary or secondary domestic sewage but, nevertheless, they may negatively impact receiving waters.

Urban stormwater wetlands were first surveyed by Strecker *et al.* (1992), who documented the performance of 25 natural and constructed wetlands treating runoff. The implementation of the technology and the knowledge base continued to build, resulting in a compilation of data from 76 wetlands worldwide by Wong *et al.* (1999), and from 49 wetlands in the United States by Carleton *et al.* (2001). In North America, these are all FWS systems, and that is the predominant wetland type elsewhere as well. A typical configuration consists of a sedimentation basin as a forebay, followed by some combination of marshes and deeper pools. Design guidelines have now been promulgated by a number of sources (for example: Schueler, 1992; Breen and Lawrence, 1998; Wong *et al.*, 1999; LEC, 2000; Center for Watershed Protection, 2001).

Agricultural stormwater occurs as runoff from crops and pastures. Early work on constructed wetlands for row crop runoff control was centered in the state of Maine (Higgins *et al.*, 1993) (Wengrzynek and Terrell, 1990; U.S. Department of Agriculture, 1991). Runoff from sugarcane is being treated in the huge FWS wetlands (>16,000 ha) called stormwater treatment areas (STAs) of South Florida (Goforth, 2001). However, there has also been significant application of constructed wetlands at a more modest scale for vegetable farming (Rushton and Bahk, 2001). Nonetheless, effective control of runoff may require consideration of the entire watershed (Crumpton, 2000). More recently, attention has been on controlling pasture runoff (Tanner *et al.*, 2003).

Industrial stormwaters have received less attention than urban and agricultural sources. These typically will contain contaminants specific to the industry in question. For instance, rain runoff from a fertilizer plant site will potentially contain high levels of nutrients, whereas rain runoff from a petroleum facility will carry hydrocarbons. For instance, in South Africa, a number of reed bed wetlands were established to treat waters generated from truck washing operations at oil industry depots (Wood, 1993). An HSSF wetland treated runoff from a 0.8-ha vehicle yard in Surprise, Arizona, with 54–92% removal of oil and grease (Wass and Fox, 1993).

Although rainfall runoff is the source of water for many event-driven wetlands, there are other situations in which a treatment wetland is subjected to episodic flows. These include wetland systems that treat stormwater, urban or agricultural runoff, combined sewer overflows, and treatment systems that are operated in batch mode. Whatever the source, wetlands that receive pulsed flows utilize the suite of wetland removal mechanisms in different ways than their continuous-flow counterparts.

## 14.1 SOURCE CHARACTERIZATION

### Incoming Flows

The amount of water to be expected from a given watershed is variable and is keyed to rainfall in the contributing basin. The land uses and soils in the contributing basin are an important modifier of the runoff and infiltration. The antecedent dryness in the basin is also a contributing factor. Detailed methods of estimating runoff are available, for instance, the SCS (U.S. Soil Conservation Service) method (McCuen, 1982), which has been summarized by Novotny (1995). There are also numerous computer models that account for very detailed features of the contributing watershed and produce both the quality and quantity of the runoff to be treated. The focus here is the treatment wetland, and it will be presumed that the incoming flows will be characterized to the appropriate level of detail.

For purposes of rough estimation, the rational formula may be used to estimate peak flow, $Q_P$ (ASCE, 2006):

$$Q_P = C_R \cdot I \cdot A_b \tag{14.1}$$

where

$A_b$ = watershed area, $m^2$
$C_R$ = runoff coefficient, dimensionless
$Q_P$ = peak runoff flow, $m^3/h$
$I$ = average rainfall intensity, m/h

The values of $C_R$ are a function of land use and storm intensity, as well as the climatological region. An example of the range of $C_R$ for an arid region is shown in Table 14.1. To illustrate the regional effect, consider that agricultural land in Arizona is rated at $C_R = 0.1–0.2$ for high return-frequency events, whereas the value for agricultural land in South Florida is $C_R \approx 0.5$. Water is used consumptively in arid regions, whereas the high water table and rainfall in a wet region leads to more runoff. Modifications to this simplest formula have been presented on a site-specific basis (see, for example, Brezonik and Stadelmann, 2002; ASCE, 2006).

**TABLE 14.1**
**Runoff Coefficient (C) for Use in the Rational Method in Maricopa County, Arizona**

| | Return Period | | | |
|---|---|---|---|---|
| **Land Use** | **2–10-year** | **25-year** | **50-year** | **100-year** |
| **Streets and Roads** | | | | |
| Paved roads | 0.75–0.85 | 0.83–0.94 | 0.9–0.95 | 0.94–0.95 |
| Gravel roadways and shoulders | 0.6–0.7 | 0.66–0.77 | 0.72–0.84 | 0.75–0.88 |
| **Industrial Areas** | | | | |
| Heavy | 0.7–0.8 | 0.77–0.88 | 0.84–0.95 | 0.88–0.95 |
| Light | 0.6–0.7 | 0.66–0.77 | 0.72–0.84 | 0.75–0.88 |
| **Business Areas** | | | | |
| Downtown | 0.75–0.85 | 0.83–0.94 | 0.9–0.95 | 0.94–0.95 |
| Neighborhood | 0.55–0.65 | 0.61–0.72 | 0.66–0.78 | 0.69–0.81 |
| **Residential Areas** | | | | |
| Lawns—flat | 0.1–0.25 | 0.11–0.28 | 0.1–0.3 | 0.13–0.31 |
| Lawns—steep | 0.25–0.4 | 0.28–0.44 | 0.3–0.48 | 0.31–0.50 |
| Suburban | 0.3–0.4 | 0.33–0.44 | 0.36–0.48 | 0.38–0.5 |
| Single family | 0.45–0.55 | 0.5–0.61 | 0.54–0.66 | 0.56–0.69 |
| Multi-unit | 0.5–0.6 | 0.55–0.66 | 0.6–0.72 | 0.63–0.75 |
| Apartments | 0.6–0.7 | 0.66–0.77 | 0.72–0.84 | 0.75–0.88 |
| Parks/cemeteries | 0.1–0.25 | 0.11–0.28 | 0.12–0.3 | 0.13–0.31 |
| Playgrounds | 0.4–0.5 | 0.44–0.55 | 0.48–0.6 | 0.5–0.63 |
| Agricultural areas | 0.1–0.2 | 0.11–0.22 | 0.12–0.24 | 0.13–0.25 |
| Bare ground | 0.2–0.3 | 0.22–0.33 | 0.24–0.36 | 0.25–0.38 |
| Undeveloped desert | 0.3–0.4 | 0.33–0.44 | 0.36–0.48 | 0.38–0.5 |
| Mountain terrain (slopes > 10%) | 0.6–0.8 | 0.66–0.88 | 0.72–0.95 | 0.75–0.95 |

*Source:* Adapted from the *Drainage Design Manual for Maricopa County*, Volume 1.

For modeling purposes, the time series of water flows entering the treatment wetland inlet hydrograph is sometimes needed. This time series is driven by the pattern of rainfall associated with a particular event (represented by a *hyetograph*), together with the collection and transfer characteristics of the basin. In some instances, such transfer is solely by gravity, but in some cases pumps may be used. Usually, the inlet hydrograph contains both a rising and falling limb, although the rising limb is normally quite steep (for an example, see Figure 14.1).

In some instances, the events are separated by interevent periods of no inflow to the treatment wetland. These periods are important because the wetland will act as a batch reactor during much of these no-inflow durations. A typical sequence is (1) wetland filling with no outflow, (2) flow through with both inflow and outflow, (3) draining with no inflow, and (4) finally, a batch-holding mode with neither inflow nor outflow. The rates and duration of these inflows and outflows are in part controlled by structures. The volume of water in the wetland is also in part controlled by structures, but evapotranspiration may be an important component during interevent periods. The durations of no-flow periods are very much a function of the climatological region in which the system is located. For instance, in Florida, during the rainy season, the most frequent periods are measured in hours (4–20 hours, Wanielista and Yousef, 1991). In contrast, the dry season of central north island New Zealand has periods of months with no rain, and interevent periods can be several months (Tanner *et al*., 2005b).

Many stormwater wetlands are fed by pumps. These may range in size from small sump pumps for small localized urban systems to the huge pumps that send water to the Florida stormwater treatment areas (a.k.a. STAs, or treatment wetlands). Those Florida inflow and outflow pumps are among the largest in the world, up to three stories tall, with capacities up to 10,000,000 $m^3$/d (Figure 14.2). A key feature of pumped systems is intermittent feed to the treatment wetland, usually at fixed but incremental rates, corresponding to the number of pumps that are operated, for periods of time dictated by conditions in the contributing watershed.

There is typically a hydraulic limitation to the event size that may be treated in a given wetland. It is generally not feasible to size a wetland to treat the 100-year return frequency storm because of cost and footprint considerations and because of the need for maintenance water for the (large) wetland under normal conditions. Consequently, part of the design decision process is the determination of the maximum design storm to be treated. Even if the system is sized to treat

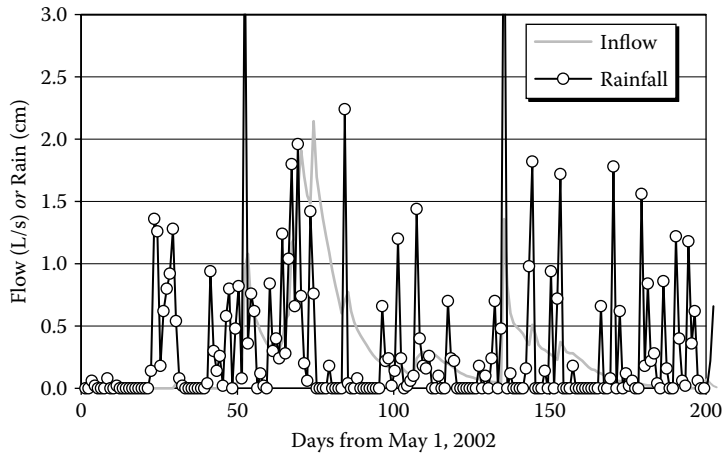



**FIGURE 14.1** Time series of flows entering a treatment wetland from an improved pasture in Toenepi, New Zealand. Several rain events occurred during this winter wet season, as indicated by the repeated spikes in inflow. The runoff coefficients were 0.23–0.30 and were the result of tile drains. (Adapted from Tanner *et al*. (2005b) *Agriculture, Ecosystems and Environment* 105(1–2): 145–162.)

a modest-sized storm flow, there remains the possibility that dry conditions might persist in some years to the point that the integrity of the wetland ecosystem is jeopardized. If such detrimental dryout conditions are expected, then a source of irrigation water for ecosystem maintenance should be identified.

### Incoming Concentrations and Loads

Concentrations of most parameters in stormwater are time dependent, as are the flows. Stormwater concentrations and loads are episodic due to periods of dryfall and deposition, followed by the first flush of runoff after rain, followed by exponential decreases in runoff constituent concentrations as storages rinse from the landscape, and finally, dry conditions and deposition until the next storm event. The time series of concentrations in the inflow to the wetland is called the *chemograph*. An example chemograph for an agricultural runoff wetland, targeting nitrogen reduction, is shown in Figure 14.3.

In some watersheds, the chemograph is not synchronized with the hydrograph, but instead provides higher concentrations early in the inflow event. This phenomenon is termed *first-flush* behavior, referring to the surge of pollutants contained in the first water to leave the contributing basin. For instance, Wanielista and Yousef (1991) report that, in Florida, the first 25 mm of runoff from urban systems typically carries 90% of the pollution. However, for the large agricultural



**FIGURE 14.2** The outflow pump station from STA1E (Stormwater Treatment Area 1 East) of the Everglades Protection Project. The capacity of this pump station is 9,740,000 $m^3$/d, and it serves to drain a 2,700-ha FWS wetland.

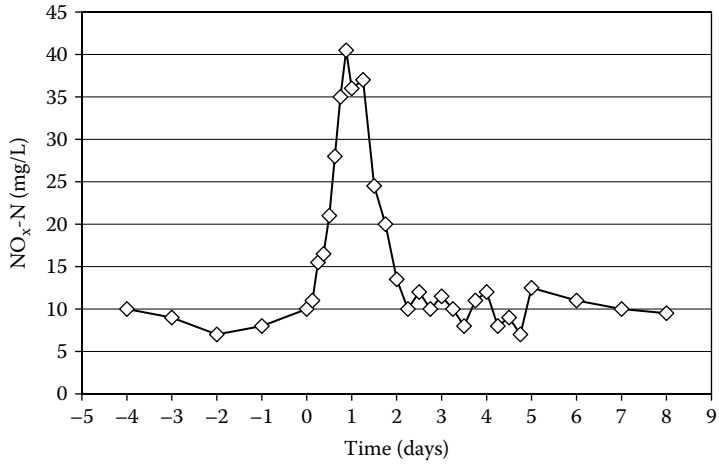



**FIGURE 14.3** The time series of nitrate nitrogen arriving at an agricultural runoff treatment wetland in McDowell County, North Carolina. The rain event started at time zero and drove runoff that entered the wetland by stream flow. (Adapted from Kao and Wu (2001) *Water Science and Technology* 45(3): 169–174.)

watershed of South Florida, there is no such first-flush effect. Despite the site-specific nature of the chemograph, there is no reliable database documenting dynamic time series of concentrations in any given watershed. Of necessity, average concentrations of some sort must be used, of which the flow-weighted concentration is most useful. This may be evaluated on a long-term basis or as an event mean concentration for the inlet water:

$$C_e = \frac{\int QCdt}{\int Qdt} \tag{14.2}$$

where

$C$ = instantaneous concentration, mg/L
$C_e$ = event mean concentration, mg/L
$Q$ = instantaneous flow, $m^3/h$
$t$ = time, h

and where integration is over the period of one event. The numerator is the mass of pollutant entering or leaving during the event. As described in Chapter 6, the performance of the wetland may be described in terms of the loads applied to and emanating from the system.

Tables 14.2 and 14.3 provide long-term mean concentrations for constituents in urban stormwater. The averages are flow-weighted to provide realistic estimates of the total constituent load that escapes during multiple storm events. Instantaneous concentrations may rise considerably higher than these averages. Pollutant concentrations and loads generally range from low levels from undeveloped and park lands to low-density residential and commercial, to agricultural, to higher-density residential and commercial, and finally to high-density commercial, industrial, and agricultural land uses. Mean concentrations per event for $BOD_5$ vary from below detection for undeveloped lands to 20 mg/L for high-density urban areas. Total suspended solids concentrations vary from about 10 mg/L for undeveloped areas up to 150 mg/L for high-density urban areas. Typical concentrations for other stormwater pollutants are also summarized in Tables 14.2 and 14.3.

The mass loading rates represent normalized pollutant loads that are somewhat independent of local rainfall amounts. Because pollutant loads per area per time are relatively constant between similar land use areas, variable local rainfall washes these loads off the land in a few large events or over many smaller events. Urban pollutant loads increase with the imperviousness of the watershed. Although 20 to 40% of the material on street surfaces is organic, it does not biodegrade easily because it comes from leaf and wood litter, rubber, and road-surface material (Novotny, 1992). The high metal content of highway solids comes from vehicle emission. Novotny (1992) reported that the average total nitrogen load from urban lands is 5 kg/ha·yr (1 to 38.5 kg/ha·yr), and the total phosphorus load averages 1 kg/ha·yr (0.5 to 6.25 kg/ha·yr).

Constructed wetlands are being increasingly used to treat runoff from intensive animal operations (CH2M Hill and Payne Engineering, 1997; Tanner *et al.*, 2003) and from crops (U.S. Department of Agriculture, 1991; Crumpton, 2000). Concentrations and loads from agricultural land uses vary considerably. Flows and loads are typically highest from areas with high animal densities or high fertilization rates. Runoff pollutant concentrations from animal feedlots can be extremely high unless runoff is collected and treated. Pollutants from feedlot runoff typically include high levels of organic and inorganic solids and associated nutrients. Nutrient concentrations and loads from row crops and pastures depend on fertilization practices and type of soil.

There exist many computer models for the amounts of contaminants that may be expected in runoff from different

**TABLE 14.2**
**Pollutant Concentrations for Source Areas for Stormwaters**

| Constituent | TSS[a] (mg/L) | TP[b] (mg/L) | TN[c] (mg/L) | *E coli*[a] (1,000 #/mL) | Cu[a] (µg/L) | Pb[a] (µg/L) | Zn[a] (µg/L) |
|---|---|---|---|---|---|---|---|
| Residential roof | 19 | 0.11 | 1.5 | 0.26 | 20 | 21 | 312 |
| Commercial roof | 9 | 0.14 | 2.1 | 1.1 | 7 | 17 | 256 |
| Industrial roof | 17 | — | — | 5.8 | 62 | 43 | 1390 |
| Comm./res. parking | 27 | 0.15 | 1.9 | 1.8 | 51 | 28 | 139 |
| Industrial parking | 228 | — | — | 2.7 | 34 | 85 | 224 |
| Residential street | 172 | 0.55 | 1.4 | 37 | 25 | 51 | 173 |
| Commercial street | 468 | — | — | 12 | 73 | 170 | 450 |
| Rural highway | 51 | — | 22 | — | 22 | 80 | 80 |
| Urban highway | 142 | 0.32 | 3 | — | 54 | 400 | 329 |
| Lawns | 602 | 2.1 | 9.1 | 24 | 17 | 17 | 50 |
| Landscaping | 37 | — | — | 94 | 94 | 29 | 263 |
| Driveway | 173 | 0.56 | 2.1 | 17 | 17 | — | 107 |
| Gas station | 31 | — | — | — | 88 | 80 | 290 |
| Auto recycler | 335 | — | — | — | 103 | 182 | 520 |
| Heavy industrial | 124 | — | — | — | 148 | 290 | 1,600 |

*Source:* Data from Center for Watershed Protection (2001) *New York State Stormwater Management Design Manual*. Report by the Center for Watershed Protection (CWP) for the New York State Department of Environmental Conservation, Albany, New York.

[a] Data from Claytor and Schueler (1996) *Design of Stormwater Filtering Systems*. Center for Watershed Protection: Ellicott City, Maryland.

[b] Average of data from Steuer *et al.* (1997) *Sources of contamination in an urban basin in Marquette, Michigan, and an analysis of concentration, loads, and data quality*. Water Resources Investigation Report 97–4242, U.S. Geological Survey; Bannerman *et al.* (1993) *Water Science and Technology* 28(35): 241–259; Waschbusch (2000) *Sources of phosphorus in stormwater and street dirt from two urban residential basins in Madison, Wisconsin, 1994–1995*. Proceedings of the National Conference on Tools for Urban Water Resource Management and Protection; U.S. Environmental Protection Agency: Washington, D.C, pp. 15–55.

[c] Data from Steuer *et al*. (1997) *Sources of contamination in an urban basin in Marquette, Michigan, and an analysis of concentration, loads, and data quality*. Water Resources Investigation Report 97–4242, U.S. Geological Survey.

landscapes. Novotny (1995) summarized the characteristics of six models for urban watersheds, and seven for agricultural watersheds, and the number has grown since that time. A large part of the design of event-driven treatment wetlands is the determination of the flows and loads to be treated.

## Hydrology of Pulsed and Seasonal Systems

Event-driven wetlands are dynamic in all respects, and the principal underlying hydraulics exhibit variable water depths and flows. The behavior is strongly conditioned by the nature

**TABLE 14.3**
**Typical Concentration Data for Pollutants in Urban Stormwater**

| Constituent | Units | Urban Runoff | Source |
|---|---|---|---|
| TSS | mg/L | 54.5 | Smullen and Cave (1998) |
| TP | mg/L | 0.26 | Smullen and Cave (1998) |
| TN | mg/L | 2.00 | Smullen and Cave (1998) |
| Cu | µg/L | 11.1 | Smullen and Cave (1998) |
| Pb | µg/L | 50.7 | Smullen and Cave (1998) |
| Zn | µg/L | 129 | Smullen and Cave (1998) |
| Fecal coliforms | 1,000 CFU/mL | 1.5 | Schueler (1999) |

*Source:* Data from Center for Watershed Protection (2001) *New York state stormwater management design manual*. Report by the Center for Watershed Protection (CWP) for the New York State Department of Environmental Conservation, Albany, New York; pooled NURP/USGS Smullen and Cave (1998) *Updating the U.S. nationwide urban runoff quality database*. 3rd International Conference on Diffuse Pollution, 31 August–4 September 1998; and Schueler (1999) *Watershed Protection Techniques* 3(1): 551–596.

of inflow and outflow structures that may be designed to improve detention and treatment (Somes and Wong, 1997). The most general situation may have several complicating factors, but here a simple and common case is explored for purposes of illustration. It will be presumed that the wetland is relatively small and consequently behaves as a level-pool system, with no gradients in stage. Stormwater wetlands experience event flows and concentrations, followed by periods of batch operation. It is then necessary to account for the dynamics of water storage within the wetland.

The dynamic, level-pool water mass balance for the wetland for unsteady state inflows and meteorology is

$$\frac{d(Ah)}{dt} = Q_i - Q_o + A(P - ET - I) \tag{14.3}$$

where

$A$ = wetland wetted area, $m^2$
$Q_i$ = inflow, $m^3/d$
$Q_o$ = outflow, $m^3/d$
$P$ = rainfall, m/d
$ET$ = evapotranspiration, m/d
$h$ = water depth, m
$I$ = infiltration, m/d
$t$ = time, d

The stage is often controlled by an outlet structure, here supposed to be a rectangular weir. Thus the outflow-stage relation would be given by the Francis formula (French, 1985):

$$Q_o = \beta(h_{ow})^{\alpha} \tag{14.4}$$

where

$h_{ow}$ = height over the weir, m ($h_{ow} = H - H_w$)
$H$ = wetland stage, m
$H_w$ = elevation of weir, m
$Q_o$ = outflow, $m^3/d$
$\alpha$ = calibration exponent, dimensionless
$\beta$ = calibration constant, $(m^3/d)/m^{\alpha}$

It is noted that the wetted area is not constant but changes with stage according to the bathymetry of the wetland, represented by the stage–area–volume relationship (see Chapter 2). Further, the wetland catches rainfall over an area typically greater than the wetted area but loses water to *ET* from just the wetted area. Infiltration further contributes to water losses and may be a significant fraction of the output from the wetland, as in the case of the Hidden River wetland in Tampa, Florida (Carr and Rushton, 1995).

An example of this analysis is illustrated in Figure 14.4. For illustration, a wetland of 2,500 $m^2$ is subjected to a 5-cm rain event and the associated runoff from the contributing basin. The presumed conditions and wetland design are shown in Table 14.4. The wetland does not have a level bottom; rather, there is a presumed quadratic stage–volume relation, which implies that the wetted area increases linearly with stage. The hypothetical sequence of events is as follows:

- The wetland starting condition is partially full, with the water level 10 cm below the outlet weir.
- A steady rain begins at time zero and persists for one day.
- Runoff from the watershed begins after $t$ = 0.5 days and persists for 1.5 days.
- The wetland loses some water to *ET* (1.0 cm/d), and some to infiltration (1.0 cm/d).
- For the first half day ($t$ = 0–0.5 days), the wetland fills due to direct collection of rain.
- At $t$ = 0.65 days, the wetland has filled to the top of the outflow weir, and outflow commences.
- For the next half day ($t$ = 0.5–1.0 day), the wetland fills because of direct rain and incoming runoff.
- For the next day ($t$ = 1.0–2.0 days), the wetland fills because of incoming runoff.
- During $t$ = 2.0–3.1 days, there is no inflow, and the wetland is draining over the outflow weir. At $t$ = 3.1 days, the level has decreased to the top of the weir, and outflow stops.
- After time $t$ = 3.1 days, the wetland loses water to *ET* and infiltration, and the level continues a slow drop.

**TABLE 14.4**
**Parameters for a Hypothetical Stormwater Wetland Event Scenario**

| | |
|---|---|
| **Basin** | |
| Area = | 120,000 $m^2$ |
| Runoff coefficient = | 0.5 |
| **Wetland** | |
| WWAR = | 2 % |
| Nominal maximum depth = | 1.00 m |
| Wetland area full = | 2,500 $m^2$ |
| Wetland volume full = | 1,250 $m^3$ |
| Stage area: $A = aH$, $a$ = | 2,500 $m^2/m$ |
| Weir height = | 0.90 m |
| Weir coefficient = | 31,623 $(m^3/d)/(m^{1.5})$ |
| *ET* rate = | 1.00 cm/d |
| Infiltration rate = | 1.00 cm/d |
| Rain catchment = | 3,000 $m^2$ |
| **Rainfall** | |
| Period = | 1.00 d |
| Rain rate = | 5.00 cm/d |
| Basin catch = | 6,000 $m^3$ |
| Inflow start = | 0.50 d |
| Inflow end = | 2.00 d |
| Inflow rate = | 2,000 $m^3/d$ |
| Runoff = | 3,000 $m^3$ |
| Storm/wetland volume ratio = | 2.40 |

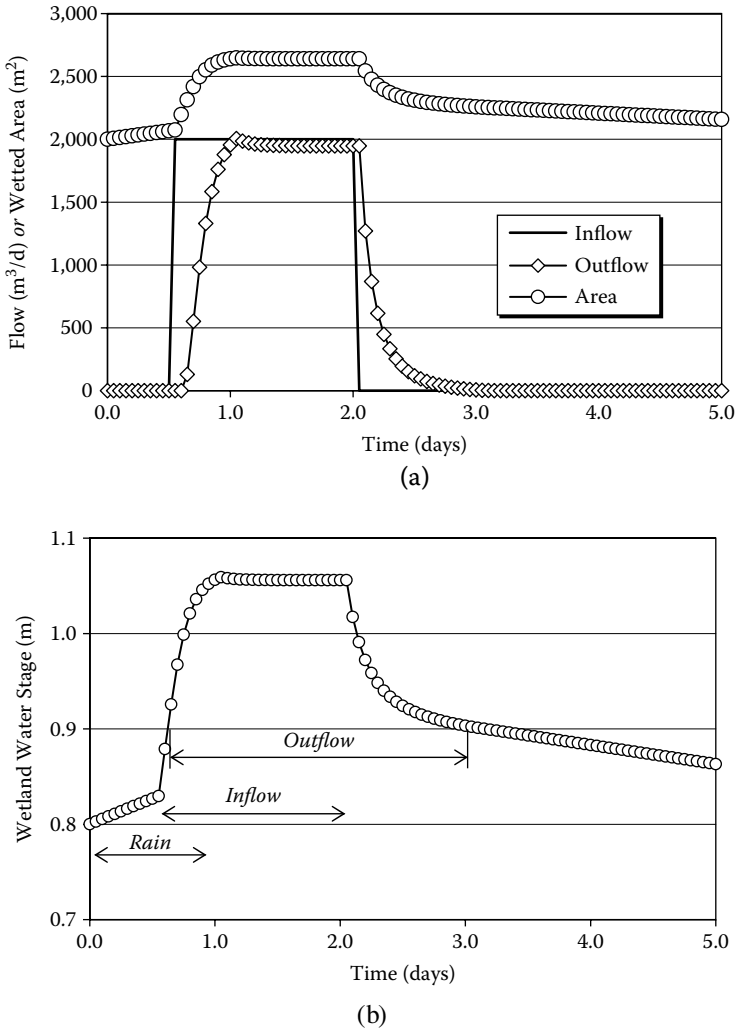



**FIGURE 14.4** Response of a hypothetical stormwater wetland to a one-day steady rain. Runoff into the wetland begins after half a day. The wetland fills, and outflow persists for just over three day. Infiltration and *ET* deplete the water after the event. See Table 14.4 for parameters.

Because of the assumed bathymetry, the wetted area first grows and then shrinks through this course of events.

This simple model has been found to fit event wetland data quite well (Kadlec, 1994; Hey *et al.*, 1994; Kadlec, 2001a). However, there are several different inlet and outlet structures that may be used, and the level-pool model must be modified accordingly for other situations.

## Flow and Capture

Event-driven wetlands operate with periods of flow through and periods of water-holding or batch processing. Any water that does not escape the wetland during a particular event will be held until the next event, and possibly longer. Therefore, it is subject to the water quality improvement functions of the wetland for not only the event duration but also the interevent period. Conversely, water that enters and leaves the wetland during the event is subject to treatment only during the (possibly brief) period of detention during the event. Therefore, at the first level of consideration, it would seem desirable to contain the entire volume of water associated with an event within the wetland. This requirement is often referred back to the contributing watershed in terms of the rain amount and runoff coefficient that generated the storm volume. The wetland may be sized to nominally contain the runoff volume associated with a rain event of specified amount or return frequency.

One limit on performance is the expulsion of water that resides in the wetland at the time of event initiation. This water is at a background concentration if the events are widely spaced. In this example, the time for background to be reached was on the order of two or three days. The length of time for the wetland water surcharge to dissipate, and thus for stage to reach the weir elevation, was also of the order of two or three days. If the events are more closely spaced, then the resident antecedent water will not be at background but rather will exhibit a concentration representing only partial treatment of the previous event. A second limit on performance is the

ultimate removal that would be associated with a continuous water input at the event average flow rate. This is likely to be a relatively low percentage reduction because of the high flow in the event.

A complicating factor in analysis is that wetlands are not hydraulically simple and do not operate on a basis of plug-flow displacement. Therefore, the fraction of incoming water that remains in the system for a particular event is not determined merely from displacement. Depending on the configuration of the wetland, some of the very first water that enters the wetland may find its way to the exit far in advance of nominal retention time during the event. This is partly due to preferential flow paths, and partly due to the positioning of water-level control structures.

If it is assumed that the wetland behaves hydraulically as several mixed tanks in series, it is reasonable to use the known detention time distribution to compute how much of the incoming water is still in the wetland after a period of flow of known magnitude. Figure 14.5 shows the results as a function of the nominal number of wetland volumes of water that have passed through the system. If the flow were plug flow, then the retention is 100% until the nominal detention time is reached, after which the new event water begins to exit. For less ideal flow conditions, the amount of the new event water that is held is less.

The presumption of Figure 14.5 is that all of the wetland water is involved in flow. That can be far from true because of wetland design factors (see Appendix B). For instance, if there is a point inlet to the system, and a point outlet aligned directly opposite, then the incoming pulse of water can travel directly to the outlet, never reaching zones to the side of the most direct flow path. That renders the corner areas essentially out of the flow path and reduces the volumetric efficiency quite markedly (Agunwamba, 2006). Walker (1998) investigated this situation using computational fluid mechanics (simulations) of unvegetated FWS basins. He determined that aspect ratio (L:W) was the major determinant of such shortcircuiting and calculated a retention-displacement chart (Figure 14.6).

For individual, separated inflow events, the concept of retained volume provides a strong correlating factor for performance. If the event is small, it is wholly contained and held until the next displacing flow. Treatment proceeds in the batch mode for the interevent duration. If the event is large, then it is processed by flow through at the detention time of the event flow. Because such flows are typically large, the wetland will have a short detention time, which results in decreased treatment performance. Therefore, mass reductions decrease (exponentially) as the number of nominal displacements caused by the event increases.

The storage and treatment potential for individual events may ultimately be linked to the sequence of events that are likely to occur over a long period of record. The three required probability distributions are for the event duration, the event intensity, and the interevent duration (Wong and Somes, 1995). To further complicate matters, when the wetland basin is "full," water will be diverted away (bypassed). Thus, a wetland that drains quickly, i.e., one with short detention time, will be in a position to detain more water upon refilling than a wetland that drains slowly. Wong and Geiger (1997) examined the runoff patterns for Melbourne, Australia, and concluded that the hydrologic effectiveness (the percentage of runoff that is exposed to treatment) was inversely proportional to the wetland detention time.

These factors all point to the existence of a set of performance determinants that go beyond those for continuous-flow treatment wetland systems. Batch interevent rate coefficients and background concentrations are important. The internal flow patterns are also important as determinants of the concentration and timing of the flushed antecedent water. The dynamics of water flow and storage of the wetland are important because they determine the volumetric constraints on performance.

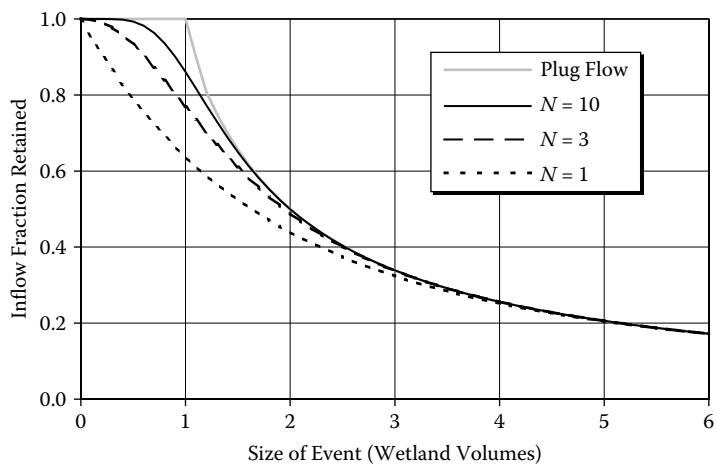



**FIGURE 14.5** Fraction of the inflow retained for a range of numbers of tanks in series for event volumes of different nominal displacements.

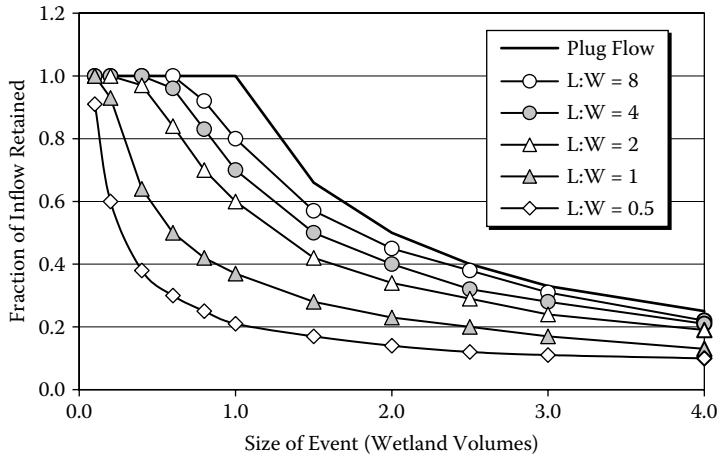



**FIGURE 14.6** Fraction of an inflow event contained in a wetland as a function of the event volume and the aspect ratio (L:W). (From Walker (1998) *Ecological Engineering* 10(3): 247–262. Reprinted with permission.)

Ultimately, the computed description of performance will either be expressed as long-term mass removal or by dynamic modeling that produces time series of outflow and effluent concentrations in response to the inlet time series. Most of the stormwater wetland literature reports mass removals over some period of record. However, there are now first-generation dynamic models for some applications. For instance, there is a flow and phosphorus model for application in south Florida called the Dynamic Model for Stormwater Treatment Areas (DMSTA) (Walker and Kadlec, 2005).

## 14.2 TECHNOLOGY STATUS

Data reporting falls into two general categories: event time-series analyses, and summaries of removal efficiencies. Two concepts are needed to organize these bodies of information: event mean concentrations and mass balancing. Because the concentration pulses and flow pulses are often out of sync, the event mean concentration (EMC) is used:

$$EMC = \frac{\Sigma(VC)}{\Sigma V} \tag{14.5}$$

where

$C$ = concentration in a water parcel, g/m$^3$ = mg/L

$V$ = water volume in the parcel, m$^3$

In effect, the EMC is the mass average concentration over the course of an event and may be calculated for both inlet and outlet flows of various types for the wetland. If only the directed inflow and outflow are considered, the EMC reduction is defined as

$$\text{Concentration Reduction} = \frac{(EMC_i - EMC_o)}{EMC_i} \tag{14.6}$$

The outflow from the wetland corresponding to a given inflow event may total less than the inflow due to infiltration and evapotranspiration. The mass load reduction for a given pollutant is

$$\text{Load Reduction} = \frac{(V_i EMC_i - V_o EMC_o)}{V_i EMC_i} \tag{14.7}$$

### Urban Stormwater

Runoff from roofs, lawns, parking lots, and other urban landscape features often contains several classes of pollutants (see Table 14.2). There will be small amounts of nitrogen and phosphorus and biochemical oxygen demand (BOD), but possibly relatively greater amounts of total suspended solids (TSS), metals, and perhaps pesticides. An important subset of urban runoff control is focused on highway runoff (Shutes *et al.*, 2001; Bulc and Sajn Slak, 2003; Pontier *et al.*, 2003). However, the greater number of systems are used to treat rainwater runoff from more generalized urban areas. Here the focus is upon gravity-driven constructed systems, as opposed to pumped or natural systems. Such systems are often quite small, serving small catchments (Figure 14.7).

In this section, attention is on gravity-fed, constructed systems treating urban stormwater. Pumped systems have been included in other summaries, such as Carleton *et al.* (2001) and Wong *et al.* (1999), but many of these do not reflect the random and episodic nature of wetlands receiving rain-driven events and are omitted here, as are natural wetland systems. A sampling of TSS performance is given in Table 14.5, in which it is clear that there is a wide spectrum of performances for all common constituents, with median removals far from 100%.

Removal percentages are insufficient for design purposes, and hence it is necessary to find relationships between system characteristics and performance measures. Carleton *et al.* (2001) concluded that long-term pollutant removals could be described in terms of the same kinds of first-order, steady-flow design equations currently employed for wastewater treatment wetlands. They present rate coefficients that



**FIGURE 14.7 (A color version of this figure follows page 550)** An urban stormwater wetland in the city of Blue Mountains, Australia.

are intended to be used in a procedure such as that described by Wong and Geiger (1997) and Wong *et al*. (2006). As discussed in Chapter 6, that method has inherent drawbacks, and calibrations for event-driven systems provide only central tendencies. Additionally, rate coefficients derived from single events may not be adequate for a time series of meteorologically driven flows.

Data sets analyzed by Strecker *et al.* (1992) and extended by Carleton *et al.* (2001) include about half natural wetland systems that receive stormwater. Strecker *et al.* (1992) in fact found that natural wetlands performed somewhat better than constructed wetlands, but the use of natural wetlands is not encouraged because of real and perceived negative impacts (U.S. EPA, 1993e). The median areal rate coefficients reported

**TABLE 14.5**
**Suspended Solids Reduction in Constructed Urban Runoff Treatment Wetlands**

| Name | Location | Reference | WWAR Area Ratio (%) | HLR (cm/d) | Reduction (%) |
|---|---|---|---|---|---|
| Crookes | Australia | Raisin *et al.* (1997) | 0.1 | 21.83 | 12 |
| Mays Chapel | Maryland | Carleton *et al.* (2001) | 0.6 | 5.55 | 11 |
| Shop Creek | Colorado | Carleton *et al.* (2001) | 0.6 | — | 25 |
| Franklin Farms | Virginia | Carleton *et al.* (2001) | 0.8 | 17.16 | 62 |
| Lake Munson | Florida | Maristany and Bartel (1989) | 1.1 | 5.19 | 93 |
| Slovenia Highway | Slovenia | Bulc and Sajn Slak (2003) | 1.1 | — | 74 |
| DUST Marsh | California | Meiorin (1989) | 1.8 | — | 64 |
| Crestwood | Virginia | Carleton *et al.* (2000) | 2.4 | 3.69 | 58 |
| Greenwood | Florida | McCann and Olson (1994) | 2.5 | 2.57 | 68 |
| Queen Anne | Maryland | Carleton *et al.* (2001) | 3.8 | — | 65 |
| Clear Lake | Minnesota | Carleton *et al.* (2001) | 4.9 | 1.71 | 76 |
| Tampa Office Pond | Florida | Carleton *et al.* (2001) | 5.1 | 8.16 | 55 |
| West Lafayette | Indiana | Harbor *et al.* (2000) | 6.3 | — | 75 |
| Lake McCarrons | Minnesota | Carleton *et al.* (2001) | 6.6 | 7.38 | 83 |
| Hidden River | Florida | Carr and Rushton (1995) | 19.5 | 1.04 | 86 |
| Elbow Valley | Calgary | Amell (2004) | — | — | 72 |
| Hallam Valley Low | Australia | Wong *et al.* (2006) | — | 400 | 94 |
| Hallam Valley High | Australia | Wong *et al.* (2006) | — | 220 | 94 |
| Kaohsiung | China | Kao *et al.* (2001a) | — | 7.10 | 37 |
| Villanova | Pennsylvania | Rea (2004) | — | 8.16 | 70 |
| **Median** | | | | **7.10** | **68** |

*Note:* All are FWS except for Slovenia, which is HSSF.

by Carleton *et al.* (2001) for FWS constructed gravity-fed systems (N = 9) are

- Total Phosphorus: 8.3 m/yr
- Ammonia: 5.0 m/yr
- Nitrate: 6.7 m/yr

These are very low compared to the *k*-values reported for continuous-flow systems in the preceding chapters. Individual events are afforded much better treatment.

Carleton *et al.* (2001) also suggest that wetland-to-watershed area ratio (WWAR) may be used as a predictor of performance. Their regression equations are developed for small data sets, including natural wetland systems, and have low correlation coefficients ($R^2 \approx 0.15$–$0.45$). Within the range $0 < WWAR < 0.1$, the data scatter was particularly large. Regressions of the data in Table 14.5 are unsatisfactory; in general, the fractions remaining increase slightly with WWAR.

Duncan (1998; as referenced and reported by Wong *et al.*, 1999) suggested the use of regression equations for purposes of sizing. Relations are based on analysis of subsets of 76 stormwater wetland systems, presumably including pumped systems, inclusive of those in the United States and complemented by systems in Australia. The proposed empirically derived relations are

$$\text{TSS Percent Remaining} = 78q^{0.33}C_i^{-0.49} \quad R^2 = 0.80 \qquad (14.8)$$

$$\text{TP Percent Remaining} = 12q^{0.44} \quad R^2 = 0.71 \qquad (14.9)$$

$$\text{TN Percent Remaining} = 14q^{0.43} \quad R^2 = 0.78 \qquad (14.10)$$

where

$C_i$ = inlet concentration, g/m$^3$
$q$ = hydraulic loading rate, m/yr

The range of loading rate data for this regression is approximately $2 < q < 200$ m/yr, and the percents remaining ranged from 10 to 100%; about 30 systems were involved in each regression.

### Agricultural Stormwater

The acquisition of knowledge about wetland treatment of agricultural runoff has often been spurred by regional water quality problems as well as by the slow growth of experiences on small-scale systems. Nutrients from agriculture have been implicated in eutrophication of marine environments (nitrogen) and freshwater (phosphorus) environments. Agricultural runoff is episodic by nature, and there is an extensive literature on predicting the amounts of water and particulate matter that leaves the fields. However, fertilization practices dictate the availability of nutrients to be washed off the fields, and hence the patterns of loading are different from those in urban or industrial settings. Non-point source (NPS) pollution from agriculture may occur when nutrients are applied at rates greater than crops can utilize or when timing of nutrient applications occurs in close proximity to heavy rains (Stone *et al.*, 2003).

A fraction of the fertilizers applied to fields is unavoidably lost to runoff and shallow groundwater. Tile drainage systems collect subsurface waters and vent them to receiving streams, where their nitrogen content, together with phosphorus, may cause problems. Wetlands are being used at various points in the agricultural landscape, corresponding to drainage areas ranging from individual fields (U.S. Department of Agriculture, 1991; Tanner *et al.*, 2003), to small-order streams (Stone *et al.*, 2003), to large regional landscape units of thousands of hectares (Reilly *et al.*, 2000). Some of these receive pumped water at relatively constant rates, and these have been included under the discussion of continuous flow (but possibly variable flow) wetlands. Those that receive water as a result of meteorological events are considered here.

### Target: Nitrogen

Nitrogen compounds are among the principal pollutants of concern in fresh and marine waters because of their role in eutrophication, their effect on the oxygen content of receiving waters, and their potential toxicity to aquatic invertebrate and vertebrate species. For example, the nitrogen content of the streams and rivers of the midwestern United States is of particular importance at this point in history because of hypoxia in the Gulf of Mexico, together with the associated ecological and economic consequences (Diaz and Solow, 1999). Both point and nonpoint sources contribute to the nitrogen content of waters within the Mississippi River drainage basin. About 60% of the waterborne total nitrogen is in the form of nitrate (Goolsby and Battaglin, 2000).

The source of the nitrogen is about two thirds from agriculture and one third from other sources, including urban runoff, atmospheric deposition, and point sources. The result is a surface-water nitrate concentration, in the upper portions of the basin, of about 4 mgN/L.

Federal and state officials have agreed on an action plan that includes a component intended to promote restoration and enhancement of natural systems for nitrogen retention and denitrification (U.S. EPA, 2001a). The premier natural system that has the capability to effectively remove nitrate from surface water is the free water surface wetland, based simply on its ability to place contaminated water in intimate contact with the biogeochemical cycle that removes nitrogen. More than half of the presettlement wetlands in the upper Mississippi River basin have been lost to drainage (Dahl, 1990). It is therefore synergistic to restore wetlands that are positioned to effectively function for nitrate reduction. Considerable attention has therefore been devoted to constructed and restored wetlands for nitrogen control in the midwestern United States (Kovacic *et al.*, 2000; Hey, 2002; Mitsch *et al.*, 2005; Hey *et al.*, 2005; Kadlec, 2005b). Similarly, the Chesapeake Bay in the eastern United States is impacted by agricultural runoff from farms in Maryland (Whigham *et al.*, 1999; Jordan *et al.*, 1999). Studies on seven constructed

(restored) wetlands demonstrated that nutrients could be removed from field runoff. Wetland systems have also been implemented for runoff control from other row crops, such as potatoes (Wengrzynek and Terrell, 1990; U.S. Department of Agriculture, 1991; Higgins *et al.*, 1993).

The problem of excessive nitrogen runoff has also been identified as detrimental to the Baltic Sea. The role of wetlands in southern Sweden has been thoroughly explored. Data were acquired on eight constructed wetlands and used to extrapolate the effects of 40 wetlands of total area 92 ha on the runoff from a 22,400-ha catchment (Arheimer and Wittgren, 1994; 2002). In Finland, retention performance of constructed wetlands has been studied at four agricultural study sites (Puustinen and Koskiaho, 2003). In Norway, field studies at five sites were conducted over several years, focusing on nutrients and sediments (Braskerud *et al.*, 2000; Braskerud, 2001b; 2002a; 2002b; 2003).

Rural settings often contain pastures, which form a source of nutrients and TSS to the drainage waters. FWS wetlands are a convenient method of intercepting some of the nutrient loads, and have been implemented in Australia (Raisin *et al.*, 1997) and New Zealand (Tanner *et al.*, 2003; Tanner *et al.*, 2005b). Typical on-farm systems are not large (Figure 14.8).

### Target: Phosphorus

Although phosphorus has also been studied in the various small-scale studies discussed previously, much of the research on agricultural phosphorus removal in treatment wetlands has originated from the Everglades protection projects of South Florida. The receiving Everglades ecosystem is conditioned by very low total phosphorus, in the range of 6–12 μg/L. Even modest amounts of field runoff phosphorus pose a threat to that ecosystem (Davis, 1994). Tests were performed on dozens of mesocosm tanks, thirty 0.2-ha test cells, and four 2-ha field scale wetlands. In parallel, a 1,500-ha, five-cell wetland, the Everglades Nutrient Removal Project (ENRP) was operated as a research device. These FWS systems variously contained emergent vegetation, submerged plants, or algae, and testing was conducted over the period 1994–2005. A summary of results is given by Walker and Kadlec (2005). Details of ENRP results were presented in a special issue of *Ecological Engineering* (Vol. 27, Issue 4, 2006). Although these studies were directed toward the design of pumped event-driven systems, the trials were in general conducted at conditions of steady flow. Plans to study event operation were abandoned because the full-scale wetlands were placed in operation and became the study platforms.

There are presently six Stormwater Treatment Areas (STAs) in operation, aggregating over 16,500 ha and treating an annual average flow of $4.4 \times 10^6$ m$^3$/d (annual average HLR = 2.7 cm/d). These FWS wetlands have been in operation for periods of two to eight years and receive episodic deliveries of pumped inflows, occasioned by the flat topography of South Florida. An example of the flows and total phosphorus concentrations over the life of STA6 is shown in Figure 14.9. Summaries of performance may be found in annual reports (e.g., SFWMD, 2006).

### Industrial Stormwater

The stormwater runoff flows and quality from "industrial" facilities cannot be generalized because those facilities are quite different in the chemicals that find their way into runoff. A broad definition of industrial is taken here, excluding urban and agricultural runoff. This category includes inorganic sources of nitrogen, such as nitrates and ammonia from fertilizer manufacturing plants (TCI, 2005), and urea from airport de-icing (Thoren *et al.*, 2004).

The Simplot engineered FWS wetland (TCI, 2005) was designed to intercept, retain, and treat previously uncontrolled, meteorologically driven, nitrogen-contaminated groundwater forced to the surface in wet weather conditions and otherwise



**FIGURE 14.8** A constructed FWS wetland system treating tile drainage from a pasture in Toenepi, New Zealand. Wetlands are incised to capture tile drainage by gravity.

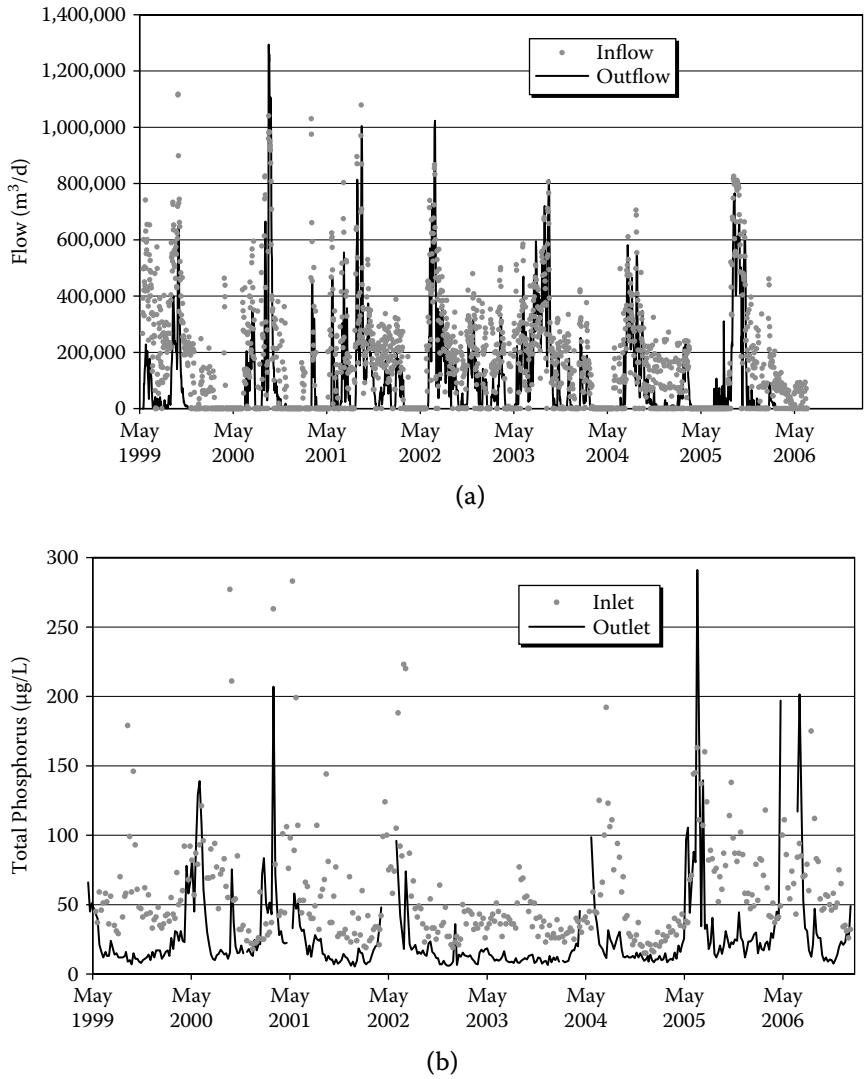



**FIGURE 14.9** Flows (a) and TP concentrations (b) entering and leaving STA6 in Florida. The average inflow rate was 161,000 $m^3$/d; the average outflow was 103,000 $m^3$/d. The arithmetic mean inlet TP concentration was 58 µg/L, and 25.2 µg/L at the outlet.

contributing nutrients to the Assiniboine River. A 4-cell, 22-ha constructed wetland was implemented in Autumn 2003, near Brandon, Manitoba. Approximately 100,000 $m^3$ of nutrient-rich water entered the wetland in 2004, over the April-through-November operating period, corresponding to an annualized HLR of 0.19 cm/d (0.28 cm/d instantaneous). Flows varied by a factor of ten, depending upon rainfall. Inlet concentrations were 400 mg/L total nitrogen (TN) and 165 mg/L ammonia nitrogen, and were reduced by 39% for total nitrogen and 82% for ammonia. The system was zero discharge in 2004, losing 460 mm of water. This corresponds to design net *ET* of 500 mm in a typical eight-month operating period.

The 18-ha FWS Kalmar Dämme wetland was built near Kalmar Airport in southeast Sweden (Thoren *et al*., 2004). Urea was used as the runway de-icing agent, and annually contributed about 41 metric tons of total nitrogen to receiving waters during 1998–2001. Airport runoff joined agricultural runoff from a 48-$km^2$ catchment. The wetland removed 17% of the TN load from all sources, but removal of urea was estimated to be 38%, based on source allocation. Winter removals were much lower than summer.

These two examples serve to illustrate the applicability of constructed wetlands to controlling nitrogen from sources other than municipal and animal wastewaters, or urban and agricultural runoff. Both represent high nitrogen loadings, which were treated to a good degree.

## Batch Systems

Not all event flows are caused by meteorology: the introduction of water to a treatment wetland may be caused solely by a human decision. That is the case with batch-operated systems, in which water is added episodically and drained episodically. If this sequence occurs on a rapid cycle, then there is no significant period of batch holding, as would be the case for alternative of reciprocating wetlands

(Behrends *et al.*, 2001). However, if the holding time is long, then the mode would be termed sequential batch operation. The water may be simply held without flow during the batch time, or it may be continuously recycled (Sikora *et al.*, 1995a). Therefore, like the interevent period of storm-driven wetlands, batch operation is an option for treatment wetlands. The operational advantage is that water may be held until a specific water quality objective is met. The disadvantage is that some form of storage is necessary if the batch mode is to be used to treat continuous flows.

Although it is often assumed that time in a batch system is equivalent to transit time in a continuous-flow system, this is a perilous assumption for wetlands. There are several reasons why these are not equivalent.

- *Hydraulics.* Continuous-flow systems display a distribution of detention times, with different elements of water spending different lengths of time in the flowing wetland. In a batch system that is filled and drained very quickly, every element of water spends exactly the same time in the ecosystem. The result is that the batch system in effect reflects a circumstance strongly akin to plug flow. Consequently, only the distribution of reaction rate coefficients attributable to mixtures will affect the apparent rate of disappearance of a lumped pollutant category (see Chapter 2).
- *Sorption.* In continuous-flow systems, the substrate is exposed to a relatively constant water phase contaminant concentration, which causes the sorption capacity to be equilibrated and plays no further role. In a batch system, the sorbed pollutants may be used up by microbial and vegetative processes, along with the water-phase materials. Therefore, at the end of a batch, which is also the beginning of the next, the sorption sites may be empty and capable of immediate removal of pollutants to sorption storage. This phenomenon was reported by Sikora *et al.* (1995a).
- *Vegetation.* The environment for vegetation is different in the batch mode, with diffusion and transpiration flows providing the transfer of dissolved substances, such as oxygen and nutrients. In the flow mode, advective processes are present and may dominate. Stein *et al.* (2003) conducted microcosm studies on batch-loaded planted gravel bed systems and compared them to accompanying flow through systems. There were differences attributable to plants, including more pronounced seasonal effects and larger differences among plant species in the batch mode. The result was superior nitrogen and phosphorus removal in the batch mode in all seasons and from all conditions studied.
- *Microbes.* Microbial populations respond to their environment and evolve accordingly over a period of time. In a flow through system, there are gradients in water quality from inlet to outlet. Over the course of time, microbial populations adapt to different conditions near the inlet and outlet. Therefore, the water "sees" different types of microbes during its travel through a flowing system, which are the result of a stable growth–death balance at each location. In contrast, the microbes in a batch system undergo a transient in their population and relative abundance, i.e., they are in a growth phase.

Despite these differences in site characteristics, there is typically a near-exponential decline with time of pollutant concentrations in batch wetlands, often with a residual concentration. This is seen at the microcosm level (e.g., Gale *et al.*, 1993; Van Oostrom and Russell, 1994), the mesocosm level (e.g., Burgoon, 1993), and at field scale (Lakhsman, 1981; Sikora *et al.*, 1995a; Kadlec, 2001a). Declines in contaminant concentration are well described by the *k-C** model using the exponential decay because of the lack of detention time distribution (DTD) effects (Figure 14.10). However, the rate coefficients in batch systems may differ considerably from those determined under flow through conditions. For instance, Burgoon (1993) found a factor of two difference, with higher rate coefficients for CBOD for batch operation. Kadlec (2001a) also found higher values for phosphorus *k*-values for batch operation.

Experiences on full-scale batch systems are limited. Studies were conducted on three batch systems at Humboldt, Saskatchewan, in which lagoon waters were treated in wetlands utilizing a fill-hold-drain strategy (Lakhsman, 1982). In general, BOD was reduced from 90 mg/L to below 20 mg/L in four days, total Kjeldahl nitrogen (TKN) from 20 to below 3 mg/L in 20 days, and phosphorus from 12 to below 3 mg/L in 20 days. Given an operating depth of 0.3 m,

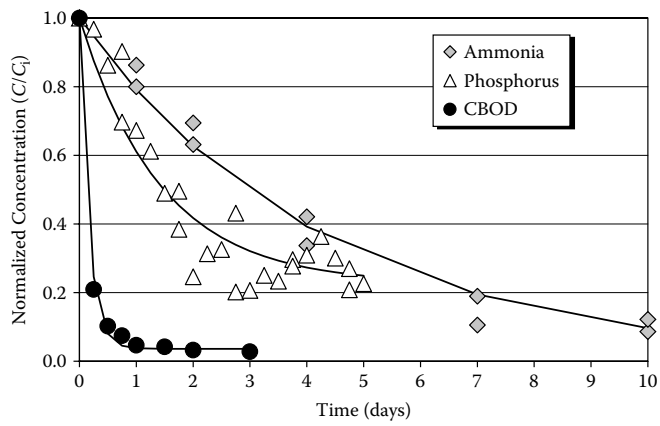



**FIGURE 14.10** Decline of pollutant concentrations in batch systems. Ammonia data is from FWS microcosms of Gale *et al.* (1993). Phosphorus data is from an FWS field-scale wetland of Kadlec (2001a). CBOD data is from SSF mesocosms of Burgoon (1993). All fit lines are for the *k-C** model using an exponential decay.

the equivalent hydraulic loading to reduce BOD was about 7 cm/d, but reduction of nutrients needed about 1.5 cm/d.

The Oak Hammock, Manitoba, headquarters and visitor center of Ducks Unlimited, Canada, near Winnipeg, utilizes a lagoon batch wetland system to treat domestic wastewaters (Pries, 1994). Waste enters unvegetated lagoon one, which functions as a facultative pond. It then travels to a second lagoon with partial vegetative cover of cattails, and finally to a FWS cattail wetland and then to receiving waters (also a marsh). About once per month, the water is drained from the third cell and it is refilled from cell two, which is in turn is filled from cell one. The system stores water during the winter. The discharge from the three-cell system averages 20 CFU/100 mL fecal coliforms, 0.37 mg/L total phosphorus, 1.3 mg/L TKN, and 0.2 mg/L oxidized nitrogen. The system is very lightly loaded, with the summer discharge totaling about 1 m/yr on less than 1 ha.

There is no large-enough database from which to forecast the behavior of batch systems. Ultimately, it would be desirable to be able to compare the long-term average performance of sequential batch operation to that of continuous flow.

### Combined Sewer Overflow (CSO)

In many communities in Europe and North America, there is but one sewer collecting domestic wastewater and storm flows. Wastewater treatment facilities are usually designed to treat flows up to but not including the storm flow. During large rain events, the excess is bypassed, which is termed a combined sewer overflow (CSO). One of the options for treating these event overflows is use of treatment wetlands. Several types of wetlands have been constructed for this purpose, and they are all event driven.

An FWS system to treat CSO was implemented at Houten-Oost, Netherlands, in 1984 (Lageveen and Waardenburg, 1985; van Oorschot, 1990). The CSOs are directed to a series/parallel network of FWS wetlands containing bulrushes (*Scirpus lacustris*) and common reeds (*Phragmites australis;* Figure 14.11). The wetlands comprise 1.5 ha and operate at a depth of about 0.5 m. Discharge from the wetlands is pumped to a cascade aerator and then to a canal. A brief monitoring study during the summer of 1987 provided data on two overflow events, both completely contained within the wetlands. Water quality improved rapidly during storage in the wetlands.

Severn Trent Water built many HSSF CSO systems in the mid-1990s in the United Kingdom. All were based upon dry-weather treatment in rotating biological contactors (RBCs), which bypassed flows above about six times the dry weather flow (Green and Martin, 1996). The bypassed water was treated in reed beds. In general, these systems provide significant improvement to the CSO waters (Figure 14.12). Twelve of these systems were surveyed in 2003–2004 and found to be producing effluent concentrations well below the regulatory levels (Rousseau *et al.*, 2005a). Varying degrees of sludge accumulation, weed growth, surface blinding, and unequal flow distribution were found, but these seemed to have only minor effects on treatment performance.



**FIGURE 14.11** The CSO treatment wetland at Houten-Oost, Netherlands.

A vertical flow design is in use for several stormwater and CSO systems in the state of Hessia in Germany (Frechen *et al.*, 2004; Uhl and Dittmer, 2005; Dittmer *et al.*, 2005). These are underdrained, planted basins that fill with about 1 m of water, which is then drained through about 1 m of sand or soil. The drainage from these retention soil filters (RSF) is controlled at 0.09–2.7 $m^3/d$ per $m^2$ of filter area. Design is for 30–40 m/yr total hydraulic load. The performance at Fulda, Germany, over a 23-event, two-year period was more than 90% removal of chemical oxygen demand (COD), BOD, TSS, and ammonia, and 70–90% of phosphorus.

## 14.3 TSS IN EVENT-DRIVEN WETLANDS

Event-driven treatment wetlands utilize the same suite of TSS processes as continuous-flow systems, but there are important differences in the mode of operation. In contrast to continuous-flow wetlands, there is a greater need to account for the flow and chemical dynamics. Often, the source-water TSS is unknown. Performance data acquisition is difficult because flows occur during storm conditions, and close-interval sampling is needed to define the course of a single event. Internal to the wetland, processes operate on a flow-and-stop time sequence. The moderately stable gradients established by steady flows may no longer be present. The between-event periods are close to batch operation. Small events may not completely displace the antecedent water in

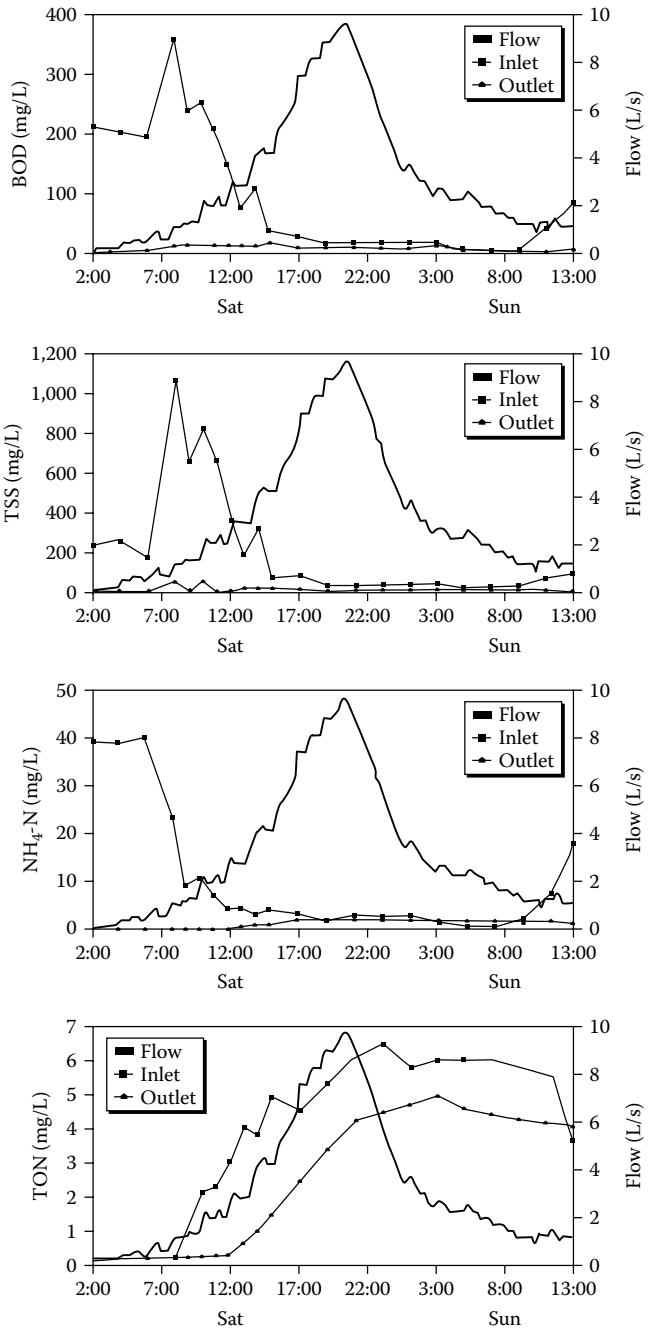



**FIGURE 14.12** Flow and concentrations in a CSO reed bed at Lighthorne Heath, Warwickshire, United Kingdom, in a November 1993 storm event. (From Green and Martin (1996) *Water Environment Research* 68(6): 1054–1060. Reprinted with permission.)

the wetland, but large events may replace it many times over. The flow velocities accompanying an event may be large and may cause resuspension.

## Dynamic Responses

The dynamics of both water and solids influence the performance of event-driven wetlands. If the event period is long compared to the detention time of the wetland under the event flow, then the first part of the event completely flushes the antecedent water, and the outflow concentration ramps up to the level dictated by the sustained event flow (Figure 14.13). After the cessation of event flow, the wetland outflows and outlet concentrations ramp back down to nonflow conditions. The majority of the treatment occurs during the period of high flow, and the expulsion of the highly treated antecedent water contributes little to overall performance during the event.

In contrast, if the event period is short compared to the detention time of the wetland under the event flow, then the event does not completely flush the (highly treated) antecedent water. Ramp-up of the exported solids load starts to occur but stops far short of a new steady state determined by the inflow (Figure 14.14). Ramp-down of the mass export is primarily due to the lower flows; TSS remained elevated in the immediate after-event period.

Data acquisition for individual events involves close-interval sampling, with numerous samples required to define an individual response. Consequently, studies akin to those at Des Plaines are few in number. It is easier and less costly to use flow-proportional autosampling, but the result is then restricted to the EMC for the event.

A second aspect of event wetland dynamics is the variability from event to event during the course of time. Defining such patterns is also a data-intensive effort, which has been accomplished in only a few instances (e.g., Carr and Rushton, 1995). Those researchers examined the water quality from a system for 83 storm events over a period of 30 months (Figure 14.15). The inflows to the Hidden River urban stormwater wetland followed the pattern of the wet and dry seasons for this Florida site, and it functioned with zero discharge in the dry months. Incoming TSS concentrations were quite variable during the year, but the outflow displayed spikes primarily during the dry season, under no-outflow conditions.

## Intersystem Performance

As for continuous-flow systems, there is a sizeable database of stormwater wetland performance. In the United States, this database originated with the work of Strecker *et al.* (1992) and Schueler (1992), and grew through continued efforts of Godrej *et al.* (1999). Table 14.5 shows that there is no clear trend for constructed urban stormwater wetlands with the WWAR if the ratio is greater than 1%. The median reduction is about 70%. In contrast, agricultural runoff does not display any trend at all with loading or WWAR (Table 14.6). That is because field runoff contains soil particles from the fields that can contain a large fraction of large and heavy mineral particles, which settle readily. Median reduction is 57%, but that is also achieved by very small wetlands treating tilled field runoff.

In contrast to continuous-flow system data, event systems have much greater variability in design. Loadings are extremely variable, as are the geometries and arrangements of deep pools, shallow marsh zones, and interconnecting channels.

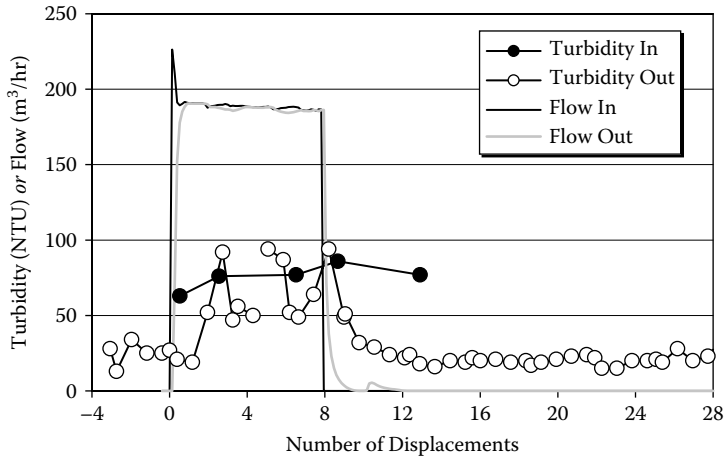



**FIGURE 14.13** Change in effluent turbidity for a 2.5-day pumping event for Des Plaines wetland C1. The nominal detention time for this pumping rate was 0.32 days. (From unpublished data.)

Therefore, the intrasystem database gives an impression of what has been accomplished, rather than what might be accomplished by "optimal" designs. Percent mass removals are high in many cases, with 60% of the U.S. data for stormwater wetlands of both kinds showing greater than 60% removal (Figure 14.16).

As noted in the discussion of urban stormwater systems, a regression has been proposed that relates TSS effluent concentrations to hydraulic loading and inlet TSS (Equation 14.8). That regression is entirely consistent with the intersystem continuous-flow FWS wetland data (see Figure 7.18 in Chapter 7) and may also be used to represent continuous-flow systems.

## 14.4 PHOSPHORUS IN EVENT-DRIVEN WETLANDS

Periodic runoff events from fields in agricultural production, or from streets and parking lots, convey phosphorus to receiving water bodies. FWS treatment wetlands have proven to remove phosphorus from event flows as well as from steady flows. Small pulses are very well treated, but those of longer duration receive much less phosphorus removal.

### Flow Pulses

The dynamics of tracer and phosphorus removal and export were studied at the Des Plaines River wetland demonstration site. Wetland pumping events were monitored for all hydrologic variables, including pumping, rain, storage change, and outflow. Tracer input and response were monitored at inflow, outflow, and interior stations. Phosphorus was measured at high frequency (every six hours) for the outflows, and at lower frequency for inflows and interior stations. The events were isolated in time, with sufficient interevent spacing to allow complete equilibration before the subsequent event. A calibrated dynamic water mass balance was developed as the framework for interpreting results (Kadlec, 2001a).

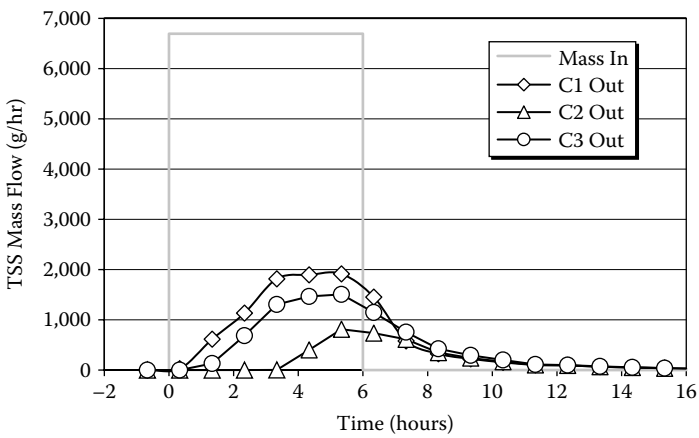
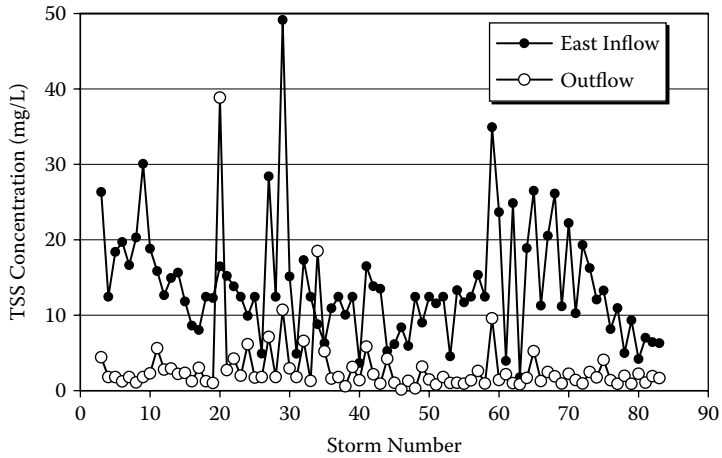
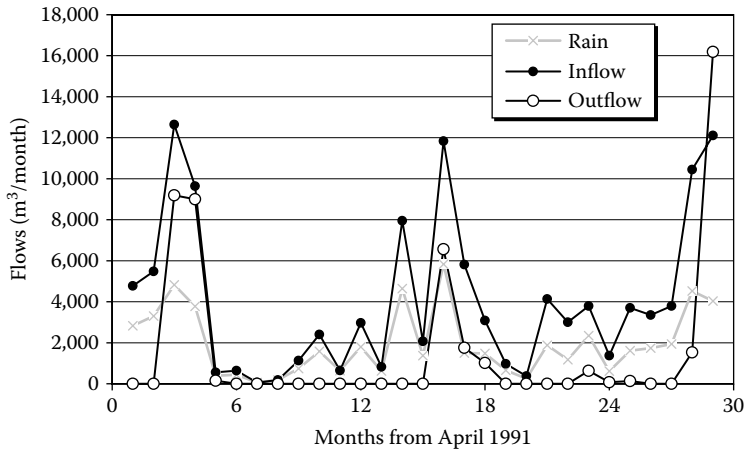



**FIGURE 14.14** Mass inputs and outputs for three wetlands at Des Plaines, driven by a six-hour pumping event. Nominal detention times at the event flow rate were 7.2, 15.4, and 8.2 hours for C1, C2, and C3, respectively. The wetlands were therefore not completely flushed of initial water. Mass removals were 73, 90, and 79%, respectively. (From unpublished data.)




(a)




(b)

**FIGURE 14.15** Results of a 30-month study of the performance of the Hidden River, Florida, urban stormwater treatment wetland. TSS and flows were monitored for 83 rain events. (Data from Carr and Rushton (1995) *Integrating a native herbaceous wetland into stormwater management*. Report by the Southwest Florida Water Management District, July 1995, Boca Raton, Florida.)

## TABLE 14.6
## Suspended Solids Entering and Leaving FWS Agricultural Runoff Treatment Wetlands

| Name | Location | Reference | WWAR (%) | Mean HLR (cm/d) | Data (years) | Inlet (mg/L) | Outlet (mg/L) | Reduction (%) |
|---|---|---|---|---|---|---|---|---|
| McDowell Co. | North Carolina | Kao *et al.* (2001a) | — | — | 1 | 280 | 88 | 69 |
| ENRP | Florida | Gu *et al.* (2006) | — | 3.10 | 5 | 5 | 2 | 58 |
| Wetland D | Norway | Braskerud (2001a) | 0.03 | 340 | 7 | — | — | 50 |
| Wetland A | Norway | Braskerud (2001a) | 0.06 | 140 | 7 | — | — | 59 |
| Wetland B | Norway | Braskerud (2001a) | 0.07 | 120 | 7 | — | — | 56 |
| Wetland C | Norway | Braskerud (2001a) | 0.07 | 160 | 7 | — | — | 53 |
| Alastaro | Finland | Koskiaho *et al.* (2003) | 0.50 | 12.80 | 2 | 56 | 46 | 18 |
| Flytträsk | Finland | Koskiaho *et al.* (2003) | 3.00 | 3.80 | 2 | 29 | 26 | 12 |
| St. John Valley | Maine | Higgins *et al.* (1993) | 4.80 | 0.67 | 2 | 611 | 11 | 97 |
| Hovi | Finland | Koskiaho *et al.* (2003) | 5.00 | 1.90 | 2 | 530 | 170 | 68 |
| B1 | Maryland | Jordan *et al.* (1999) | 9.29 | 0.59 | 1.3 | 165 | 135 | 18 |
| Foster | Maryland | Jordan *et al.* (1999) | 11.40 | 0.48 | 1.3 | 150 | 35 | 77 |
| B2 | Maryland | Jordan *et al.* (1999) | 13.00 | 0.42 | 1.3 | 110 | 30 | 73 |
| Braun | Maryland | Jordan *et al.* (1999) | 19.17 | 0.30 | 1.3 | 70 | 55 | 21 |
| **Median** | | | | | | | | **57** |

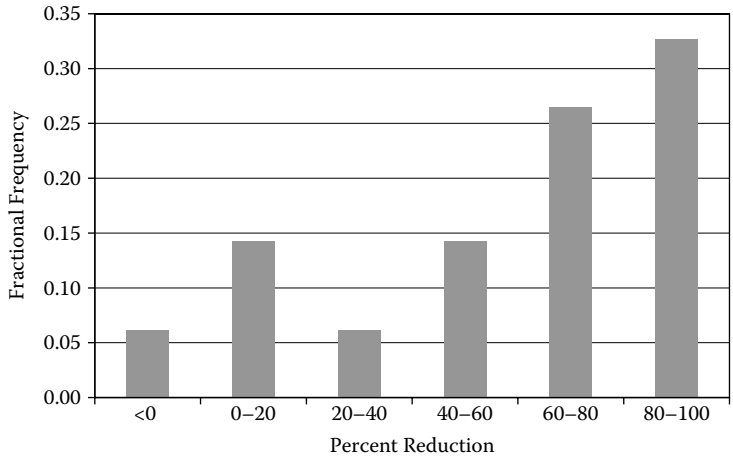



**FIGURE 14.16** Distribution of TSS reductions in stormwater wetlands (N = 49). (Data from Schueler (1992) *Design of stormwater wetland systems: guidelines for creating diverse and effective stormwater wetlands in the mid-Atlantic region*. Metropolitan Washington Council of Governments: Washington, D.C.; Whigham *et al.* (1999) *Nutrient retention and vegetation dynamics in restored freshwater wetlands on the Maryland coastal plain*. Final Report, U.S. EPA Agreement CB-993043-04, Smithsonian Environmental Research Center: Edgewater, Maryland; Carleton *et al.* (2001) *Water Research* 35(6): 1552–1562.)

Internal hydraulics were characterized by tanks in series (TIS) models calibrated to tracer studies. The residence time distribution was describable by five TIS for wetland C1, for instance. Dynamic mass balances were used, in conjunction with a first-order areal removal model, to describe the time sequence of phosphorus concentrations and flows. Parameter estimation produced instantaneous rate coefficients.

A typical response time series and the model representation are shown in Figure 14.17. This two-day pumping event nominally flushed the wetland more than seven times, and the hydraulic loading rate was 87 cm/d. As a result, the outflow concentration during pumping was 213 µg/L, just below that of the incoming water to the wetland, which was 258 µg/L. During the course of pumping, this corresponds to a high removal rate of 14 gP/m$^2$·yr. After the pumping stops, outflow quickly falls to zero, and the phosphorus concentrations in the wetland waters relax down to a background level of about 54 µg/L. Removal is sustained at high levels, with $k$ = 101 m/yr. Prior studies at this site indicated that this high rate of removal is not sustained during continuous-flow operation. The results of this study suggest that great caution must be exercised in transferring steady-flow results to event-driven wetlands. For instance, the calculation approach of Wong and Geiger (1997), using transferred steady-flow rate constants and $C^*$ values, may not be the best descriptor of pulsed treatment wetlands.

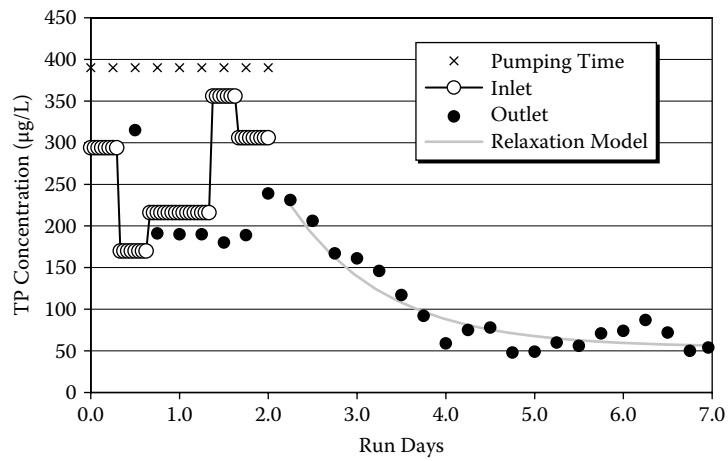



**FIGURE 14.17** Response of wetland C1 at Des Plaines, Illinois, to a two-day pumped-pulse addition. The two-day pumping nominally flushed the wetland 7.2 times. The concentration dropped after pumping to 54 µg/L. The model line has parameters $P$ = 5, $k$ = 101 m/yr, and $C^*$ = 54 µg/L. The incoming river water averaged 258 µg/L TP, and during pumping the average outflow was at 213 µg/L.

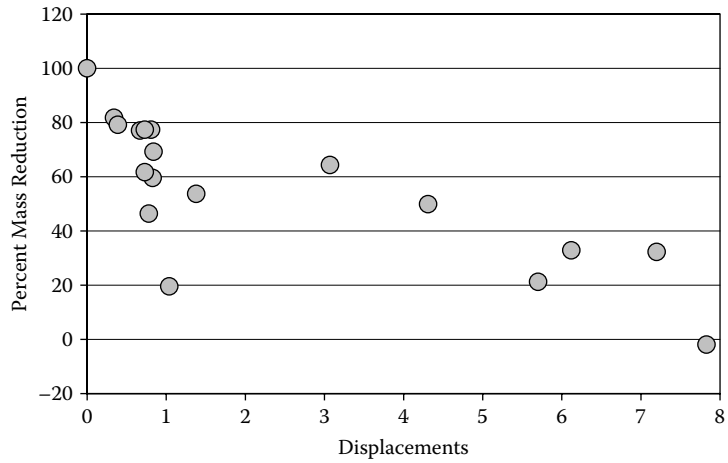



**FIGURE 14.18** The effect of varying displacement on the phosphorus removal efficiency of event-driven wetlands. (From Kadlec (2001a) In *Transformation of Nutrients in Natural and Constructed Wetlands*. Vymazal (Ed.), Backhuys Publishers, Leiden, The Netherlands, pp. 365–392. Reprinted with permission.)

The performance of stormwater wetlands for isolated events is contingent on the flow rate and duration of the event, and the capacity of the wetland. The relevant variable is the number of nominal displacements of the antecedent wetland water that occur, as discussed in the preceding section on capture. When these concepts are applied to the Des Plaines data set, Figure 14.18 results. Phosphorus mass removal decreases as the number of displacements increases. At very low displacement fraction, only a small amount of (treated) antecedent water is "bumped" out of the system. As the number of displacements becomes large, the system approaches continuous-flow performance at the event flow rate.

## Event Sequences

In gravity-fed systems and some pumped stormwater facilities, events are not necessarily isolated but occur in a time series with event-to-event overlap at times. The time series of event treatment may be viewed on a storm-by-storm basis, in which the mass reduction for each event is considered separately regardless of the interevent duration. For instance, the performance of the Hidden River, Florida, wetland was documented for 83 consecutive rain events (Figure 14.19). Because the outlet concentrations depend on event size, wetland storage, and interevent spacing, there is no obvious simple correspondence

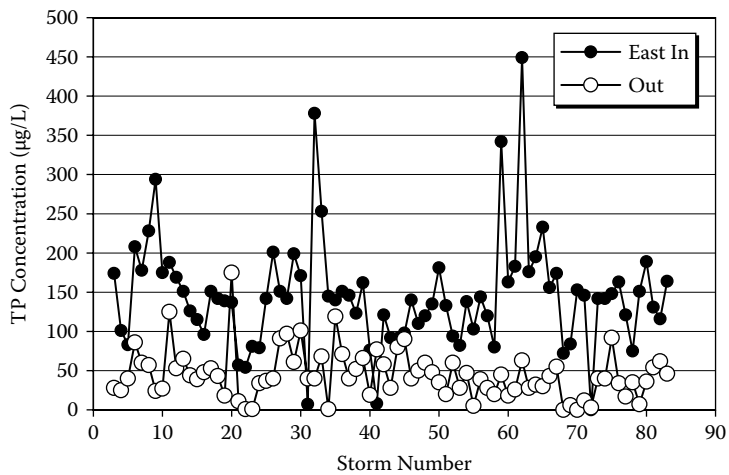



**FIGURE 14.19** Results of a 30-month study of the phosphorus removal performance of the Hidden River, Florida, urban stormwater treatment wetland. Total phosphorus concentrations and flows were monitored for 83 rain events. See Figure 14.15 for the flow time series. (Data from Carr and Rushton (1995) *Integrating a native herbaceous wetland into stormwater management*. Report by the Southwest Florida Water Management District, July 1995, Boca Raton, Florida.)

between outflow concentrations and inflow concentrations and flow rates. Dynamic mass balance modeling is required to analyze this complex set of interactions.

## Urban Stormwater

Over two decades of research directed at stormwater treatment have produced much information on constructed wetland systems to improve runoff quality. Initial surveys tabulated data for natural and constructed systems, receiving either pumped or gravity inflows (1992). Data collections have continued, which build upon previous efforts (Godrej *et al.*, 1999; Carleton *et al.*, 2001). Emphasis here is on constructed or engineered systems that receive gravity flows, because most pumped systems have been discussed under the heading of continuous-flow wetlands. In addition to project reports and published papers, the performance of some treatment wetland systems for stormwater improvement has been catalogued in a Stormwater Best Management Practices (BMP) Database (ASCE, 2003). This database provides access to BMP performance data in a standardized format for roughly 200 BMP studies conducted over the past 15 years, including wetlands. More statistical details on specific systems have been organized in a database by URS (1999).

The amount of phosphorus leaving a watershed during a specific event is often the object of research and predictive models. For example, Brezonik and Stadelmann (2002) found good correlation for event runoff volumes in Minnesota, with rainfall amount, drainage area, and percent impervious area ($R^2 = 0.78$). However, simple correlations for event mean concentrations and event mass loads were weak, and varied strongly with land use.

Urban stormwater wetland studies are hampered by the difficulties involved in tracking flows and concentrations in field situations. In gravity systems, the rainy weather that provides inflows is not conducive to easy sampling. Tracking of just one event requires many inflow and outflow water quality samples and accompanying chemical analyses. Nevertheless, there now exist several very detailed studies, such as that of Carr and Rushton (1995) at the Hidden River, Florida, wetland, which was monitored several times in each of 83 rain events. Similarly, Godrej *et al.* (1999) studied 33 rain events at the Crestwood, Virginia, wetland.

Sample results for reduction of phosphorus from constructed, gravity-fed systems treating urban runoff waters are shown in Table 14.7, which reports information from the summary of Carleton *et al.* (2001) as well as other sources. It must be borne in mind that percent removals, even if based upon load or event-mean concentrations, may reflect a low level of an incoming constituent. Despite such limitations, it is seen that reductions average 40%. WWAR is seen to not be a very good predictor of removals for this data set across multiple sites and conditions, which supports an observation of Schueler (1992).

Equation 14.9 indicates a positive correlation of phosphorus remaining with increases in hydraulic loading rate. This trend is

**TABLE 14.7**
**Phosphorus Entering and Leaving Constructed Urban Runoff Treatment Wetlands**

| Name | Location | Reference | Area Ratio | HLR (cm/d) | % Reduction TP |
|---|---|---|---|---|---|
| Armstrong Slough | Florida | Carleton *et al.* (2001) | 0.3 | 34.65 | 39.7 |
| Mays Chapel | Maryland | Carleton *et al.* (2001) | 0.6 | 5.55 | –7 |
| Shop Creek | Colorado | Carleton *et al.* (2001) | 0.6 | — | 36 |
| Franklin Farms | Virginia | Carleton *et al.* (2001) | 0.8 | 17.16 | 14.9 |
| Lake Munson | Florida | Maristany and Bartel (1989) | 1.1 | 5.19 | 62.7 |
| Highway | Slovenia | Bulc and Sajn Slak (2003) | 1.1 | — | 75 |
| Spring Lake | Minnesota | Carleton *et al.* (2001) | 1.2 | — | –7 |
| DUST Marsh | California | Meiorin (1989) | 1.8 | — | 48 |
| Franklin County | Ohio | Carleton *et al.* (2001) | 2.4 | 10.60 | 16 |
| Crestwood | Virginia | Carleton *et al.* (2001) | 2.4 | 3.69 | 45.9 |
| Greenwood | Florida | Carleton *et al.* (2001) | 2.5 | 2.57 | 61.5 |
| Queen Anne | Maryland | Carleton *et al.* (2001) | 3.8 | — | 39.4 |
| Clear Lake | Minnesota | Carleton *et al.* (2001) | 4.9 | 1.71 | 54 |
| Tampa Pond | Florida | Carleton *et al.* (2001) | 5.1 | 8.16 | 65 |
| Lake McCarrons | Minnesota | Carleton *et al.* (2001) | 6.6 | 7.38 | 41 |
| Hidden River | Florida | Carr and Rushton (1995) | 19.5 | 1.04 | 70 |
| Spring Creek | North Dakota | Carleton *et al.* (2001) | — | 0.22 | 39.6 |
| Elbow Valley | Calgary | Amell (2004) | — | — | 36 |
| Kaohsiung | China | Kao *et al.* (2001a) | — | 7.10 | 70 |
| **Median** | | | | **5.6** | **41** |

*Note:* All are FWS except for Slovenia, which is SSF.

intuitively correct, but the confidence in such correlations cannot be high because of the numerous factors that contribute to variability.

First-order models are suggested to be used to describe event-driven systems, including urban runoff wetlands (Konyha *et al.*, 1995; Wong and Geiger, 1997; Carleton *et al.*, 2001; Wong *et al.*, 2006). These may be used in the time-average sense, in which annual concentrations and flows are presumed to be treated according to the rate coefficients derived for continuous systems (Wong and Geiger, 1997). Because urban stormwater HLRs are close to the HLRs for point-source-driven wetlands, it is reasonable to expect that stormwater wetlands would perform somewhere near the average for the continuous-flow FWS database set. For instance, the median reduction for nitrate in 282 continuous-flow marsh wetlands was 41% at an average HLR = 4.7 cm/d; the median reduction for constructed stormwater marshes, with a median HLR = 5.6 cm/d, was also 41% (Table 14.7).

### Agricultural Runoff

A fraction of the fertilizers applied to fields is unavoidably lost to runoff and shallow groundwater. Tile drainage systems collect subsurface waters and vent them to receiving streams, where their phosphorus content may cause problems. Wetlands are being used at various points in the agricultural landscape, corresponding to drainage areas ranging from individual fields (U.S. Department of Agriculture, 1991; Tanner *et al.*, 2003), to large regional landscape units of thousands of hectares (SFWMD, 2004). The latter are typified by the stormwater treatment areas in South Florida, which strip phosphorus from agricultural and other runoff to protect the Everglades. Systems that receive pumped water at relatively constant rates have been included under the discussion of continuous-flow (but possibly variable-flow) wetlands. Those that receive water as a result of meteorological events are considered here.

The design concepts for agricultural runoff wetlands vary from very high loadings primarily intended for sediment trapping, to very low loadings intended to provide for nutrient removal via microbiological processes and nutrient cycling. Accordingly, WWARs vary greatly (Table 14.8), from less than 0.1–20%. Runoff total phosphorus varies from 0.07–1.15 mg/L on average. It is difficult to draw generalities from data sets such as Table 14.8, but it is clear that very high loading rates do not allow time for interaction with the wetland biogeochemical cycle. It is also evident that wetlands produce or pass through only very low background levels of phosphorus and therefore may be expected to reduce TP to very low levels if sufficient area is provided.

First-order models may be used to describe phosphorus removal in agricultural runoff wetlands over averaged periods of operation (Walker, 1995; Wong *et al.*, 2006; Tanner *et al.*, 2005b). However, it is advantageous to consider system dynamics; and in design, it may be necessary to utilize Monte Carlo simulation of the incoming event train characteristics that may be expected. There is also the question of the transferability of rate coefficients from continuous-flow systems to event-driven systems.

It is also possible to use dynamic first-order modeling to predict phosphorus concentrations in stormwater wetlands (Walker and Kadlec, 2005). This approach is difficult because it is first necessary to measure (calibration mode) or forecast (prediction mode) the inlet time series of flows and concentrations. The prediction mode, in turn, implies the availability of a watershed model that produces the inlet time series.

## 14.5 NITROGEN IN EVENT-DRIVEN WETLANDS

In the pulse-dynamic mode, nitrogen processing and storages might be expected to exhibit different behavior than for continuous flow. In this section, it is shown that microbial processes can respond very quickly to changing circumstances, but the same is not necessarily true for vegetative processes. Internal hydraulics change with time, from batch operation during interstorm periods to extreme through-flow at the times of intense storms.

In almost all stormwater cases, nitrogen loadings are low enough to place them in the category of agronomic control. However, nitrate is often the dominant species, and microbial processing may be of considerable importance. Often, ammonia concentrations are very low, and it is therefore unclear what proportion of stormwater nitrogen is processed microbially rather than by plant uptake.

### Nitrate Pulses

Studies have been conducted in which wetlands receive pulses of nitrate, either by virtue of a concentration spike at constant flow, or by an increase in flow at constant concentration. In general, the response of the wetland is to reduce such new additions without delay.

#### Concentration Pulses and Nitrate Reduction

Bowmer (1987) used bromide and nitrate spikes in an HSSF wetland at Griffith, New South Wales, Australia, which was treating rendering plant effluent. Nitrate was clearly reduced at all points of the response time series, indicating consumption of nitrate in comparison to bromide. Cooper (1994) followed much the same protocols for an FWS wetland near Rotorua, New Zealand. The nitrate response at the wetland outlet showed considerable removal during passage (Figure 14.20). It may be shown that the ratio of nitrate to bromide should follow the disappearance time curve for batch removal because the nitrate travels with a packet of water whose transit time is characterized by bromide. Therefore, the plot of the ratio (Figure 14.20) follows an exponential decline, according to a first-order disappearance. The rate coefficient so determined is 0.038 $h^{-1}$, or in areal terms, 233 m/yr. This is an *intrinsic* value, based upon actual travel time. If the nominal detention time is used, the rate constants will be

**TABLE 14.8**
**Phosphorus Entering and Leaving Agricultural Runoff Treatment Wetlands**

| Reference | Wetland | Location | WWAR (%) | Mean HLR (cm/d) | Data (years) | Total Phosphorus In (mg/L) | Total Phosphorus Out (mg/L) | Total Phosphorus Removal (%) |
|---|---|---|---|---|---|---|---|---|
| Braskerud (2002a) | A | Norway | 0.06 | 170 | 7 | 0.17 | 0.10 | 41 |
| Braskerud (2002a) | C | Norway | 0.07 | 181 | 7 | 0.25 | 0.17 | 32 |
| Braskerud (2002a) | F | Norway | 0.08 | 163 | 4 | 0.22 | 0.17 | 21 |
| Braskerud (2002a) | G1 | Norway | 0.21 | 123 | 3 | 0.43 | 0.27 | 37 |
| Braskerud (2002a) | G2 | Norway | 0.38 | 67 | 3 | 0.43 | 0.24 | 44 |
| Jordan *et al.* (1999) | B1 | Maryland | 9.29 | 0.59 | 1.3 | 1.15 | 0.65 | 43 |
| Jordan *et al.* (1999) | B2 | Maryland | 13.00 | 0.42 | 1.3 | 0.97 | 0.40 | 59 |
| Jordan *et al.* (1999) | Braun | Maryland | 19.17 | 0.30 | 1.3 | 1.15 | 0.63 | 45 |
| Jordan *et al.* (1999) | Foster | Maryland | 11.40 | 0.48 | 1.3 | 0.20 | 0.25 | –25 |
| Kovacic *et al.* (2000) | A | Illinois | 4.00 | 2.19 | 3 | 0.21 | 0.17 | 21 |
| Kovacic *et al.* (2000) | B | Illinois | 5.88 | 1.45 | 3 | 0.12 | 0.08 | 35 |
| Kovacic *et al.* (2000) | D | Illinois | 3.13 | 2.80 | 3 | 0.11 | 0.13 | –18 |
| Koskiaho *et al.* (2003) | Hovi | Finland | 5.00 | 1.90 | 2 | 0.51 | 0.19 | 62 |
| Koskiaho *et al.* (2003) | Alastaro | Finland | 0.50 | 12.80 | 2 | 0.12 | 0.11 | 6.5 |
| Koskiaho *et al.* (2003) | Flytträsk | Finland | 3.00 | 3.80 | 2 | 0.07 | 0.06 | 14.5 |
| Miller *et al.* (2002) | G | Illinois | 0.16 | 16.50 | 4 | 0.060 | 0.060 | –1 |
| Tanner *et al.* (2005a) | Toenepi | New Zealand | 1.00 | 6.90 | 2 | 0.015 | 0.070 | –76 |
| Tanner *et al.* (2003) | Titoki | New Zealand | 2.00 | 19.00 | 1 | 0.075 | 0.015 | 80 |
| Stone *et al.* (2003) | Conestoga | Pennsylvania | 0.10 | 32.20 | 1.5 | 0.055 | 0.036 | 35 |
| Ballaron (1988) | Crookes | Australia | 0.10 | 21.83 | 3 | 0.330 | 0.241 | 27 |
| South Florida Water Management District (unpublished data) | STA1W | Florida | — | 3.95 | 5 | 0.138 | 0.055 | 60 |
| South Florida Water Management District (unpublished data) | STA2 | Florida | — | 3.57 | 5 | 0.075 | 0.018 | 76 |
| South Florida Water Management District (unpublished data) | STA5 | Florida | — | 3.03 | 5 | 0.171 | 0.106 | 38 |
| South Florida Water Management District (unpublished data) | STA6 | Florida | — | 4.57 | 7 | 0.067 | 0.028 | 58 |
| **Median** | | | | | | | | **36** |

*Note:* All are FWS except for Conestoga, which is SSF.

reduced by a factor of 1.8, corresponding to the 56% volumetric efficiency of this wetland. This wetland showed a short circuit path down its central channel (Cooper, 1994). Similar treatment wetland studies have been conducted for constructed and natural HSSF wetlands (respectively, Casey and Klaine, 2001; Burns and Nguyen, 2002). In the case of the constructed wetland, the event tests were not of sufficient duration to clear the wetland of nitrate or tracer, thus making interpretation difficult. In the case of the natural subsurface flow system, there was difficulty in placing wells to sample the outflow. Trujillo (1997) utilized simultaneous bromide and nitrate impulse additions to large FWS mesocosms to determine the hydraulic characteristics and the intrinsic rate of nitrate removal. The short detention times (6–18 hours) implied that plant uptake was not important, and nitrate dose removal was attributed to denitrification. Intrinsic denitrification rate coefficients ranged from 16 to 113 m/yr.

## Flow Pulses and Nitrate Reduction

The dynamics of tracer and nitrate removal and export were studied at the Des Plaines River wetland demonstration site. Wetland pumping events were monitored for all hydrologic variables, including pumping, rain, storage change, and outflow. Tracer input and response were monitored at inflow, outflow, and interior stations. Nitrate was measured at high frequency (every six hours) for the outflows, and at lower frequency for inflows and interior stations. The events were isolated in time, with sufficient interevent spacing to allow complete equilibration before the subsequent event. A calibrated dynamic water mass balance was developed as the framework for interpreting results (Kadlec, 2001a). Internal hydraulics were characterized by TIS models calibrated to tracer studies. The residence time distribution was describable by three TIS for wetland EW3, for instance. Dynamic nitrate mass balances were used, in conjunction with a first-order areal removal model, to describe the

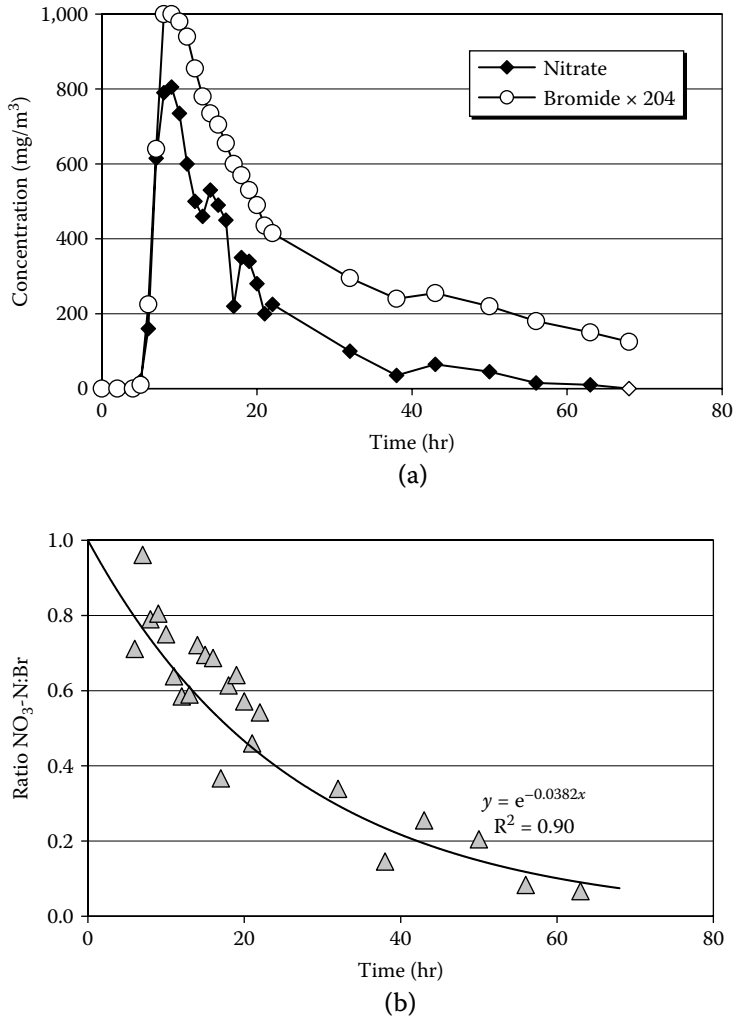



**FIGURE 14.20** Concentrations of nitrate and bromide at the outlet of an *Azolla-Carex* wetland pond at Rotorua, New Zealand. The nominal detention time was 52 hours. Nitrate is scaled to bromide. (Data from Cooper (1994) *Water Science and Technology* 29(4): 141–150.)

time sequence of nitrate concentrations and flows. Parameter estimation produced instantaneous rate coefficients (Werner and Kadlec, 2000b).

A typical response-time series and the model representation are shown in Figure 14.21. For the first day of this two-day pumping event, the outflow builds to equal the inflow, but the outflow concentration is essentially that of the antecedent water in the wetland, about 0.05 mg/L. Both data and the model calibration show that the wetland exports about 0.42 mg/L nitrate, reduced from 1.35 mg/L in the inflow, at the full-flow steady state. However, the early flush of clean antecedent water gives a lesser event-mean outflow concentration of 0.29 mg/L.

The removal rate coefficient is 125 m/yr, but the mass removal is a modest 42 g/m$^2$·yr (116 mg/m$^2$·d) over the duration of the event. It is noteworthy that the wetland response is very fast, both hydraulically and in terms of concentration reduction. Storage in the wetland biomass and sediments apparently is not critical to the response. This is convenient in calculations because only water-phase storages and concentrations suffice to characterize the behavior. However, biomass provides the locus for denitrifiers and their carbon supply.

The performance of stormwater wetlands for nitrate reduction is contingent on the flow rate and duration of the event, and the capacity of the wetland. The relevant variable is the number of nominal displacements of the antecedent wetland water that occur, as discussed in the preceding section on capture. When these concepts are applied to the Des Plaines data set, Figure 14.22 results. Nitrate mass removal decreases as the number of displacements increases. At very low displacement fraction, only a small amount of (treated) antecedent water is "bumped" out of the system. As the number of displacements becomes large, the system approaches its continuous-flow performance at the event-flow rate.

## Ammonia Pulses

It is uncommon that a pulse of ammonia-nitrogen would reach a stormwater treatment wetland, because urban and

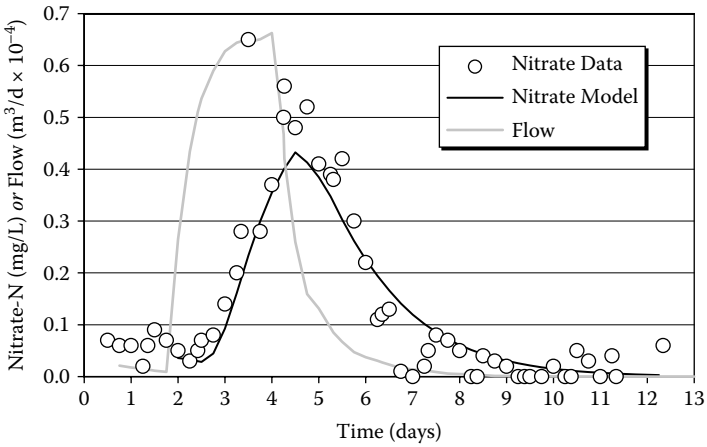



**FIGURE 14.21** Response of wetland EW3 to a two-day flow pulse containing nitrate at 1.35 mg/L at Des Plaines, Illinois.

agricultural runoff do not contain much ammonia. The circumstance arises when pretreatment that includes nitrification fails. For instance, Bavor *et al.* (1995) reported about three such events per year over a four-year period of operation of the Byron Bay, Australia, pretreatment system, some of which reached 20–45 mg/L for brief periods. These were reduced to less than 3 mg/L by the Byron Bay treatment wetlands.

Daily sampling was performed at the Commerce Township, Michigan, treatment wetland, which was preceded by an oxidation ditch pretreatment plant that conducted full nitrification. The mean monthly influent ammonia-nitrogen to the treatment wetland averaged 0.42 mg/L, and the mean monthly effluent ammonia-nitrogen from the treatment wetland averaged 0.35 mg/L over a five-year period. However, these averages were influenced by two WWTP upsets that sent high ammonia to the treatment wetland over 68 days in 2000 and nine days in 2002. These peaked at approximately 13 mg/L and 20 mg/L, respectively. If these months are excluded, the mean monthly influent ammonia-nitrogen to the treatment wetland averaged 0.085 mg/L, and the mean monthly effluent ammonia-nitrogen from the treatment wetland averaged 0.086 mg/L.

The average *k*-value during the period 1999–2003 was 12.0 m/yr, which is close to the median expected value based on other wetlands. The year 2000 event had an event-mean wetland inlet ammonia concentration of 4.1 mg/L, which was reduced to 2.3 mg/L. In the year 2002 event, 5.6 mg/L was reduced to 3.3 mg/L (Figure 14.23). The corresponding ammonia removal rate constant, for 3TIS at the prevailing water temperature of 20°C, was 40 m/yr. This high value, together with the synchrony between the inlet and outlet concentrations, signals a rapid response of the ammonia removal mechanisms within the wetland. Removal may be attributed to microbial processes or to algal uptake. In either case, water-phase storages are adequate for calculations.

## Event Sequences

In many instances, the nitrogen arrives at the stormwater treatment wetland in a complex time series dictated by

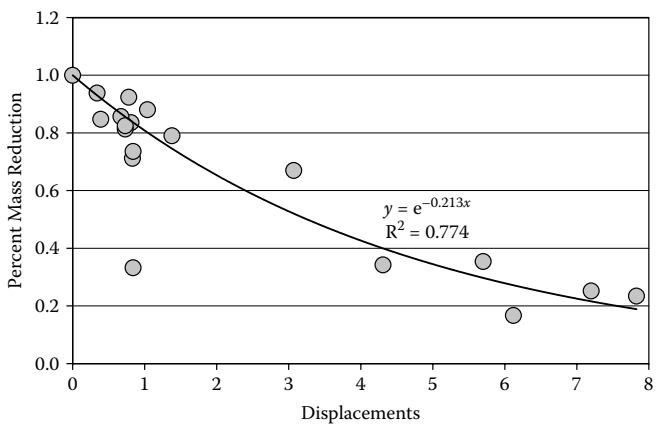



**FIGURE 14.22** The effect of varying wetland water displacement on the nitrate mass removal efficiency of event-driven wetlands at Des Plaines, Illinois. (From unpublished data.)

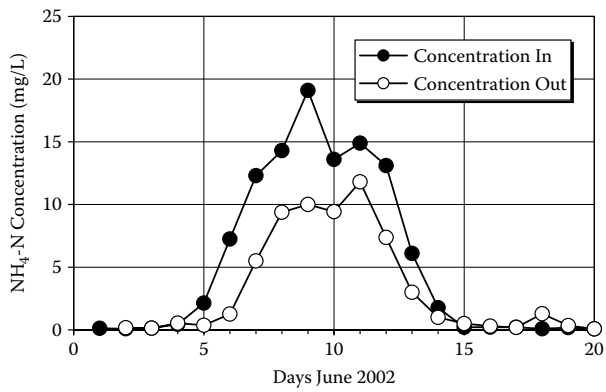



**FIGURE 14.23** Response of the Commerce Township, Michigan, treatment wetland to an ammonia pulse from the wastewater treatment plant in June 2002. (From Commerce Township unpublished daily monitoring data.)

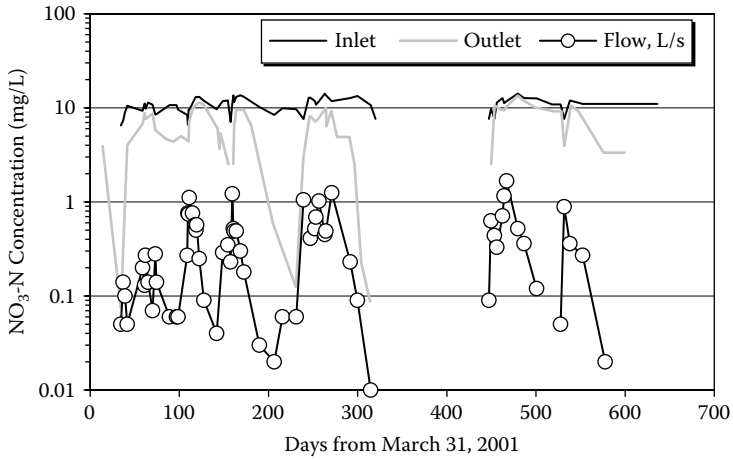



**FIGURE 14.24** Nitrate nitrogen entering and leaving the Toenepi, New Zealand, constructed tile runoff pasture wetland. (Data courtesy C. Tanner.)

watershed characteristics. An example is the constructed wetland at Toenepi, New Zealand, which treats tile drainage from a pasture. In the case of inorganic nitrogen, especially nitrate, it has been found that treatment tracks the flow and concentration behavior of the inflow (Figure 14.24). This suggests that the removal processes adapt very quickly to the changing inlet conditions, and provide nitrate reduction that is well-described by a first-order removal model.

### Nitrogen Reduction in Urban Stormwater

The same information base that was used for phosphorus is in major part also a source of information for nitrogen species. The reader is referred to the phosphorus section of this chapter for a discussion of those sources.

Sample results for reductions in nitrogen species from constructed, gravity-fed urban systems are shown in Table 14.9, which reports information from the summary of Carleton *et al.* (2001) as well as other sources. It must be borne in mind that percent removals, even if based upon load or event-mean concentrations, may reflect a low level of an incoming constituent. Despite such limitations, it is seen that nitrate reduction has the highest median percentages (45%), compared to ammonia (31%) and TN (30%). This is in accord with the higher nitrate rate constants observed in continuous-flow systems, and with low amounts of incoming ammonia and an irreducible background of organic nitrogen, which affects TN reduction. WWAR is seen to not be a very good predictor of removals for this data set across multiple sites and conditions, which supports an observation of Schueler (1992). Hydraulic loading has been suggested as an appropriate correlator of TN reduction, as given in Equation 14.10 (Wong *et al.*, 1999).

First-order models are suggested to be used to describe event-driven systems, including urban runoff wetlands (Konyha *et al.*, 1995; Wong and Geiger, 1997; Carleton *et al.*, 2001; Wong *et al.*, 2006). These may be used in the time-average sense, in which annual concentrations and flows are presumed to be treated according to the rate coefficients derived for continuous systems (Wong and Geiger, 1997). Because urban stormwater HLRs are close to the HLRs for point-source-driven marshes, it is reasonable to expect that stormwater wetlands would perform somewhere near the average for the continuous-flow FWS database set. For instance, the median reduction for nitrate in 72 continuous-flow FWS wetlands was 53% at an average HLR = 7.1 cm/d; the median reduction for constructed stormwater marshes, with an average HLR = 5.4 cm/d, was 45% (Table 14.9). The median plug-flow areal rate constants reported by Carleton *et al.* (2001) for FWS constructed gravity systems (N = 9) were cited above to be 5.0 m/yr for ammonia, and 6.7 m/yr for nitrate. As noted above, these are very low compared to the *k*-values reported for continuous-flow systems.

It is also possible to use dynamic first-order modeling to predict nitrogen species concentrations in stormwater wetlands (Wong *et al.*, 2006). This approach is difficult because it is first necessary to measure (calibration mode) or forecast (prediction mode) the inlet time series of flows and concentrations. In the prediction mode, this in turn implies the availability of a watershed model that can produce the inlet time series. There is also a question of the transferability of rate coefficients from continuous-flow situations. It is possible, if not probable, that the nitrogen processes that occur on an event time scale of hours are different from those that control on a time scale of months. Therefore, it is necessary to determine if different model calibrations are needed for the event-driven case.

### Agricultural Runoff

The design concepts for agricultural runoff wetlands vary from very high loadings primarily intended for sediment trapping to very low loadings intended to provide for nutrient removal via microbiological processes and nutrient cycling.

**TABLE 14.9**
**Removal Performance of Constructed, Gravity-Fed, Urban Stormwater Wetlands**

| System | Location | Wetland Area (ha) | Drainage Area (ha) | Area Ratio (%) | $NH_3$ (%) | $NO_x$-N (%) | TN (%) | Reference |
|---|---|---|---|---|---|---|---|---|
| Clear Lake | Minnesota | 21.4 | 433 | 4.94 | 55 | — | — | Carleton *et al.* (2001) |
| Crestwood | Virginia | 0.070 | 2.90 | 2.41 | 55 | 39 | 22 | Carleton *et al.* (2001) |
| DUST | California | 22 | 1,200 | 1.83 | 10 | 15 | — | Carleton *et al.* (2001) |
| Franklin County | Ohio | 6.1 | 260 | 2.35 | — | — | — | Carleton *et al.* (2001) |
| Franklin Farms | Virginia | 0.125 | 16.2 | 0.78 | −1 | 60 | — | Carleton *et al.* (2001) |
| Greenwood | Florida | 5.24 | 213 | 2.46 | 10 | −13 | −11 | Carleton *et al.* (2001) |
| Lake McCarrons | Minnesota | 2.50 | 38.0 | 6.58 | | 35 | 35 | Carleton *et al.* (2001) |
| Lake Tahoe | California | 0.066 | 1.0 | 1.84 | −53 | 85 | — | Carleton *et al.* (2001) |
| Mays Chapel | Maryland | 0.240 | 39.3 | 0.61 | 22 | 28 | — | Carleton *et al.* (2001) |
| Queen Anne | Maryland | 0.240 | 0.64 | 3.75 | 56 | 55 | 23 | Carleton *et al.* (2001) |
| Shop Creek | Colorado | 1.42 | 223 | 0.64 | — | 21 | 41 | Carleton *et al.* (2001) |
| Spring Lake | Minnesota | 26 | 2,260 | 1.15 | −86 | — | −14 | Carleton *et al.* (2001) |
| Tampa Office | Florida | 0.129 | 2.55 | 5.08 | 39 | 65 | — | Carleton *et al.* (2001) |
| Motorway | Slovenia | 0.009 | 0.75 | 1.13 | 89 | 76 | 80 | Bulc and Sajn Slak (2003) |
| Kaohsiung | Taiwan | 0.120 | — | — | 91 | — | — | Kao *et al.* (2001a) |
| Blacktown, NSW | Australia | 0.450 | 75 | 0.60 | −6 | 16 | 58 | Knoll and Bavor (2004) |
| Calamvale, QLD | Australia | 0.187 | 160 | 0.12 | 50 | 50 | 30 | Greenway and Jenkins (2004) |
| Coorparoo, QLD | Australia | 0.800 | 197 | 0.41 | — | 78 | — | Greenway and Jenkins (2004) |
| **Median** | | | | **1.83** | **31** | **45** | **30** | |

*Note:* All are FWS except for Slovenia and Taiwan.

Accordingly, WWARs vary greatly (Table 14.10), from less than 0.1%–20%. Runoff total nitrogen varies 2–20 mg/L on average and is dominated by nitrate in most crop runoff cases. Ammonia-nitrogen is typically less than 1 mg/L entering the wetlands. However, pasture runoff may contain episodically high organic nitrogen. It is difficult to draw generalities from data sets such as Table 14.10, but it is clear that very high loading rates do not allow time for interaction with the wetland biogeochemical cycle, or for microbial processes. The Norwegian and Pennsylvania data in Table 14.10 exemplify this lack of nitrogen removal performance high loading. It is also evident that wetlands produce or pass through a background level of about 1–2 mg/L of organic nitrogen and therefore should not be expected to reduce TKN or TN to below about 2 mg/L, even at low loadings (high WWAR).

First-order models may be used to describe agricultural runoff wetlands over averaged periods of operation (Stone *et al.*, 2003; Tanner *et al.*, 2005b). However, it is advantageous to consider system dynamics; and in design, it may be necessary to utilize Monte Carlo simulation of the incoming event-train characteristics. There may be a question of the transferability of rate coefficients from continuous-flow systems to event-driven systems.

## 14.6 METALS IN EVENT-DRIVEN WETLANDS

Trace metals are typically present in urban stormwaters but are generally not found in levels of concern in agricultural runoff. Zinc is usually the highest, at up to 150 μg/L, copper at up to 40 μg/L, and lead up to 70 μg/L. Cadmium and chromium are usually below 10 μg/L.

The mechanisms of metal removal in treatment wetlands have been discussed in Chapter 11, and these mechanisms are also operational in event wetlands. It may be surmised that most of the removal of metals is associated with particulate settling, partitioning, and formation of precipitates. The effect of cattails on the bulk concentration and partitioning of metals was found to be minimal in FWS-constructed urban stormwater wetlands (Goulet and Pick, 2001). Surface sediments with and without cattails (*Typha latifolia*) had similar concentrations of total metals (Fe, Mn, Zn, and Cu). Generally, half of the total metal concentration in sediments was in the form of metal oxides, monosulfides, and adsorbed onto organic matter, but half of the total Zn was associated with the persistent organic fraction.

The removals of some common trace metals in stormwater wetlands are listed in Table 14.11. Lead and cadmium

## TABLE 14.10
## Nitrogen Species Entering and Leaving Agricultural Runoff Treatment Wetlands

| | | | | | | Oxidized N | | | Ammonia N | | | Organic N | | | Total N | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Reference** | **Wetland** | **Location** | **WWAR (%)** | **Mean HLR (cm/d)** | **Data (years)** | **In (mg/L)** | **Out (mg/L)** | **Rem. (%)** | **In (mg/L)** | **Out (mg/L)** | **Rem. (%)** | **In (mg/L)** | **Out (mg/L)** | **Rem. (%)** | **In (mg/L)** | **Out (mg/L)** | **Rem. (%)** |
| Braskerud (2002a) | A | Norway | 0.06 | 170 | 7 | 2.28 | 2.29 | 0 | 0.16 | 0.17 | −5 | 0.76 | 0.64 | 16 | 3.21 | 3.10 | 3 |
| Braskerud (2002a) | C | Norway | 0.07 | 181 | 7 | 2.16 | 2.18 | −1 | 0.37 | 0.36 | 3 | 0.99 | 0.81 | 18 | 3.49 | 3.35 | 4 |
| Braskerud (2002a) | F | Norway | 0.08 | 163 | 4 | 0.75 | 0.73 | 3 | 0.07 | 0.07 | −10 | 0.79 | 0.70 | 11 | 1.60 | 1.50 | 6 |
| Braskerud (2002a) | G1 | Norway | 0.21 | 123 | 3 | 2.77 | 2.50 | 9 | 0.68 | 0.65 | 3 | 1.70 | 1.24 | 27 | 5.14 | 4.40 | 14 |
| Braskerud (2002a) | G2 | Norway | 0.38 | 67 | 3 | 2.77 | 2.57 | 7 | 0.68 | 0.65 | 3 | 1.70 | 1.16 | 32 | 5.14 | 4.38 | 15 |
| Jordan *et al.* (1999) | B1 | Maryland | 9.29 | 0.59 | 1.3 | 0.25 | 0.07 | 72 | 0.36 | 0.31 | 14 | 2.52 | 3.10 | −23 | 3.13 | 3.48 | −11 |
| Jordan *et al.* (1999) | B2 | Maryland | 13.00 | 0.42 | 1.3 | 0.78 | 0.01 | 99 | 0.39 | 0.17 | 57 | 2.36 | 1.67 | 29 | 3.53 | 1.85 | 48 |
| Jordan *et al.* (1999) | Braun | Maryland | 19.17 | 0.30 | 1.3 | 0.04 | 0.04 | 0 | 0.17 | 0.14 | 13 | 1.96 | 1.58 | 19 | 2.17 | 1.76 | 19 |
| Jordan *et al.* (1999) | Foster | Maryland | 11.40 | 0.48 | 1.3 | 0.33 | 0.11 | 67 | 0.27 | 0.14 | 48 | 2.71 | 1.29 | 52 | 3.31 | 1.54 | 53 |
| Kovacic *et al.* (2000) | A | Illinois | 4.00 | 2.19 | 3 | 13.00 | 11.20 | 14 | 1.00 | 0.80 | 20 | 0.00 | 0.30 | — | 14.00 | 12.30 | 12 |
| Kovacic *et al.* (2000) | B | Illinois | 5.88 | 1.45 | 3 | 10.30 | 7.40 | 28 | 0.10 | 0.10 | 0 | 0.00 | 0.20 | — | 10.40 | 7.70 | 26 |
| Kovacic *et al.* (2000) | D | Illinois | 3.13 | 2.80 | 3 | 8.50 | 5.90 | 31 | 0.20 | 0.10 | 50 | 0.00 | 0.30 | — | 8.70 | 6.30 | 28 |
| Koskiaho *et al.* (2003) | Hovi | Finland | 5.00 | 1.90 | 2 | 7.90 | 5.14 | 35 | 0.11 | 0.08 | 20 | 1.80 | 1.05 | 41 | 9.80 | 6.27 | 36 |
| Koskiaho *et al.* (2003) | Alastaro | Finland | 0.50 | 12.80 | 2 | 6.80 | 7.07 | −4 | 0.04 | 0.06 | −36 | 1.56 | 1.77 | −14 | 8.40 | 8.90 | −6 |
| Koskiaho *et al.* (2003) | Flytträsk | Finland | 3.00 | 3.80 | 2 | 2.40 | 2.16 | 10 | 0.09 | 0.04 | 55 | 0.61 | 0.65 | −7 | 3.10 | 2.85 | 8 |
| Miller *et al.* (2002) | G | Illinois | 0.16 | 16.50 | 4 | 12.50 | 11.12 | 11 | — | — | — | — | — | — | — | — | — |
| Tanner *et al.* (2005a) | Toenepi | New Zealand | 1.00 | 6.90 | 2 | 11.79 | 8.10 | 31 | 0.08 | 0.16 | −100 | 14.34 | 0.10 | 99 | 26.20 | 8.61 | 67 |
| Tanner *et al.* (2003) | Titoki | New Zealand | 2.00 | 19.00 | 1 | 6.50 | 0.40 | 94 | 0.25 | 0.42 | −68 | 1.10 | 1.80 | −64 | 8.50 | 5.70 | 33 |
| Stone *et al.* (2003) | Herrings | North Carolina | 0.78 | 9.00 | 3 | 6.60 | 2.00 | 70 | 0.50 | 0.70 | −40 | 0.80 | 0.90 | −12 | 7.90 | 3.60 | 54 |
| Ballaron (1988) | Conestoga | Pennsylvania | 0.10 | 32.20 | 1.5 | 20.96 | 19.85 | 5 | 0.08 | 0.08 | 0 | 2.30 | 1.67 | 27 | 23.34 | 21.60 | 7 |
| | **Median** | **(ex. Norway)** | | | | **6.70** | **3.85** | **29** | **0.20** | **0.144** | **13** | **1.70** | **1.16** | **23** | **8.40** | **5.70** | **26** |

*Note:* All are FWS except for Conestoga, which is SSF; Rem. = Removal

**TABLE 14.11**
**Metals Reduction in Constructed Urban Runoff Treatment Wetlands**

| Name | Location | Reference | Percent Reduction | | | | | Area Ratio | HLR (cm/d) |
|---|---|---|---|---|---|---|---|---|---|
| | | | Cd | Cu | Pb | Ni | Zn | | |
| Shop Creek | Colorado | Carleton *et al.* (2001) | — | –15 | — | — | 24 | 0.6 | — |
| Lake Munson | Florida | Carleton *et al.* (2001) | — | –4 | 56 | — | 59 | 1.1 | 5.19 |
| Slovenia Highway | Slovenia | Bulc and Sajn Slak (2003) | 100 | 93 | 100 | — | 98 | 1.1 | — |
| DUST Marsh | California | Meiorin (1989) | — | 31 | 88 | — | 33 | 1.8 | — |
| Crestwood | Virginia | Carleton *et al.* (2000) | 28 | 66 | 75 | — | 29 | 2.4 | 3.69 |
| Greenwood | Florida | McCann and Olson (1994) | 0 | 59 | 60 | — | 69 | 2.5 | 2.57 |
| Tampa Office Pond | Florida | Carleton *et al.* (2001) | — | — | — | — | 51 | 5.1 | 8.16 |
| Lake McCarrons | Minnesota | Meiorin (1989) | — | — | 74 | — | | 6.6 | 7.38 |
| Hidden River | Florida | Carr and Rushton (1995) | 88 | 79 | 83 | — | 84 | 19.5 | 1.04 |
| Kaohsiung | China | Kao *et al.* (2001a) | — | — | 85 | — | 95 | — | 7.10 |
| Sanford | Florida | Harper *et al.* (1986) | 71 | 40 | 55 | 70 | 41 | — | — |
| Norco | Louisiana | Hawkins *et al.* (1997) | — | 33 | — | — | 78 | — | — |
| Aiken | South Carolina | Nelson *et al.* (2004) | — | 83 | — | — | 60 | — | — |
| Orlando | Florida | Kent *et al.* (1997) | — | — | — | 34 | — | — | — |
| Orlando | Florida | Kent *et al.* (1997) | — | — | — | 44 | — | — | — |
| Orlando | Florida | Schiffer (1989) | — | — | — | 25 | — | — | — |
| Irondequoit Creek | New York | Coon *et al.* (2000) | — | — | 29 | — | — | — | — |
| **Median** | | | **71** | **49** | **74** | **39** | **60** | **2.4** | **5.2** |

*Note:* All are FWS except for Slovenia, which is HSSF.

have the highest reductions, with a median of 70–75%. Zinc, copper, and nickel are less effectively removed, with median removals of 60, 50, and 40%, respectively. Stormwater wetlands consistently remove a good fraction of metals as long as the WWAR is greater than about 1%.

The removal of metals implies their storage in the wetland or pond used to treat the stormwater. Sediments build up in treatment systems, and so the concentrations in the sediments reflect both storage and sediment accretion. Contaminants can affect the biota that inhabit these sediments. Ontario defines a lowest-effect level as the 5th percentile of a biological test result distribution, and the severe-effects level as the 95th percentile. Studies in Ontario by Wren *et al.* (1997) have determined the levels found in stormwater system sediments in the Guelph region and compared them to Ontario sediment standards (Table 14.12). Some metals eventually

**TABLE 14.12**
**Metals Accumulation in the Sediments of Stormwater Pond Sediments (mg/kg)**

| Metal | Age 6–8 Years | Age 10–17 Years | LEL | SEL |
|---|---|---|---|---|
| Cd | <1 | 1.4 | 0.6 | 10 |
| Cr | 12.2 | 19.0 | 26 | 110 |
| Cu | 27.8 | 45.7 | 16 | 110 |
| Fe | 10,800 | 13,700 | 20,000 | 40,000 |
| Ni | 8.8 | 13.4 | 16 | 820 |
| Pb | 39.4 | 90 | 31 | 250 |
| Zn | 194 | 300 | 120 | 820 |

*Note:* LEL = lowest effect level, SEL = severe effect level.

*Source:* Data from Wren *et al.* (1997) *Wildlife and contaminants in constructed wetlands and stormwater ponds: current state of knowledge and protocols for monitoring contaminant levels and effects in wildlife*. Technical Report Series No. 269, Canadian Wildlife Service: Burlington, Ontario, Canada.

exceeded the lowest-effect level but not the severe-effect level. The studies of Goulet and Pick (2001) found Fe, Cu, and Zn all below the lowest-effect level at two stormwater wetlands of age four and nine years. Thus, although there are buildups to be expected, the amounts of trace metals appear to generally be below levels that would cause concern.

## 14.7 PESTICIDES IN EVENT-DRIVEN WETLANDS

A wide variety of pesticides are often found in agricultural runoff. For some, the removal rates in wetlands have been determined, as discussed in Chapter 13, but usually in continuous-flow systems. One of the most studied is atrazine, by virtue of its pervasive use, and there is a small body of information on its behavior in event-driven treatment wetlands.

Atrazine transport, sorption, and identity loss were studied at the Des Plaines River constructed wetland site and in accompanying laboratory work. Sorption was effective for soils and sediments, but the more-organic materials, such as litter, showed a stronger affinity for atrazine than the mineral base soils of the wetland cells at Des Plaines (Alvord and Kadlec, 1995; 1996). Atrazine was found to degrade on those sediments according to a first-order rate law, with a half-life of 40 to 90 days. However, degradation was faster on cattail litter, precluding a rate measurement. The estimated half-life on cattail detritus was on the order of five days.

Outflows from the Des Plaines wetland cells contained reduced amounts of atrazine compared to the river water inputs. During 1991, atrazine peaked in the river due to two rain events. Only about 25% of the incoming atrazine was removed in wetland cell EW3, but 95% was removed in wetland cell EW4 due to longer detention time. Either time-averaged or dynamic models, both embodying a nonideal mixing model, fit the data for wetland EW3 (see Figure 13.7 in Chapter 13). The value of $k \approx 14$ m/yr calibrated the models to the data.

Runes *et al.* (2003) conducted field experiments near Portland, Oregon, in a constructed wetland consisting of five sequential cells, each approximately 3 m × 40 m filled with a silt loam, and planted with *Typha latifolia* at >75% coverage. Water came from a container nursery facility and was delivered episodically and dosed with atrazine. Flow events were ranked by intensity (low: <250 L/min, medium: 250–500 L/min, high: >500 L/min) and frequency of the events (low: <6 events/week, medium: 6–9 events/week, high: >9 events/week). Atrazine treatment efficiency at the outlet of the constructed wetland during a seven-day period ranged from 76–82% in 1998 (experiments 1–3) and 83–84% in 1999 (Table 14.13), but zero treatment was found in one monitoring period. For all experiments, deethylatrazine and deisopropylatrazine accounted for 13–21% of the initial application. Hydroxyatrazine was rarely detected in the water.

**TABLE 14.13**
**Percentage Atrazine Removed in FWS Wetland Cells Under Event-Driven Operation**

| Runoff Frequency | Runoff Intensity | Cell 1 | Cell 3 | Cell 5 |
|---|---|---|---|---|
| High | Medium | 3 | 80 | 82 |
| High | Medium | 90 | 69 | 76 |
| Medium | High | 99 | 94 | 80 |
| High | High | 27 | 76 | 83 |
| Low | Low | 55 | 48 | 84 |

*Source:* Data from Runes *et al.* (2003) *Water Research* 37: 539–550.

## 14.8 DYNAMIC MODELING

At a simple level, the dynamic hydrologic tanks-in-series model of Equations 14.3 and 14.4 may easily be extended to include one or more contaminants. The segmented *P*TIS model is continued. The dynamic pollutant mass balance for the first segment, for unsteady state, nonuniform flow is

$$\frac{d(AhC)_1}{dt} = Q_i C_i - Q_1 C_1 - I A_1 C_1 - k A_1 (C_1 - C^*) \qquad (14.11)$$

where

$A_1$ = area of tank 1, $m^2$
$C_i$ = inlet concentration, mg/L
$C_1$ = tank 1 outlet concentration, mg/L
$C^*$ = background concentration, mg/L
$Q$ = flow, $m^3/h$
$I$ = infiltration, m/h
$k$ = areal removal rate coefficient, m/h
$t$ = time, h

This equation, together with the dynamic water mass balance, may be solved numerically for a time series of inflows and incoming concentrations. This model has been described and exercised by Alvord and Kadlec (1996) for atrazine, and Kadlec (2001a) for phosphorus. This level of modeling has been utilized for describing phosphorus movement in treatment wetlands and surroundings in south Florida (Munson *et al.*, 2002). An example of a typical model fit for nitrate was shown earlier in Figure 14.21, and for atrazine in Figure 13.7 in Chapter 13.

### The Dynamic Model for Stormwater Treatment Areas (DMSTA)

A dynamic model of more complexity has been developed for phosphorus in agricultural runoff in south Florida (Kadlec and Walker, 1999; Walker and Kadlec, 2005). A simple extension of the *k*-*C** design models is the addition of phosphorus storage in the biota (solids) of the wetland ecosystem. During periods of high phosphorus availability, that storage will increase, and in periods of phosphorus famine, the storage will decrease. At all times, the ecosystem produces the residual sediments containing unavailable phosphorus, such as characterized earlier models (Walker, 1995). This extension has been named the Dynamic STA Design Model

(DMSTA). It is an unsteady state model that removes phosphorus to permanent burial in proportion to the amount of labile phosphorus in storage. There is an implied labile pool of phosphorus in addition to the permanently buried phosphorus. That labile pool is presumed to be drawn down by a return flux, or bleed-back. Temporal variations in the water budget (flows, rainfall, evapotranspiration, seepage, storage) are also simulated.

The goals of this modeling project were to provide a tool with three major attributes:

1. Provide capability for treatment wetland design calculations including dynamic events in pulse-flow systems. The implication is for at least one stationary storage compartment for phosphorus.
2. Allow for hydraulic flow patterns that deviate from the ideal extremes of one stirred tank and plug flow. Physical subdivision of large wetland cells is expensive but carries the implication of better hydraulic efficiency.
3. Allow for variability in, and choices for, the ecosystem community type. There was evidence of superior performance of SAV systems over emergent systems, and a continuing suggestion that periphyton systems could achieve lower phosphorus.

Given these priorities, the model was constructed to embody the concept of maximum simplicity while retaining essential descriptive features. Adding too many compartments and pathways has been found to result in unwieldy computer code and long simulation times (Fitz *et al*., 1996; HydroQual Inc., 1998). DMSTA keeps the size and complexity of the code to a level that permits rapid simulations (i.e., minutes) on a personal computer. Nevertheless, it is easily utilized for simulations of over 30 years' duration.

Phosphorus storage occurs in several compartments: plants, litter, algae, etc. (Wang and Mitsch, 2000). However, the computational overhead for tracking changes in several compartments is large, and calibration data for the many transfers is essentially absent. A single, aggregated storage compartment was therefore used.

Three dynamic mass balances are required at this level of modeling: one for the surface water budget, one for surface water phosphorus, and one for the dynamic, solid-phase phosphorus. To account for flow patterns, a specification is needed for the number of well-mixed units over which these mass balances are computed These considerations fix a model structure but not the rules for transfers. For the water budget, essentially one rule is needed: the stage–discharge relation for the wetland. There are three major phosphorus pathways that require transfer rules. Consequently, the chosen model needs three phosphorus removal parameters, in addition to hydraulic parameters. The model explains approximately 85% of the observed variance across data sets with a standard error of approximately 12% (Figure 14.25). The interested reader may find details at the website http://www.wwwalker.net/dmsta/.

Adaptations of this level of modeling are currently in development, such as a version for nitrogen, the Constructed Wetlands Algorithm for Nutrient Transformation (CWANT) (Howell *et al*., 2005).

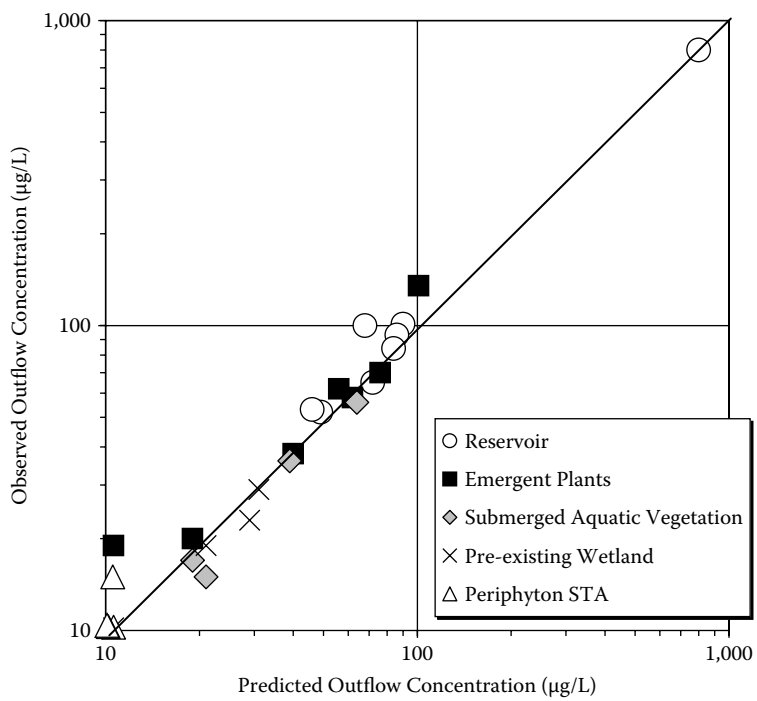



**FIGURE 14.25** Comparison of observed and predicted concentrations for calibration data sets for DMSTA.

**TABLE 14.14**
**Phosphorus Entering and Leaving the Hidden River Urban Stormwater Wetland Near Tampa, Florida, 1991–1993, for 83 Rain Events**

| | Ortho P (μg/L) | Total P (μg/L) |
|---|---|---|
| **Rain (45%)** | | |
| 10% | 1 | 0 |
| 25% | 4 | 3 |
| 50% | 6 | 10 |
| 75% | 10 | 17 |
| 90% | 13 | 25 |
| **Mean** | **7** | **13** |
| **90%/50%** | **2.17** | **2.50** |
| **Inlet (55%)** | | |
| 10% | 22 | 67 |
| 25% | 28 | 88 |
| 50% | 40 | 110 |
| 75% | 49 | 127 |
| 90% | 66 | 178 |
| **Mean** | **44** | **116** |
| **90%/50%** | **1.66** | **1.62** |
| **Outlet (27%)** | | |
| 10% | 3 | 7 |
| 25% | 7 | 27 |
| 50% | 11 | 40 |
| 75% | 16 | 60 |
| 90% | 28 | 86 |
| **Mean** | **14** | **45** |
| **90%/50%** | **2.55** | **2.15** |
| Mass reduction | 67% | 70% |

*Note:* The majority of the water leaves by ET (41%) and seepage (31%).

*Source:* Data from Carr and Rushton (1995) *Integrating a native herbaceous wetland into stormwater management*. Report by the Southwest Florida Water Management District, July 1995, Boca Raton, Florida.

### Variability

For event wetlands, the variability in performance is comprised of three contributions: (1) event-to-event inflow and concentration behavior, (2) annual seasonal patterns, and (3) stochastic scatter in removal efficiencies. Stormwater wetland performance variability is further compounded by the flow and concentration variations within events. Here, an example of the resultant variability for the Hidden River wetland in Tampa, Florida, is presented, based upon Carr and Rushton (1995) and the statistical analysis of URS (1999). Event-mean concentrations for 83 events form the basis of the analysis (Table 14.14). The ranges for each nitrogen species, across the percentile points of the distributions, are typically quite wide. Median outlet concentrations are reduced from either the rainfall or the overland flow inputs for nitrate and ammonia nitrogen, but organic nitrogen medians, and consequently TN medians, are higher in the output. The 90th percentile outlet values are higher than the medians by a considerable margin, a factor of 2.5 for oxidized nitrogen and 1.56 for TN. Thus, excursions for individual events may be expected, as is the case for continuous-flow systems. More percentile statistics may be found in URS (1999).

## SUMMARY

Event-driven wetlands include urban and agricultural stormwater systems, batch-operated wetlands, and combined sewer overflows that are treated in constructed wetlands. Most of these systems are FWS, but in Europe, CSOs are treated in HSSF wetlands. Urban and agricultural stormwater wetlands are usually comprised of a sedimentation basin preceding a marsh-pond complex. The amounts of water and pollutants that reach such wetlands are time variable in character. The time series of stormwater systems is driven by regional meteorology and the runoff characteristics of the catchment.

The primary intent of stormwater wetlands is to trap as much sediment as possible in the inlet basin so as not to clog the vegetated portion of the system with accumulated solids. The secondary focus of stormwater systems is the reduction of nutrients. Phosphorus and nitrogen compounds are reduced by the same internal processes as for continuous-flow wetlands. However, the hydraulic sequence for an event-driven wetland is a displacement of antecedent water, followed possibly by a flow through period, followed by batch holding during the interevent period. Metals and pesticides are also reduced in these systems.

Attempts at simplistic performance correlations have been largely unsatisfactory because of the high degree of variability in system designs and flow characteristics. Although it is intuitively obvious that a larger WWAR is desirable, this effect has not been tightly quantified. The use of continuous-flow rate constants holds some promise, but new calibrations are needed. Dynamic modeling seems to be necessary, but is only feasible for large projects. Until more knowledge is gained, technology-based designs are likely to continue to be the rule.

# *Part II*

## *Implementation*



# 15 Evolution of Sizing Methods

The purpose of Part II of *Treatment Wetlands, Second Edition*, is to provide information on how to design, construct, and operate a wetland for the purpose of water quality improvement. It is recognized that wetlands are almost never stand-alone treatment devices but rather form part of a treatment train. Other components may be mechanical, such as clarifiers or filters, or more natural, such as settling basins or lagoons. More than one type of wetland may be involved at the same site—FWS, VF, HSSF, or biosolids systems. In this chapter, historical sizing methods are reviewed and placed in perspective.

Design of treatment wetlands may be roughly divided into two categories: sizing calculations and physical specifications. Sizing procedures may in turn be predicated on the particular application, whether the purpose is for treatment of flows intended for surface water discharge, or for conditioning water for groundwater discharge, or for retaining and treating stormwater runoff. Further, local regulations may require compliance with specific numerical standards (performance-based) or may simply require certain design specifications according to prescriptive criteria (technology-based).

The physical aspects of treatment wetland design seem deceptively simple. Indeed, the basic elements have been on record for over a hundred years.

From Brian Mackney (Australia) via Hans Brix (1994d), an excerpt from an essay written by Nemo to the head of the Hornsby Literary Institute in 1904 reads:

> Anyone who has a little ground about his house can dispose of his dirty water as follows:
>
> Dig up a plot of ground thoroughly to a depth of fifteen to eighteen inches. Cut a channel leading from the kitchen and washhouse into the highest side of the plot and let all the dirty water drain into it. Plant the plot with plants that grow rapidly and require a great deal of water such as Arum Lilies, for instance. The dirty water will be all absorbed by the roots of the plants and a most luxuriant garden will be produced which will defy the hottest weather and will always be green and beautiful. By this means a curse will be transformed into a blessing. Twenty or thirty feet square properly worked would be enough for any ordinary family.

Of course, Nemo did not conceive of the difficulties of scale-up from a single household to all the wastewater from a city of 100,000 population. For even modest-sized systems, there is a need to consider the system layout, the number of individual cells and their arrangement, and how to move water through a large wetland complex. Questions of hydraulic and vegetation efficiency have important economic consequences. Therefore, the subsequent chapters consider the engineering of the wetland.

The first documented use of a wetland within a deliberately engineered treatment vessel appears to belong to Cleophas Monjeau (1901), as shown in Figure 15.1.

Claims for this United States patent (issued in 1901) include distributed vertical flow, a fluctuating water level, and aeration of the wastewater. This would be considered a cutting-edge wetland design even in the 21st century.

Design of wetland systems may be either performance-based or technology-based, depending upon whether the performance expectations are explicit in an operating permit or implicit in a set of statutory design specifications. Large systems are often in the former group, whereas small on-site and urban stormwater systems are in the latter. The primary focus of this chapter is performance-based design.

## 15.1 HISTORICAL PERSPECTIVES

Much of treatment wetland sizing has evolved along very simplistic lines, mostly adapted from civil and sanitary engineering. However, it is become increasingly apparent over the last 20 years that treatment wetlands are much more biologically complex than treatment processes used in the sanitary engineering field, and simplifying assumptions applied to activated sludge, trickling filter, and pond systems do not adequately describe the internal biogeochemical processes that govern treatment performance in wetlands.

Historically, mechanistic wetland models have been of little utility because of calibration difficulties. General rules of thumb for area requirements are used to good avail in conceptual design. Input–output regressions allow calculations of outlet concentrations for specified systems, and scatter plots are available to examine intersystem variability. Several variants of first-order models are now in use. Virtually all of these existing techniques fail to address significant aspects of wetland performance, including effects of hydraulics, meteorology, interactions of biota and treatment, supply-side constraints, and seasonal and stochastic variability. Soil accretion and the sustainable storage and dilution of irreducible constituents are other aspects not covered by existing design methods. There is also great danger of using empirical relations outside the envelope for which they were developed, and of unintentionally exceeding the capabilities of the fundamental wetland processes. Proper use of this imperfect set of design tools is a current design challenge.

Three principal themes have been prevalent in the history of constructed treatment wetland design: consideration of the pollutant and hydraulic loadings, first-order removal models, and regression equations. First, some general history and concepts of wetland design are discussed, and then some specifics for the principal variants of the technology.

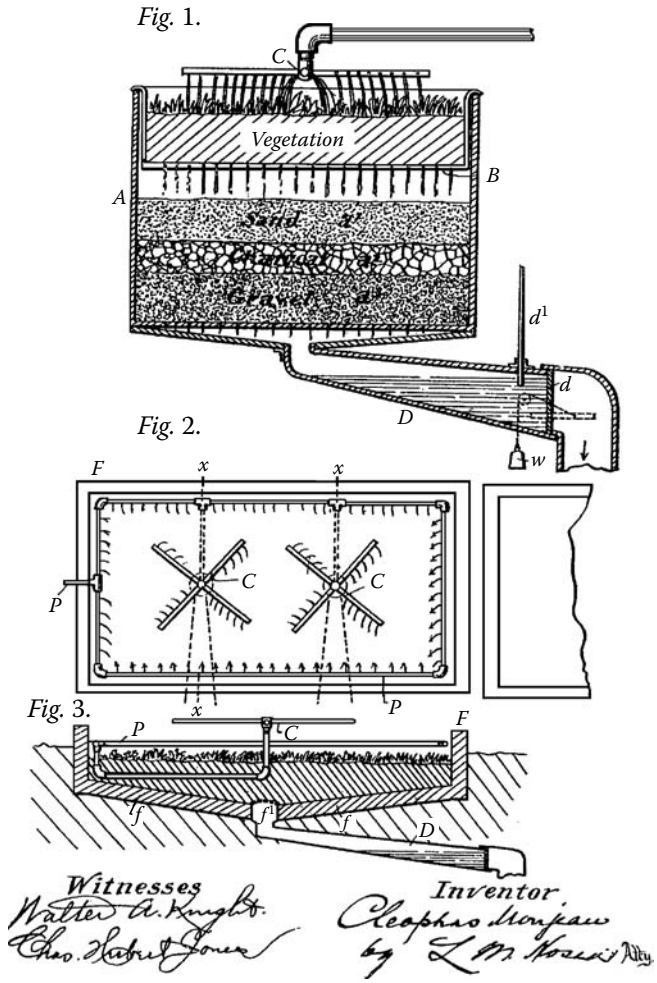



**FIGURE 15.1** 1901 U.S. patent for a treatment wetland system. (From U.S. Patent 681,884.)

## First-Order Modeling

In the mid-1980s, simple first-order models of pollutant removal in wetlands made their appearance in the treatment wetland literature (U.S. EPA, 1983b; Boon, 1985; Kadlec, 1985; Reed *et al*., 1988). This was, in part, a natural extension of models that had earlier been developed for waste stabilization ponds (Marais and Shaw, 1961; Marais, 1974; Thirumurthi, 1974). Some of these early models were founded on shaky ground because they borrowed extensively from nonwetland results and were not well-calibrated to actual treatment wetland data. Over the past two decades, considerable progress in understanding the behavior of treatment wetlands has allowed major improvements in the way that first-order models are constructed and implemented. It is the premise of this book that first-order modeling remains one of the most effective wetland design tools but that other information can and should also be brought to bear on design.

The information in Part I sets forth the many processes affecting pollutant removal in wetlands. Nearly all of the component processes are first-order within some range of constraints that are often met within wetland ecosystems. Therefore, it is not surprising that longitudinal profiles are virtually all describable by such a model, provided proper account is taken of background concentrations and the actual flow patterns observed in treatment wetlands. Further, calibrated first-order models are capable of adequately representing the trends in time series of pollutant concentrations exiting wetland systems. However, it is also the case that the many varieties of treatment wetlands, together with the variations of a given type, should not be expected to produce identical results. Plant varieties and their fractional coverage, media depth, aspect ratio, compartmentalization, and environmental factors vary from wetland to wetland. A distribution of $k$-values is to be expected, and has been observed for all the common pollutants (see Part I). Intersystem variability is thus a fact of life for wetland systems that are intimately connected to their climate and surrounding environment.

It is also true that, without exception, effluent concentrations exhibit random variation. Environmental and biological factors drive behavior on hour-to-hour and day-to-day time scales. Any model capable of describing such rapid phenomena would be quite complex, and in fact no such models currently exist. Therefore, effects such as those of wind, rain, and animal invasions must be dealt with through an understanding of the intrasystem variability present in the performance of a treatment wetland. Because such random effects are a large proportion of the wetland response, that variability must be understood, quantified, and included in design. Deterministic relations, either a rate-constant model or a loading relationship, will produce only the general trend of outlet concentrations and not the random variability.

## Previous Criticism of First-Order Modeling

Reliance on first-order modeling, although nearly universally accepted, has had one severe critic. Because that criticism appears in a U.S. government publication advocating a specific treatment wetland design procedure, it is necessary to examine this lone voice of opposition in more detail.

U.S. EPA (2000a) contended that " … the development of a rigorous model of the process, or parts of it, has not been achieved as is true with many of the wastewater treatment processes designed today."

This statement is no longer correct. Credible treatment wetland models exist at all the levels that they do for activated sludge processes, which are the most advanced in terms of modeling. The work of McBride and Tanner (2000), Langergraber (2001), Wynn and Liehr (2001), Howell *et al*. (2005) and others has carried wetland modeling to an advanced level of detail for municipal wastewater treatment. Stormwater wetland models abound (Fitz *et al*., 1996; Moustafa and Hamrick, 2002; Munson *et al*., 2002; Walker and Kadlec, 2005). However, to our knowledge, these complex models are not often in use for purposes of design, the exception being the dynamic model for stormwater treatment areas (DMSTA) (Walker and Kadlec, 2005).

Indeed, U.S. EPA (2000a) went on to conclude that the design should be based on loading, an approach that:

> … has been used for many decades by environmental engineers in the design of highly complex unit processes, including the activated sludge process and waste stabilization ponds. Only when a carefully designed series of iterative studies have been conducted, and data based on quality-controlled specifications have been analyzed, can rigorous models be provided for use in wetland system design.

It is almost unbelievable that this statement should have been made on the heels of a carefully designed series of iterative studies sponsored by U.S. EPA for the purpose of development of design equations (George *et al*., 1998) for treatment wetland systems.

A compromise position is the use of highly aggregated models that contain major features of the observed behavior of treatment wetlands. That compromise is a first-order model of some sort, possibly with temperature- or season-dependent parameters. A primary message of the U.S. EPA (2000a) is that such first-order removal models should not be used, because these are prominently absent from that manual. That is paradoxical, because the wetland publications of all of the major contributors/authors advocate the use of first-order removal models (Reed *et al*., 1988; Crites, 1994; Reed *et al*., 1995; Kadlec and Knight, 1996; Kemp and George, 1997; Crites and Tchobanoglous, 1998; Campbell and Ogden, 1999).

In recent years, first-order models have been reaffirmed as a useful tool in wetland design. Rousseau *et al*. (2004) performed a review of model-based design of horizontal subsurface-flow treatment wetlands, and concluded:

> At present, the state-of-the-art *k-C** model seems to be the best available design tool if the designer makes sure that all the assumptions are fulfilled and if he is aware of the pitfalls in the model.

Crites *et al*. (2006) reiterate the basic features of both areal and volumetric first-order models for free water surface (FWS) wetlands, and acknowledge that "… they do not directly account for the complex reactions and interactions that occur in wetlands." Crites *et al*. (2006) go on to state:

> Such an approach is the best that can be done with the currently available database and understanding of wetland processes.

In this book, the first-order model is retained as the primary tool in the design process. However, the loading approach is also retained, as a means of accounting for intersystem variability.

## Loading Specifications

An early source of guidance utilized by wetland analysts was the methods used by wastewater stabilization pond designers. A common method was the prescription of a specified areal-loading rate for biochemical oxygen demand (BOD), on the order of 40 kg/ha·d to achieve an outflow concentration of 30 mg/L (Reed *et al*., 1988). BOD loading is still the recommended design method for ponds, although with modifications, that acknowledge temperature effects (Shilton and Mara, 2005; Shilton, 2005). At 3°C, the recommended loading rate is 40 kg/ha·d, which is the winter condition for cold climate systems. As the minimum winter temperature goes up, say, to 20°C for a tropical climate, the allowable loading goes up as well to about 250 kg/ha·d. Reed *et al*. (1988) suggested using the BOD-loading rate to the wetland as a constraint, not for design sizing, with an upper limit of 100 kg/ha·d. It is perhaps coincidental that U.S. EPA (2000a) concluded that the allowable design loading to a fully vegetated FWS wetland should be 40 kg/ha·d, identical to that for temperate pond design.

The areal-loading procedure may easily be converted to a population equivalence (PE), if the person equivalence of the particular wastewater is known. For instance, the per-person BOD generation rate in the United States is approximately 63 g/PE·d (U.S. EPA, 2002c) and in Austria is specified as 60 g/PE·d (Haberl *et al*., 1998). In the United States, BOD generation is also taken to be 60 g/PE·d, but a septic tank (or other primary treatment device) is virtually always used as pretreatment for domestic wastes, and therefore 40 g/PE·d is used as the influent to a wetland (Wallace and Knight, 2006). So, for example, if a design specification is 5 $m^2$/PE for a HSSF wetland treating primary effluent, then the equivalent loading specification is 40/5 = 8 g/$m^2$·d = 80 kg/ha·d.

To compare first-order calculations to loading criteria, the hydraulic loading to the wetland must be known. The rate-constant approach separates the effect of inlet concentration from that of flow rate. The water flow per person equivalent is to some extent independent of the amounts of pollutants generated. Water usage in a typical household varies from 50 to 200 L/PE·d, depending on whether it is situated in an arid or wet location, or in a developed or developing country. Small communities have slightly greater water usage per person.

The relationship between rate-constant specification and areal-loading specification is straightforward, but not unique: the rate approach is strength (concentration) dependent and the loading method is not. The connection for a pollutant with zero background is

$$C_o = C_i \exp\left(-\frac{k}{q}\right) = C_i\left[\exp\left(-\frac{kC_i}{qC_i}\right)\right] \qquad (15.1)$$

where

$C_i$ = inlet concentration, mg/L
$C_o$ = outlet concentration, mg/L
$k$ = rate constant, m/d
$q$ = hydraulic loading, m/d

and

$qC_i$ = pollutant loading, g/$m^2$·d

It is clear from Equation 15.1 that the outlet concentration depends not only on inlet pollutant loading but also on inlet concentration.

Suppose it is required that $C_o < 25$ mg/L, and that $k = 0.1$ m/d. For $C_i = 100$ mg/L, the allowable hydraulic loading is $q = 0.072$ m/d. The corresponding pollutant loading is 7.2 g/m²·d. However, if the inlet concentration is $C_i = 200$ mg/L, the rate model suggests that the allowable hydraulic loading is $q = 0.048$ m/d and the corresponding pollutant loading is 9.6 g/m²·d. Thus, the rate approach gives different area requirements, depending on waste strength. To further complicate the matter, we might consider that the two cases differed only in water use, and that the same number of population equivalents (same mass load) was involved. Suppose PE = 100 and the per capita generation was 50 g/PE·d, for a total of 5,000 g/d. For the weak strength of 100 mg/L (more water usage), the area required would be 5,000/7.2 = 694 m² or 6.9 m²/PE. For the stronger strength of 200 mg/L (less water usage), the area required would be 5,000/9.6 = 520 m² or 5.2 m²/PE. Thus, the loading method and the rate method are not equivalent.

## Regression Equations

The principal variables that determine the outlet concentration ($C_o$) are the hydraulic loading ($q$), or the equivalent detention time ($h/q$), and the inlet concentration ($C_i$). Other influences include temperature, solar radiation, and pH. There is, therefore, the possibility of simply postulating a functional relation, such as a product-power law, and regressing intersystem wetland input–output data to fit that equation. In so doing, some of the power of intrawetland data is lost, because such regression formulae may not represent the observed longitudinal profiles for a specific wetland. However, there can be no doubt that an individual data set can be fit with a reasonable guess at a fitting formula, because there are typically three or more fitting parameters. For example, Kadlec and Knight (1996) set forth linear regressions and power law regressions for different common pollutants. A few examples are:

Gumbricht (1993a), total nitrogen leaving a single FWS SAV system ($R^2 = 0.92$):

$$C_o = 1.63 + 0.78C_i - 1.7\ln(\tau_{nom}) \tag{15.2}$$

where

$C_i$ = inlet concentration, mg/L
$C_o$ = outlet concentration, mg/L
$\tau_{nom}$ = nominal detention time, d

Kadlec and Knight (1996), nitrate leaving several FWS systems of different types ($R^2 = 0.35$):

$$C_o = 0.093\, C_i^{0.474} q^{0.745} \tag{15.3}$$

where

$q$ = hydraulic loading rate, cm/d

Tunçsiper *et al*. (2006), ammonia leaving one square meter HSSF mesocosms with various plants ($R^2 = 0.67$):

$$\left(\frac{(C_i - C_o)}{C_i}\right) = 0.371\ \text{pH} - 0.0048\, q + 0.080\, C_i - 2.286 \tag{15.4}$$

The operational data and multiple linear regression models are shown in Figure 15.2, along with the rate-constant model. Tunçsiper *et al*. (2006) concluded that the multiple linear model was better, but that is not supported by the sum of the squared errors derived from data and models (SSQE = 0.30 for rate-constant model, and SSQE = 1.98 for multiple linear model).

Great care must be exercised in defining the variable ranges of such regression models. The potential extremes of

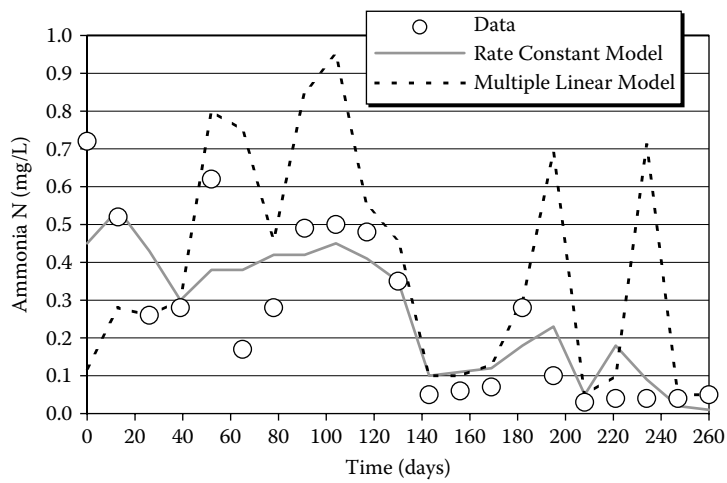



**FIGURE 15.2** Time series of ammonia concentrations exiting HSSF mesocosms, together with rate constant and multiple linear regression models. (From Tunçsiper *et al*. (2006) *Water Science and Technology* 53(12): 111–120. Reprinted with permission.)

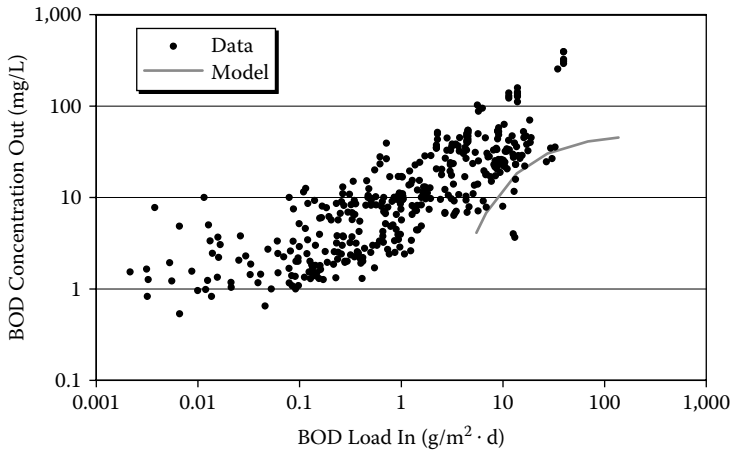



**FIGURE 15.3** A presumptive model line on the BOD-loading graph for an inlet concentration of 50 mg/L. The model is the first-order volumetric with $k_{V1} = 0.678$ d$^{-1}$ and with $C^* = 0$, but truncated at BOD < 5 mg/L per literature recommendation. A free water depth of 45 cm is presumed for the model.

variables can produce unintended anomalies that are a potential trap for designers. For instance, the Gumbricht (1993a) regression (Equation 15.2) produces negative concentrations for long-detention times, a fact that might go unnoticed by a potential user of this equation set. The Kadlec and Knight (1996) regression (Equation 15.3) suggests that $C_o$ increases with decreasing $C_i$ for $C_i < 1.0$ mg/L, which is a highly improbable circumstance. The Tunçsiper *et al*. (2006) regression (Equation 15.4) shows that removal goes up with inlet concentration and, in fact, for $C_i > 10$ mg/L, more than 100% removal is forecast. That $C_i$ is higher than the calibration range, but an unsuspecting designer could easily misapply Equation 15.4 to that condition.

These example regressions contain 3, 3, and 4 parameters, respectively. Any possibility of generalization, and use in design, is hindered by the need to predict all of those coefficients in a new situation. Consequently, regression equations have not found use in treatment wetland design.

## 15.2 FREE WATER SURFACE WETLANDS

Early FWS treatment wetlands were often natural wetlands employed to improve water quality. Although those ecosystems were preexisting, there was no issue of sizing, but rather an issue of how much water could be put into the wetland. Recipes existed, and continue to exist, covering the appropriate hydraulic loading to natural systems (e.g., for Florida). In contrast, FWS constructed wetland design sizing for continuous flows has historically been virtually entirely performance based.

Unfortunately, there has been considerable misinterpretation and misuse of these simplistic models. One of the problems has been unsupported pronouncements of model parameters. When plug flow and $C^* = 0$ are presumed, there is only one parameter remaining, which is here designated as $k_1$ (or $k_{V1}$). For instance, there is an oft-repeated value of $k_{V1} = 0.678$ d$^{-1}$ in U.S. literature for BOD reduction in FWS wetlands at 20°C (Reed *et al*., 1995; Water Environment Federation, 2001; Crites *et al*., 2006). This value is intended for use in an exponential (plug flow) model with a zero background concentration. Although no basis for this number has ever been presented, it is now possible to place it in the perspective of calibrations for large numbers of FWS systems. The suggested value, for a system 40 cm deep, corresponds to $k = 100$ m/year. This ranks at the 98th percentile of $k_1$-values for 386 wetland-years of data for FWS wetlands (Figure 15.3). From Figure 15.3, it is clear that the exponential plug-flow model with $C^* = 0$ has the wrong shape compared to intersystem data, and therefore a different calibration $k_V$ cannot fix it. Most of the time this particular single-parameter model will overestimate removals, as well as the model providing incorrect trends.

## 15.3 STORMWATER WETLANDS

Stormwater wetlands are also FWS systems, but these differ markedly in design philosophy. Early in the brief history of urban stormwater wetlands, data was collected from a variety of constructed and natural systems, and the central performance tendency of these treatment wetlands was determined (Strecker *et al*., 1992; Schueler, 1992). The time-variable nature of stormwater wetlands means that the information was and is quite sparse. Source characterization is difficult and often the result of semiquantitative rules for the amounts of water and pollutants that are washed off of various types of watersheds.

Given this meager performance base, it is not surprising that the earliest sizing rules were empirical. One method involves the specification of the wetland area compared to the contributing watershed area, the wetland-to-watershed area ratio (WWAR). This is a scaling rule, comparable to those in use for continuous-flow FWS wetlands. The implied performance of a particular WWAR is inferred from the performance data from operating systems similar to WWAR.

A second approach relies on the capture and detention of a specified amount of runoff, which is then subject to treatment during the interevent period. For instance, the anticipated runoff from a rain event of specified return frequency may be used to set the wetland water volume. Runoff from larger events is partially bypassed or runs quickly through the wetland. Again, the implied degree of treatment is that associated with holding the water during the interim periods.

A third approach involves adapting the continuous flow rate-constant method to event-driven systems. The use of the *k-C** model, with average flow rates, was suggested by Kadlec and Knight (1996) and by Wong and Geiger (1997). A number of calibrations for stormwater systems were provided in the summary of Carleton *et al*. (2001). More recently, the concept has been extended to the *P-k-C** model by Wong *et al*. (2006).

A drawback of the use of *P-k-C** calculations for event-driven wetlands is that rate constants for continuous-flow situations do not necessarily carry over to the averaged transient-flow situation (Kadlec, 2001a). As a result, dynamic modeling is to be preferred. However, the complexity of dynamic modeling is daunting, partly because of the need to construct input time sequences for a significant period of operation. Nonetheless, for large and costly projects, dynamic simulations may be warranted. Currently, these are in use for the design of the phosphorus removal wetlands in South Florida (Walker and Kadlec, 2005).

## 15.4 HORIZONTAL SUBSURFACE FLOW WETLANDS

Over the last 50 years, multiple variations of horizontal subsurface flow (HSSF) wetlands have been developed, and in more recent years, HSSF wetland technology has made significant advances. As a result, there are a large number of reference documents on HSSF wetlands that contain outmoded concepts and should no longer be used for design purposes. This section provides an overview of HSSF wetland design and how the technology has evolved.

### The Root-Zone Method

In 1952, a German scientist, Kathe Seidel, began investigating the water purification capabilities of bulrush (*Schoenoplectus lacustris*) grown in artificial-rooting environments. In the early 1960s, in collaboration with Seidel, Reinhold Kickuth at the University of Göttingen, Germany, developed a wetland treatment process known as the *root-zone method* (Wurzelraumentsorgung) (Brix, 1994d). Root-zone method wetlands are comprised of a soil media (typically clay loam to sandy clay), and calcium, iron, or aluminum additives were sometimes proposed as additives to the media to increase phosphorus adsorption (Kickuth, 1977). The soil was seen as a reactive media, a concept that has resurfaced with HSSF wetlands using expanded shale and clay aggregates (see Chapter 10). Although no longer considered a modern design concept, many wetlands designed on the principles of the root-zone method exist in Europe.

In most of the root-zone method wetlands, the exclusive use of the common reed (*Phragmites australis*) was based on the theory that root formation would increase the hydraulic conductivity of the soil matrix from approximately $10^{-5}$ m/s to $10^{-3}$ m/s within two to four years (Cooper and Boon, 1987). The intended hydraulic conductivity of the soil ($10^{-3}$ m/s) required many of these wetlands to be considerably wider than their length, and the beds were often tipped at a slope of 1% or greater to increase the gradient in the (mistaken) belief that the tipped beds would force subsurface flow through the beds (Figure 15.4).

After a few years, the collaboration between Seidel and Kickuth ended for personal reasons. Both scientists and their respective schools became rivals. The information produced by the two groups during this period was often conflicting, and resulted in confusion among practicing wastewater engineers and the governing regulatory authorities (Börner *et al*., 1998). By the early 1980s, most wetlands in Germany were being constructed on the root-zone method, although examples of the MPIP (Seidel) system were constructed in St. Bohaire, France (Liénard *et al*., 1990), and Oaklands Park, United Kingdom (Burka and Lawrence, 1990).

### Evolution of HSSF Wetland Design in Europe

In 1984, water utilities in the United Kingdom began to investigate the root-zone method as an alternative for small treatment works (Cooper and Boon, 1987). An effort was made to coordinate research efforts with Denmark, Germany,

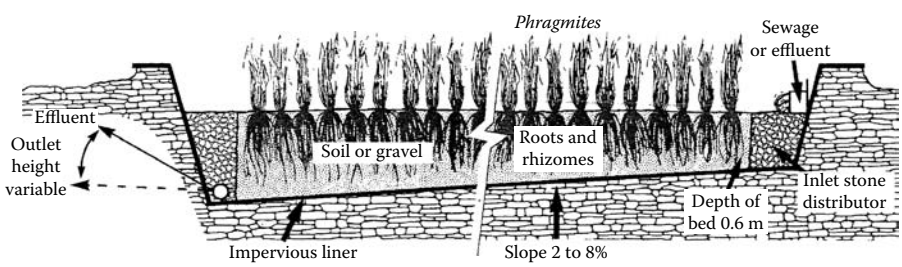



**FIGURE 15.4** Example of an early root-zone method HSSF wetland. (From Cooper and Boon (1987). In *Aquatic Plants for Water Treatment and Resource Recovery*. Reddy and Smith (Eds.), Magnolia Publishing, Orlando, Florida, pp. 153–174. Reprinted with permission.)

France, Spain, and the Netherlands. Early treatment wetlands in Denmark and the United Kingdom had designs based on the root-zone method principles. The claim of hydraulic-conductivity improvement from $10^{-5}$ m/s to $10^{-3}$ m/s did not hold true. Instead, the hydraulic conductivity either remained stable or decreased with time, remaining on the order of $10^{-5}$ m/s (Brix and Schierup, 1989a; Netter and Bischofsberger, 1990; Findlater *et al*., 1990; Coombes, 1990; Bucksteeg, 1990).

Because the anticipated increase in hydraulic conductivity did not occur, the HSSF beds did not have enough cross-sectional area to allow for subsurface flow. As a result, water would pond at the inlet and flow across the surface of the bed. Thus, the hydraulically failed mode of operation for these wetlands was overland flow—essentially a FWS wetland with a very shallow water depth. As it has turned out, there is not a large difference in treatment performance between FWS and HSSF wetlands, so the overland flow mode of operation has often been tolerated because it provides an acceptable level of treatment, even though the systems do not function as intended.

However, some of these systems were constructed with sloping beds (top and bottom surfaces both sloped). When overland flow occurred, the water would channelize, further reducing treatment performance. These experiences led to the creation of design standards for subsurface-flow wetlands based on coarser bed materials and beds with flat upper surfaces (only bottom surface sloped) (WRc, 1989).

Other root-zone method predictions also turned out to be inaccurate. The actual oxygen transfer from the plant root systems was much lower than anticipated (Brix and Schierup, 1990). In addition, actual BOD removal was slower (observed rate removal coefficient of 0.10 m/d versus the predicted 0.19 m/d), resulting in a trend towards larger bed areas (5 $m^2$/PE versus 2.2 $m^2$/PE) (EC/EWPCA Emergent Hydrophyte Treatment Systems Expert Contact Group and Water Research Centre, 1990).

Regional differences in HSSF wetland designs still exist. The practice in Germany is to use relatively fine sand (0.2 mm to 1.0 mm) (Gesellschaft zur Förderung der Abwassertechnik d.V [GFA], 1998) because it is thought to be the best compromise between available surface area for biofilm growth, suitability as a rooting media, and hydraulic conductivity (Geller, 1996). Because they are designed in accordance with Darcy's law, these beds are much wider than they are long. This design tactic minimizes the organic loading across the inlet cross-sectional area. German authorities report that the decomposition of most organic substances is essentially complete within 1 to 2 m of the inlet zone (Geller, 1997). Figure 15.5 is a schematic representation of a HSSF wetland in Germany (Geller *et al*., 1990).

The HSSF wetland systems in Austria, the Czech Republic, and the United Kingdom are usually designed with the media of similar diameter. Austrian standards recommend the use of a gravel media between 4 and 8 mm (ÖNORM B 2505, 1997). HSSF wetlands in the Czech Republic tend to use sand and gravel less than 20 mm (Vymazal, 1996). In the United Kingdom, bed materials of 3 to 6 mm and 5 to 10 mm have been recommended (EC/EWPCA Emergent Hydrophyte Treatment Systems Expert Contact Group and Water Research Centre, 1990). HSSF wetlands using coarser materials are often constructed with a length-to-width (L:W) ratio of greater than 1.0. For example, the average L:W ratio was 1.76:1 for 28 HSSF wetlands in the Czech Republic (Vymazal, 1996).

Wetland systems throughout Scandinavia are typically dimensioned using EC/EWPCA criteria. In Norway, the use of light-expanded clay aggregate (LECA) has been used as an alternative to gravel (Jenssen *et al*., 1994b). Assessment of the performance of these wetlands indicates that LECA has a high-phosphorus sorption capacity (Jenssen *et al*., 1996) compared to natural aggregates. As the technology has gained acceptance in Scandinavia, LECA remains one of the preferred media materials because of its availability and its ability to bind phosphorus (Jenssen *et al*., 2002). Phosphorus removal in LECA wetlands occurs through chemical

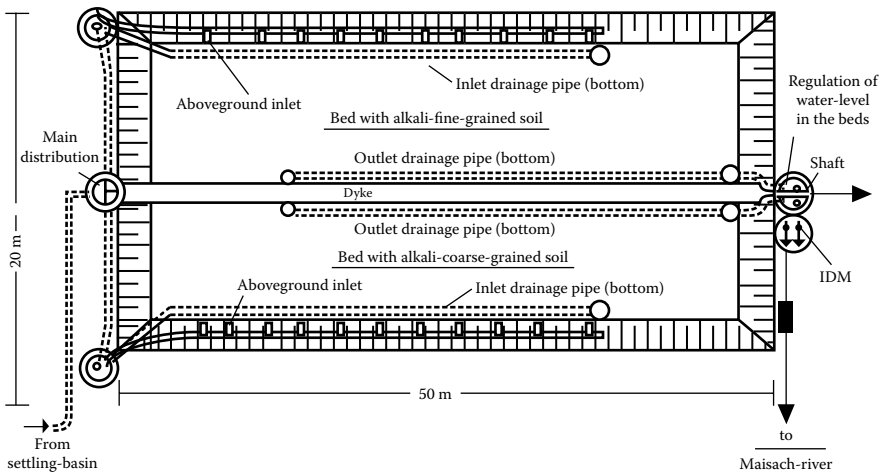



**FIGURE 15.5** HSSF wetland in Germerswang, Germany. (From Cooper and Findlater (1990) *Constructed Wetlands in Water Pollution Control.* Pergamon Press, New York. Reprinted with permission.)

adsorption to the surface of the media (Zhu *et al*., 1997). When the sorption sites are exhausted, phosphorus removal ceases. Studies in Sweden indicate that these adsorption sites are primarily associated with iron and aluminum oxides present on the surface of the particle, and are a function of the parent material and manufacturing process (Johansson, 1997). Materials with similar surface characteristics also display comparable phosphorus removal capabilities (Brooks *et al*., 2000). Phosphorus sorption capacities of various SSF aggregates are discussed in detail in Chapter 10.

The HSSF wetlands are recognized as being effective for BOD removal. However, due to the low-oxygen transfer rates in HSSF wetlands, nitrification is limited. Vertical flow wetlands that are capable of nitrification are being used increasingly in Europe (Cooper *et al*., 1997). Combinations of vertical flow and horizontal flow wetlands are being developed (Platzer, 1996; Cooper, 1999). These hybrid systems are designed with the goal of nitrification (in the vertical flow wetland) and denitrification (in the horizontal flow wetland).

## Evolution of HSSF Wetland Design in North America

In the United States between 1972 and 1976, Frederic Spangler and colleagues studied artificial wetlands (Spangler *et al*., 1976b) in conjunction with a study of the assimilation capabilities of natural wetlands. The earliest artificial marsh cells contained wire frames to support the emergent wetland plants, but this was quickly abandoned in favor of gravel media (19 to 25 mm) covered with smaller pea gravel. These early gravel-filled marsh cells could be operated with the water above or below the gravel surface, although subsurface flow was identified as the preferred mode of operation.

In 1973, Seidel obtained a U.S. patent (Seidel, 1973), and her concepts were more widely introduced in the United States (Wolverton, 1987a). By 1981, B.C. (Billy) Wolverton and his colleagues at the National Aeronautics and Space Administration (NASA), who had been researching floating-plant treatment systems, began to focus on subsurface-flow wetlands (Wolverton, 1983). These rock–reed filters were very long and narrow and used a coarse gravel media of crushed railroad ballast (Wolverton, 1987b). Although these HSSF wetlands were similar to the horizontal stages in the Max Planck Institute Process (MPIP) systems, they operated without the preceding vertical flow wetland cells. Figure 15.6 is a schematic representation of an early rock–reed filter for a single-family home (Wolverton and Wolverton, 2001).

Richard Gersberg and colleagues began to study nitrogen transformations in pilot-scale subsurface-flow wetlands (Gersberg *et al*., 1983) and larger demonstration-scale wetlands (Gersberg *et al*., 1984) in Santee, California. Early studies treated nitrified secondary effluent, although primary effluents were used in later studies. Subsequent work focused on pathogen reduction in subsurface-flow wetlands (Gersberg *et al*., 1987).

In 1988 the U.S. Environmental Protection Agency published a design manual on constructed wetlands that outlined design procedures for subsurface-flow wetlands (U.S. EPA, 1988b). This manual included a first-order plug-flow equation for BOD reduction with a temperature- and surface-area-adjustable removal rate. Media sizes ranged from a $d_{10}$ of 1 to 8 mm, with associated hydraulic conductivities of $4.9 \times 10^{-3}$ to $5.8 \times 10^{-3}$ m/s. The 1988 manual was replaced with updated manuals in 1993 (U.S. EPA, 1993c) and 2000 (U.S. EPA, 2000a).

The Tennessee Valley Authority (TVA) began demonstration of subsurface-flow constructed wetlands in 1986, based on design information obtained from the work of Kickuth, Wolverton, and Gersberg (Steiner *et al*., 1987). This led to the development of a guide for small system and single-family home wetland system design in 1991, which was

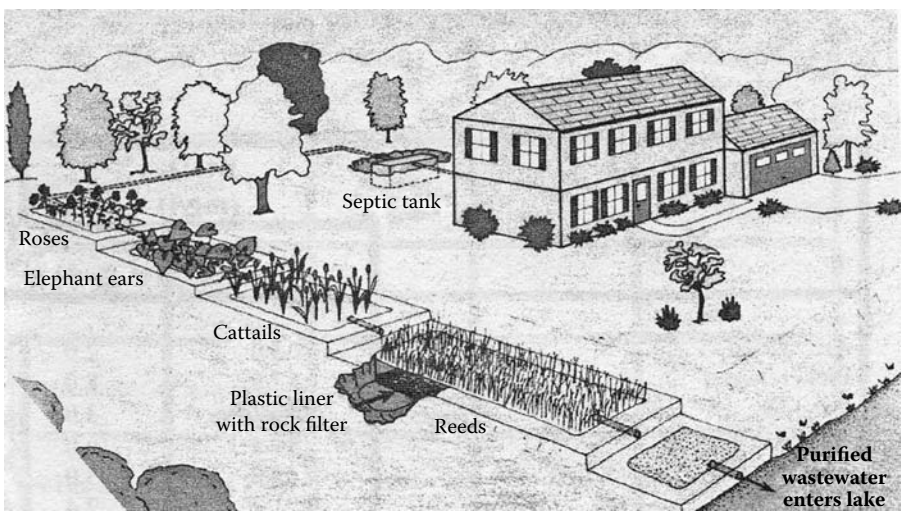



**FIGURE 15.6** Early rock–reed filter concept for treatment of wastewater from a single-family home. (From Wolverton and Wolverton (2001) *Growing Clean Water.* Wolverton Environmental Services, Picayune, Mississippi. Reprinted with permission.)

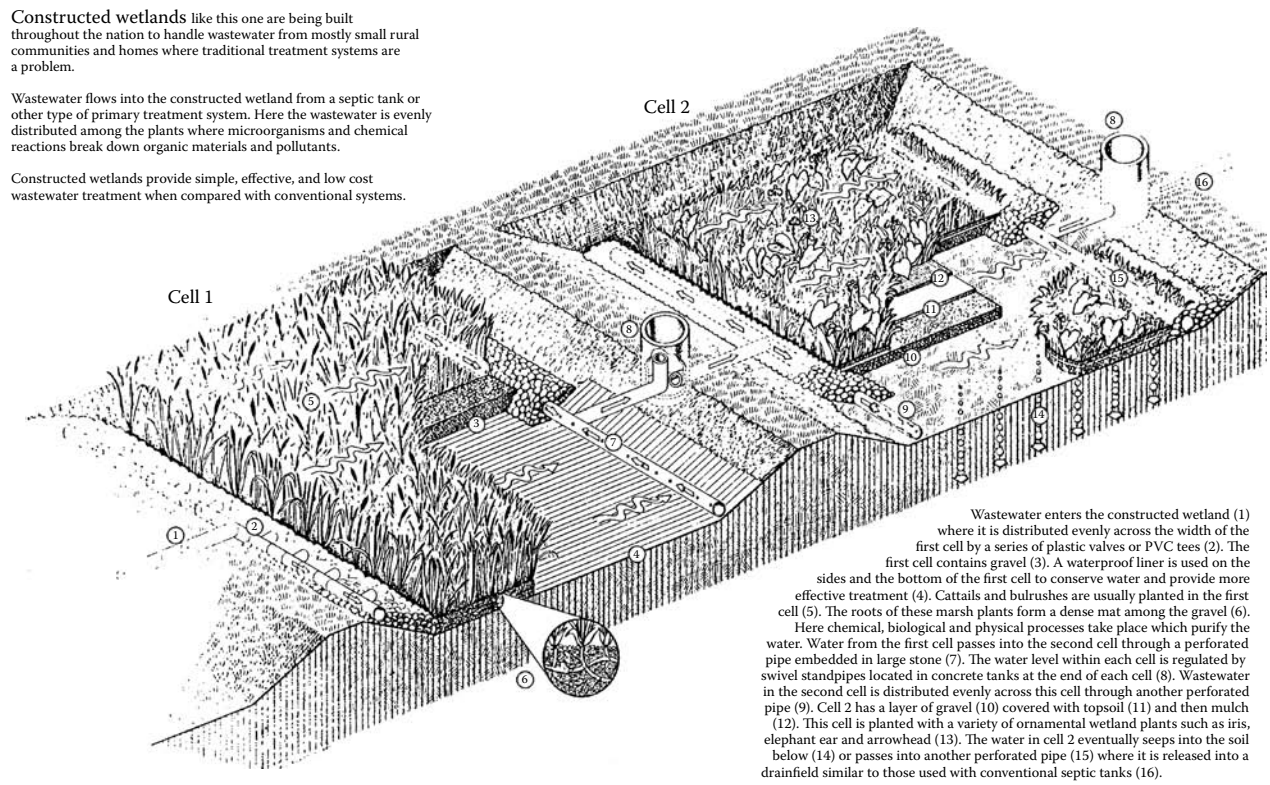



**FIGURE 15.7** HSSF wetland schematic from TVA wetland design manual. (From Steiner and Watson (1993) *General Design, Construction, and Operation Guidelines: Constructed Wetlands Wastewater Treatment Systems for Small Users Including Individual Residences.* Second Edition, TVA/WM-93/10, Tennessee Valley Authority Resource Group Water Management: Chattanooga, Tennessee.)

revised and updated in 1993 (Steiner and Watson, 1993). A schematic representation from this early guide is shown in Figure 15.7.

The sizing criterion was effectively 11 $m^2$/PE. The TVA guidelines recommended a gravel media of 3 to 6 mm, presumed to have hydraulic conductivity of $3 \times 10^{-3}$ m/s, be used to account for clogging in the gravel bed. This guideline also recommended that the cross-sectional organic loading be limited to less than 244 g BOD/$m^2$·d to minimize the potential for bed clogging and associated overland flow.

In 1993, the U.S. Environmental Protection Agency published a technology assessment of subsurface-flow wetlands (U.S. EPA, 1993f) that compared the design methods of Wolverton and TVA. This report recommended a first-order plug-flow equation for BOD reduction, corrected for temperature. Media sizes of 12 to 25 mm with a presumed hydraulic conductivity of $1.2 \times 10^{-2}$ m/d and an L:W ratio of 2:1 were recommended. Also in 1993, Region Six of the U.S. Environmental Protection Agency published design guidelines that originally recommended 50- to 250-mm rock media (U.S. EPA, 1993c); this recommendation was later revised to be in conformance with the 1993 Technology Assessment (U.S. EPA, 1993f).

In 1995, Sherwood Reed and his colleagues published design recommendations for subsurface-flow wetlands (Reed *et al*., 1995), which essentially expanded on an earlier work (Reed *et al*., 1988). Reed's work included thermal design, a first-order plug-flow equation for BOD and ammonia removal, and an empirical relationship for total suspended solids (TSS) removal. Recommended media sizes ranged from a $d_{10}$ of 2 to 128 mm, with presumed hydraulic conductivities between $1.2 \times 10^{-3}$ m/s and 2.8 m/s. Reed's work also recommended that no more than 20% of the clean-bed hydraulic conductivity be used for design purposes.

In 1996, Robert Knight and Robert Kadlec published design recommendations based on an areal-based, first-order equation set with nonzero background concentrations for BOD, total suspended solids (TSS), ammonia, nitrate, total nitrogen, phosphorus, and fecal coliform (Kadlec and Knight, 1996). This approach was later reported in an international guideline (IWA Specialist Group on Use of Macrophytes in Water Pollution Control, 2000).

Kadlec and Knight provided methods for determining hydraulic conductivity based on the grain size of the media, with the recommendation that only 10% of the clean-bed hydraulic conductivity be used for design purposes. Methods to calculate the water surface profile within the wetland were also provided. These methods were applied in a forensic analysis of a HSSF wetland with inlet ponding problems (Kadlec and Watson, 1993).

In 2000, the U.S. Environmental Protection Agency published a new design manual on constructed wetlands

(U.S. EPA, 2000a). This manual provides data on the treatment performance of a few wetland systems. Recommendations included an areal-loading rate of less than 6 g BOD/$m^2$·d to achieve an effluent concentration of less than 30 mg/L, and a loading rate less than 20 g TSS/$m^2$·d to achieve an effluent concentration less than 30 mg/L. Recommended gravel size was 20 to 30 mm. For the first 30% of the wetland bed, no more than 1% of the clean-bed hydraulic conductivity was to be used for design.

In 2006, the Water Environment Research Foundation (WERF) published design guidelines for small-scale HSSF wetlands (Wallace and Knight, 2006). This work substantially expanded on the loading chart concept proposed by U.S. EPA (2000a), by providing statistical confidence bands on the loading charts (although only for a narrow range of loading conditions). The concept of cross-sectional organic loading versus the size of the wetland bed materials was introduced as a means to forecast the clogging potential in the inlet zone of HSSF wetlands.

### Reflections on Old HSSF Design Procedures

Looking back on the development of HSSF design methods, it can be seen that early procedures of sizing HSSF wetlands involved the use of a first-order volumetric rate constant (plug flow, $C^* = 0$) for BOD reduction (Reed *et al*., 1988; Bavor *et al*., 1989; Schierup *et al*., 1990; Cooper 1990). However, in Europe and North America, the vast majority of the applications were for domestic wastewater treatment for small villages or single-family homes (Figure 1.6) for which there was no existing flow or inlet BOD information. Consequently, specifications were driven toward scaling the wetlands based upon population equivalents, which is equivalent to a loading specification. Thus, it is seen that the same two approaches to sizing used for FWS wetlands have also been used for HSSF systems.

The existence of multiple approaches invites comparisons, and the treatment wetland literature does not disappoint. Unfortunately, most such published comparisons are not appropriate. One common theme is to compute outflow concentrations for conditions of a specific (new) data set using the published central tendency rate constant from multiple preceding studies (see, for instance, Liu *et al*. (2000) or Walker and Walker (1990). Of course, it would be an accident if the new data set followed the mean prediction based on a central tendency. The scatter of rate constants among different wetlands has been shown to be quite large in the information presented in Part I (see, for example, Table 8.7) and concentration predictions depend on rate constants in an exponential fashion. A better approach is to compare the calibrated *k*-value from a new data set to the existing, known distribution of such calibration *k*-values.

There has also been a seemingly irresistible urge, in past literature, to either look for or use a single, universal *k*-value. For example, there is an oft-repeated value of $k_V = 1.104$ $d^{-1}$ in U.S. literature, for BOD reduction in HSSF wetlands at 20°C, for $C^* = 0$ (U.S. EPA, 1993c; Reed *et al*., 1995; Crites and Tchobanoglous, 1998; Water Environment Federation, 2001; Crites *et al*., 2006). When plug flow and $C^* = 0$ are presumed, there is only one parameter remaining, which is here designated as $k_1$ (or $k_{V1}$). Although no basis for this number has been presented in the literature, it is now possible to place it in the perspective of calibrations for large numbers of HSSF systems.

This suggested value, for a system 45 cm deep and with 38% porosity, corresponds to $k = 70$ m/year (see Equation 6.62). This ranks at the 80th percentile of $k_1$-values for 200 HSSF wetlands treating primary or secondary wastewaters, so most of the time this particular single-parameter model will overestimate pollutant removals.

Two values of $k_1$ are reported by Cooper *et al*. (1996): $0.06 \times 365 = 22$ m/year for secondary reedbeds, and $0.31 \times 365 = 113$ m/year for tertiary reedbeds. Kadlec and Knight (1996) reported a range of *k* and $C^*$ values. When those same systems are analyzed for $k_1$ values, the range is 20–98 m/year, with a median of 44 m/year. These historical estimates are now supplanted by a much larger worldwide database (Figure 15.8). Although the plug-flow model with $C^* = 0$ (rate constant = $k_1$) should not be used for reasons previously discussed, and it is now clear that historical rate-constant estimates using this model have not stood the test of time as more performance data for HSSF wetlands has become available. Those "old" estimates are for rate constants far higher than those reasonably supported by the modern data set. So, not only is the single-parameter model not particularly suited to HSSF wetlands, but the available calibration estimates do not achieve good data fits, either.

## 15.5 VERTICAL FLOW WETLANDS

Vertical flow (VF) wetlands are a newer type of treatment wetland, having come into widespread use in the mid-1990s. Several variations of the technology exist; pulse-loaded systems (most commonly implemented in Europe), recirculating vertical flow (used in North America but based on old standards for recirculating gravel filters), continuous ponded downflow (for mine water treatment), and fill-and-drain (tidal flow) wetlands.

The treatment performance of all these vertical flow systems is significantly affected by operational factors such as instantaneous loading rates, the frequency of loading cycles, recirculation rates, and the length of resting interval between loading cycles. The relatively short history of technology development and the poorly quantified impact of operational factors on treatment performance have not allowed design methods to evolve beyond the empirical stage.

### Design of Vertical Flow Wetlands in Europe

In 1952, a German scientist, Kathe Seidel, began investigating the water purification capabilities of *Schoenoplectus lacustris* plants grown in artificial-rooting environments. Her discoveries gave birth to modern SSF-constructed wetlands (Börner *et al*., 1998) and led to the development of a wetland

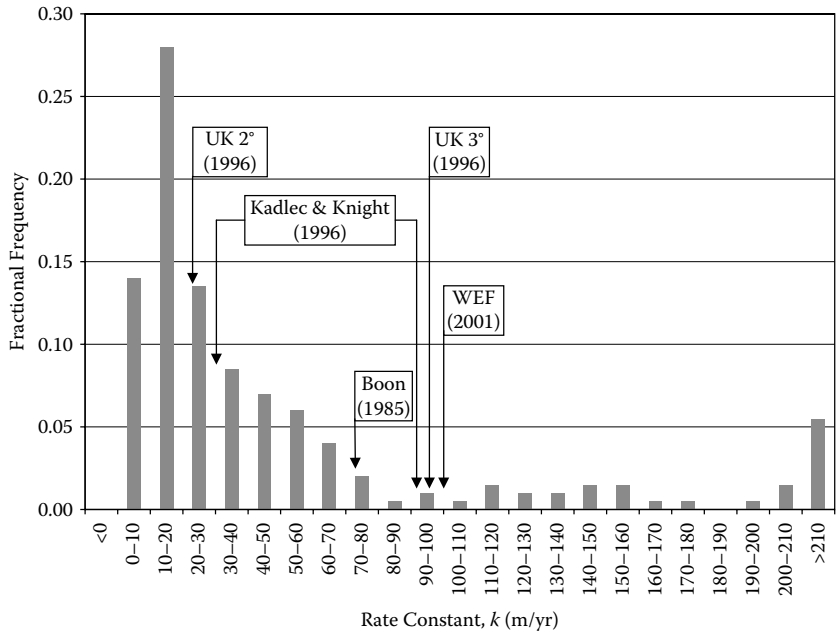



**FIGURE 15.8** Distribution of zero-background, plug flow BOD removal rate constants ($k_1$) for HSSF wetlands. The arrows mark the values from Boon (1985; 70 m/yr), Cooper *et al*. (1996; 22 m/yr for secondary, 113 m/yr for tertiary reedbeds in the United Kingdom), and Kadlec and Knight (1996; these are not the *k-C** values; rather they are $k_1$ values for the same data systems) and the median is 44 m/yr, and WEF (2001; only one number given). The bars represent values from 200 HSSF wetlands that have a median of 24 m/yr.

treatment system known as the Max Planck Institute Process (MPIP), Seidel System, or Krefeld Process (Brix, 1994d), consisting of vertical flow wetland stages followed by HSSF wetland cells (Figure 15.9) (Liénard *et al*., 1990).

In Seidel's system, coarse material (sand and gravel) served as the rooting media for a variety of wetland plants. Vertical flow cells were generally square, and subsequent horizontal flow cells were long and narrow. Plants were thought to be the

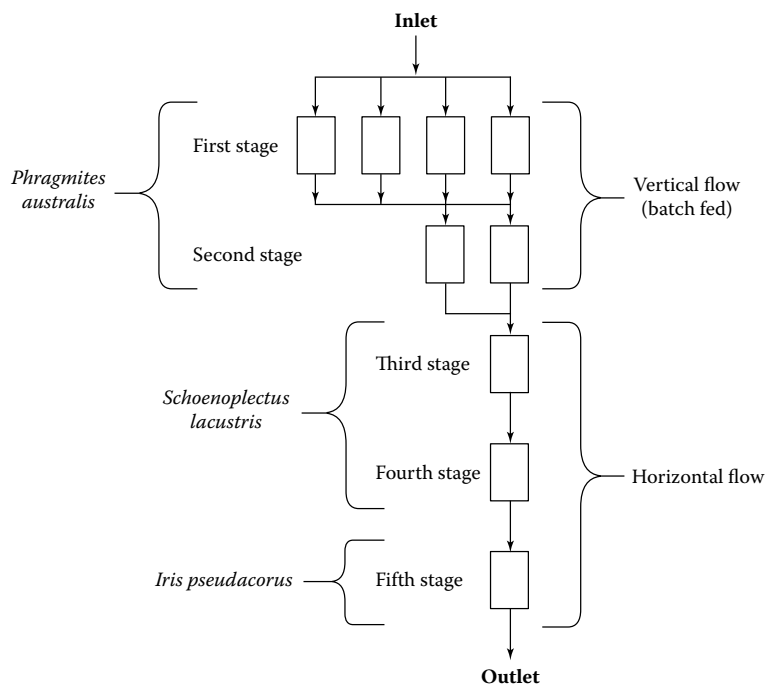



**FIGURE 15.9** Max Planck Institute Process wetland treatment system (plan view). (From Cooper and Findlater (1990) *Constructed Wetlands in Water Pollution Control*. Pergamon Press, New York. Reprinted with permission.)

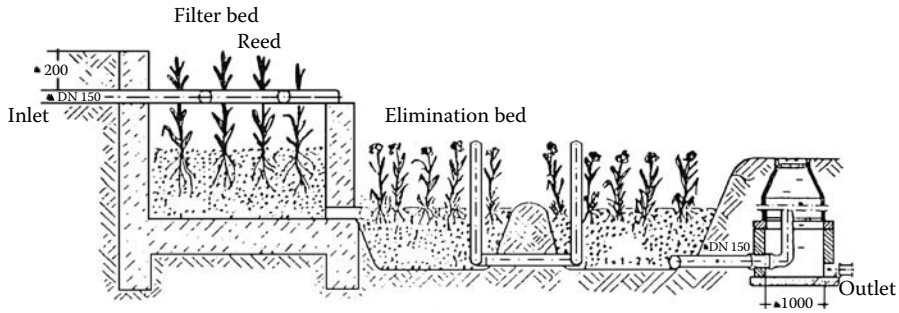



**FIGURE 15.10** Max Planck Institute Process wetland treatment system (profile view). (From Börner *et al*. (1998) In *Constructed Wetlands for Wastewater Treatment in Europe*. Vymazal *et al*. (Eds.), Backhuys Publishers, Leiden, The Netherlands, pp. 169–190. Reprinted with permission.)

main treatment mechanism in this process (Vymazal *et al*., 1998), but those ideas are now disputed (Langergraber, 2005). Figure 15.10 is a profile view of the MPIP process (Börner *et al*., 1998).

The long-term success of MPIP systems in Europe has led to a more widespread application of vertical flow systems based on pulse-loaded designs. These vertical flow cells are almost invariably linked to HSSF wetlands to create "hybrid" wetland systems. The VF–HSSF configuration is the most common, and relates back to the MPIP process, where initial VF wetlands are used to remove TSS and BOD that could create clogging problems in the downstream HSSF stages. However, an alternate configuration of HSSF–VF has been implemented (Brix, 1998), where the initial HSSF wetland is designed for BOD removal, so that nitrification is optimized in the downstream VF stage.

### Loading Specifications

Results from practice have resulted in several "rules of thumb," or scaling rules, that relate sizing of the wetland bed to an implied mass loading, expressed in population equivalents. Loading rates vary based on the implied level of treatment (BOD removal or nitrification). Various parameters have been used to calculate the footprint area of VF-constructed wetlands. Cooper (2005) summarized the most frequently used sizing criteria using the following nomenclature:

$$A = mP^b \tag{15.5}$$

where

$A$ = area of bed required, $m^2$
$b$ = exponent
$P$ = population equivalent
$m$ = scaling factor

Table 15.1 shows the various recommendations for achieving different and often unspecified goals.

### Implied Oxygen Transfer

There have also been attempts to relate VF wetland sizing to the implied oxygen transfer rate (IOTR) necessary for reductions in BOD and total Kjeldahl nitrogen (TKN). However, methods developed to date are based on traditional nitrification/denitrification stoichiometry, which has a higher oxygen demand than alternative nitrogen pathways such as anammox (see Chapter 9). It has often been assumed that reduction of

**TABLE 15.1**
**Scaling Rules for Vertical Flow Wetlands Treating Municipal Wastewater**

| Source | Country | *m* | *b* | Comments |
|---|---|---|---|---|
| Cooper *et al*. (1996) | U.K. | 1 | 1.0 | BOD only |
| Cooper *et al*. (1996) | U.K. | 2 | 1.0 | BOD and $NH_4$-N |
| Weedon (2003) | U.K. | 5.4 | 0.60 | PE < 25 |
| Weedon (2003) | U.K. | 2.4 | 0.85 | PE > 25 |
| Boutin and Liénard (2003) | France | 2.5 | 1.0 | Two stages |
| Fehr *et al*. (2003) | Germany | 4 | 1.0 | — |
| Brix and Johansen (2004) | Denmark | 3 | 1.0 | 95% BOD reduction |
| Molle *et al*. (2005a) | France | 2 | 1.0 | 1.2 in 1st stage; 0.8 in 2nd stage; COD < 60 |
| ÖNORM (2005) | Austria | 4 | 1.0 | — |
| DWA (2006) | Germany | 4 | 1.0 | — |
| Langergraber *et al*. (2006a) | Austria | 4 | 1.0 | — |

BOD is strictly through aerobic mechanisms, although anaerobic reduction is also possible. Also, ammonia produced by ammonification of organic matter is typically not accounted. Further, BOD is utilized in the reduction of oxidized nitrogen, thus relieving the need for extra oxygen transfer to reduce BOD. Conversely, BOD is usually $BOD_5$, which is only about two thirds of the ultimate BOD. As a result, the implied oxygen demand calculated in the literature from these methods should be not considered the same as the actual oxygen transfer rate.

A rough estimation of the implied total oxygen transfer rate across vertical flow stage was presented by Cooper (1999):

$$\mathrm{IOTR} = \left(\frac{Q}{A}\right)\left[(\mathrm{BOD_{in}} - \mathrm{BOD_{out}}) + 4.3(\mathrm{NH_4N_{in}} - \mathrm{NH_4N_{out}})\right] \tag{15.6}$$

where

$A$ = bed area, $m^2$
$BOD_{in}$ = inlet BOD, mg/L
$BOD_{out}$ = outlet BOD, mg/L
IOTR = implied oxygen transfer rate, $gO/m^2 \cdot d$
$NH_4$-$N_{in}$ = inlet $NH_4$-N, mg/L
$NH_4$-$N_{out}$ = outlet $NH_4$-N, mg/L
$Q$ = flow rate, $m^3/d$

Cooper (1999) recognized that this approximation does not allow for BOD removal by settlement/filtration or denitrification, nor for $NH_4$-N lost through (a) plant uptake, (b) volatilization, or (c) adsorption. Additionally, the other factors enumerated in the preceding text are not accounted. The stoichiometric factor of 4.3 used to estimate the $O_2$ needed for ammonia oxidation comes from work on nitrification in activated sludge systems. The author pointed out that $NH_4$-N for a fresh sewage is also an underestimate of the potential $NH_4$-N when all the organically bound nitrogen in urea has been hydrolyzed. Therefore, TKN is a better measure but is not often reported, especially in the European literature.

Platzer (1998; 1999) used an equation similar to Equation 15.6 in which ($BOD_{in}$–$BOD_{out}$) was replaced by 0.7 ($COD_{in}$–$COD_{out}$). Platzer also took account of the $O_2$ recovered by denitrification. The IOTR was 23 g $O_2$ $m^2 \cdot d^{-1}$ for treatment units with a low-hydraulic loading rate and 64 g $O_2$ $m^2 \cdot d^{-1}$ for units with a high-hydraulic loading rate. Platzer (1999) pointed out that in the design of VF beds for nitrification, the most important criterion is that total oxygen input has to be higher than the oxygen demand. The oxygen demand is the sum of OD for decomposition of organic material and OD for nitrification. Platzer (1999) went on to estimate oxygen added by water and induced air flow, but there was no confirmation of these estimates beyond order-of-magnitude agreement with observations.

### Computer Models

A computer model, CW2D, has been developed to predict fate and transport of pollutants in pulse-loaded vertical flow wetlands (Langergraber, 2001). The CW2D is based on the HYDRUS-2D flow model (Simunek *et al*., 1999), which can simulate both saturated and unsaturated flow, and pollutant transformations are assumed to follow the activated sludge model No. 2 (Henze *et al*., 1995). The model contains 12 components and 9 processes. Various calibration studies have been performed (Meyer *et al*., 2006; Sleytr *et al*., 2006; Langergraber *et al*., 2006a; Tietz *et al*., 2006), but the model has not been widely implemented to date as a design tool.

## 15.6 COMMON DESIGN MISUNDERSTANDINGS

### Area- and Volume-Based Rates

Chapter 6 provides a discussion of the two forms of the *P-k-C** model:

$$\frac{(C - C^*)}{(C_i - C^*)} = \underbrace{\left(1 + \frac{ky}{Pq}\right)^{-P}}_{\mathbf{A}} = \underbrace{\left(1 + \frac{k\tau y}{Ph_F}\right)^{-P}}_{\mathbf{B}} = \underbrace{\left(1 + \frac{k_V \tau y}{P}\right)^{-P}}_{\mathbf{C}} \tag{15.7}$$

where

$C$ = concentration at fractional distance $y$, $g/m^3$
$C_i$ = inlet concentration, $g/m^3$
$C^*$ = background concentration, $g/m^3$
$h_F$ = wetland free-water depth, m
$k$ = first-order areal constant, m/d
$k_V$ = first-order volumetric rate constant, $d^{-1}$
$P$ = apparent number of TIS
$q$ = hydraulic-loading rate, ($= h_F/\tau$) m/d
$y$ = fractional distance, unitless
$\tau$ = nominal detention time, days

In these, it is clear that:

$$k_V = \left(\frac{k}{\varepsilon h}\right) = \left(\frac{k}{h_F}\right) \tag{15.8}$$

where

$\varepsilon$ = wetland porosity, dimensionless
$h_F$ = wetland water depth, m

It should be clear that there is no fundamental difference among Equations 15.7A,B,C other than the presumption of the depth variability of the rate constant—which is important. However, some incorrect interpretations of Equation 15.7A have appeared in the following literature.

1. " … detention time is not considered in the model" (Water Environment Federation, 2001). Of course it is; hydraulic loading is just free water depth divided by detention time.
2. " … water depth is not considered in the model" (Water Environment Federation, 2001). Water depth

is in fact considered in a better way than for volumetric models, because $k$ is more invariant than $k_V$ in most cases, as documented in Part I.

3. " … water losses and gains are lumped into the rate constant" (U.S. EPA, 1999). No more so than for the volumetric version. If one chooses to account for flow augmentation by averaging flows in and out to obtain an average detention time (an incorrect procedure—see Chapter 2), then the value of $q = h_F/\tau$ also changes accordingly. In Kadlec and Knight (1996), the correct modifications for augmented flow situations are clearly spelled out.
4. " … because of the internal position of $C^*$, it is not possible to design a system large enough to achieve an effluent with (less than) background concentrations." (Crites *et al*., 2006) True. The alternative is to choose a model that predicts a level of treatment performance that cannot be sustained by the wetland.

### Temperature Coefficients

One of the most baffling aspects of much of the treatment wetland literature is the way in which temperature effects are (not) quantified. To explain, consider the reduction of BOD in FWS wetlands. In the very early days of the emergence of constructed wetland technology, temperature coefficients were "borrowed" from other technologies such as trickling filters. These first guesses were that $\theta = 1.09$ (Tchobanoglous, 1987) or $\theta = 1.1$ (Reed *et al*., 1988; U.S. EPA, 1988b). However, these guesses were subsequently lowered to $\theta = 1.06$ (Crites, 1994; Reed *et al*., 1995; Crites and Tchobanoglous, 1998). However, Kadlec and Reddy (2001) listed 28 theta-factor determinations for 23 FWS wetlands that showed $\theta \approx 0.98$. Only four of the 28 determinations were greater than unity, and the highest was $\theta = 1.04$. Yet, in the face of so much data to the contrary, some literature sources steadfastly maintain that $\theta = 1.06$ (Crites *et al*., 2006).

### Excursion Containment and Safety Factors

Probabilistic effects are important in the utilization of design models for predicting removal performance of treatment wetlands. Regulatory requirements often employ a standard other than a long-term average. There may be a maximum monthly concentration not to be exceeded, or a specified concentration not be exceeded for more than a certain percentage of samples. Addressing wetland system variability in the design process is discussed in further detail in Chapter 16.

## 15.7 A CRITIQUE OF DESIGN METHODS

There are several reasons why existing design techniques for treatment wetlands can no longer be regarded as satisfactory if these are used in isolation. Nevertheless, there are very useful attributes of the two most prevalent approaches, and these will be the backbone of a broader approach presented here. The two approaches are loading specification and exponential decline modeling, both with variations.

### Loading Specification

These sizing methods are area-based scaling procedures in which either the hydraulic load or the pollutant load is set to values believed to meet some performance criteria. Common variants include:

A. For domestic wastewater treatment, the per-person area requirement may be set. Depending upon treatment needs, values in the range 5–20 $m^2$/PE are found in the literature. Domestic daily water usage varies from 100–400 L/PE·d and, hence, the equivalent hydraulic-loading rate to the wetland is 0.5–8 cm/d. The presumptive inlet water quality is that of settled sewage, and the presumptive treatment goal is something similar to secondary quality (30 BOD, 30 TSS). If the inlet BOD is 200 mg/L and personal volume is 300 L/PE·d, and the design is to 10 $m^2$/PE, the BOD loading is 60 kg/ha·d. The specification of square meters per-person equivalent is a traditionally popular method in Europe.
B. For municipal wastewater treatment, the inlet pollutant loading to the wetland may be set. The implied inlet water quality is that produced by a facultative pond or roughly 50–100 mg/L BOD. The presumptive treatment goal is usually slightly better than secondary, 20–30 mg/L of BOD and TSS. This method has been advocated by U.S. EPA (2000a) with criteria of 45–60 kg/ha·d BOD and 30–50 kg/ha·d TSS to achieve these levels.
C. For wastewater polishing of secondary or better effluents, there may be regulatory hydraulic- or nutrient-loading criteria. For example, the state of Florida requires less than 75 g/$m^2$·year total nitrogen (TN) and less than 9 g/$m^2$·year TP for waters treated in impaired natural wetlands.
D. Nominal detention time is an intuitive design specification: the longer the water stays in the wetland, the more treatment it will receive. FWS wetlands often have nominal detention on the order of a week. In some instances, regulation controls or constrains allowable designs. For example, the state of Indiana constructed wetland guidance document recommends a minimum of five days' detention (Indiana Department of Environmental Management, 1997), with agency approval required for shorter detention time. The province of Alberta constructed wetlands guidelines state that a detention time of at least seven to ten days is necessary (Alberta Environment, 2000). South Dakota states a minimum recommended detention time for treatment in an artificial wetland system of seven days, with 14 days being the most desirable (SDDENR, 1991).

E. The inlet hydraulic loading to the wetland is simply related to the nominal detention time: it is the mean free water depth divided by the detention time (see Part 1, Chapter 2). Many FWS wetlands have 30–40 cm of free water depth. Consequently, seven to ten days detention corresponds to 3–6 cm/d inlet HLR. Regulatory requirements for maximum HLR sometimes are set; for example, the State of Florida requires less than 2.2 cm/d HLR for waters treated in impaired natural wetlands.

## Loading-Based FWS Sizing—An Example

The following example is borrowed from U.S. EPA (2000a) and represents only the BOD portion of the sizing.

A FWS is to treat an average flow of 18,920 $m^3$/d of lagoon effluent, with an average BOD = 50 mg/L, to an effluent concentration of BOD = 30 mg/L. What size wetland is needed? (1 mg/L = 1 $g/m^3$)

Design rule: a maximum areal-loading rate of

$$40 \text{ kg/ha·d } (= 4 \text{ g/m}^2\text{·d})$$

Design calculation: $$\frac{(18{,}920\,\text{m}^3/\text{d})(50\,\text{g/m}^3)}{A} < 4\,\text{g/m}^2\text{·d};$$

Therefore, $$A > 236{,}500\ \text{m}^2 = 23.65\ \text{ha}$$

U.S. EPA (2000a) goes on to modify this area for a peak-flow factor and for open-water area inclusion, but the foregoing calculation is the central concept.

## Loading-Based FWS Sizing—Difficulties

These simplistic single-number rules have great appeal because they are quickly applied and easily understood. And, there is a popular misconception that wetlands and other natural systems should be amenable to intuitive quantification. This is certainly not the case, as has been repeatedly proved in wetland science. The structure and function of wetland ecosystems are very complex and require commensurately complex descriptions. The foundation for loading criteria is generally solid with large numbers of wetlands being considered. However, the single-number rules extracted from that information do not convey important details. Some of the problems with an unembellished loading approach are detailed here, as the basis for added steps in a design process.

1. Inlet loading may, in some circumstances, be an inappropriate correlating variable for the effluent concentration from the treatment wetland. This is almost certainly the case for total suspended solids under long-detention time and for nitrate in many situations. In the case of TSS under light loadings, settling is complete in the inlet zone of the wetland; therefore, solids leaving the wetland are most likely to be generated within the wetland and are not remnants of incoming TSS. Therefore, the TSS in the outflow relates to wetland processes, not to inlet loading. In the case of nitrate, if there is no nitrate in the inflow, then it is generated by nitrification. As a consequence, inlet loading has no relevance for outlet concentration except in those instances where the nitrate is the dominant form of nitrogen in the incoming water.
2. Hydraulic loading alone does not acknowledge either the incoming water quality or the goals of treatment.
3. Wetlands respond differently to concentration changes than they do to flow-rate changes. Pollutant loading by itself does not distinguish these separate effects.
4. Most of the numerous controlling features of a wetland are not present in these specifications. System attributes such as aspect ratio, number of cells, and water depth, and environmental factors such as water temperature, season, rainfall, and evaporation, are not explicitly considered. In sum, such factors combine to produce great variability in performance. Often, the single-number criterion represents only the central tendency of a very wide spread of annual results. It must be remembered that when the central tendency of data is selected as a design criterion, the odds are that the design will fail 50% of the time.
5. It is very easy, and very dangerous, to extrapolate designs into areas far outside the conditions that were used to derive the criterion. For instance, the U.S. EPA (2000a) criteria were derived from limited data from systems receiving a fairly narrow range of inlet concentrations, about 30–100 mg/L for BOD, TSS, and TN. Data for polishing wetlands ($C_i$ < 30 mg/L) and animal waste wetlands ($C_i$ > 100 mg/L) provide quite a different basis.

The loading specification approach may be extended to include several target outlet concentrations and the loadings needed to achieve them. This was suggested for animal waste treatment wetlands with a sliding scale from 20 kg/ha·d to produce 20 mg/L BOD, to 100 kg/ha·d to produce 100 mg/L, on average (CH2M Hill and Payne Engineering, 1997). Ultimately, the central tendency of the load–response graph may be represented by a line, either determined from regression (see Figure 7.18) or from a *P-k-C** model (see Figure 8.10).

Despite the intuitive appeal of these single-number approaches, they are often inadequate for final design. They serve only to place a potential wetland size "in the ballpark." Probably the best use of such criteria is simple scaling of wetlands from one source to another identical source of a different flow rate. However, the load–response graph that underlies loading criteria is a useful tool and plays a major role in the expanded design-sizing process advocated in this chapter.

## Exponential Decline Models

Typically, the procedure has been to assume steady and uniform plug-flow hydraulics and pollutant loadings associated only with added water. Then, first-order, volumetric removal rules without background concentrations were applied for sizing (Reed *et al*., 1995; Crites and Tchobanoglous, 1998). Alternatively, first-order, area-based removal rules with background concentrations were applied for sizing (Kadlec and Knight, 1996; Wong and Geiger, 1997). Presumed or measured temperature coefficients are used to "correct" the rate constants for thermal effects.

Some of the problems with these approaches are:

i. Hydraulics typically are not close to plug flow. Numerous tracer studies have repeatedly, without exception, shown that there is a distribution of detention times (Rash and Liehr, 1999; Carleton, 2002). Even if the wetland is hydraulically efficient, the weathering of a pollutant mixture gives rise to the appearance of hydraulic inefficiency (Kadlec, 2003a).
ii. Pollutant removals are not necessarily volume-specific. More often, removal is area specific, resulting in rate "constants" that decrease with increasing depth (Gearheart *et al*., 1989; Kadlec, 1997b).
iii. Removals for some pollutants, such as nitrogen species, reflect the effects of internal spatially distributed loadings. Decomposing wetland litter and sediments thus create an internal loading of some substances, notably BOD, TSS, nitrogen, and phosphorus, by virtue of the large wetland biogeochemical cycle. The result is a nonzero wetland background concentration within and exiting the wetland (Kadlec and Knight, 1996; IWA Specialist Group on Use of Macrophytes in Water Pollution Control, 2000).
iv. Flows are not steady. Many phenomena lead to time variation in flows and concentrations, such as rainfall, evaporation, and daily wastewater generation cycles. These effects are especially important for event-driven wetlands used for stormwater treatment (Wong and Somes, 1995; Werner and Kadlec, 2000b).
v. Flows are not uniform. Rainfall and evapotranspiration lead to spatially variable flows and the attendant-dilution effects (Kadlec, 1989c; Platzer and Netter, 1994).
vi. Rate constants may display either seasonal or temperature dependence or be relatively uniform through the year (Kadlec and Reddy, 2001). Temperature is not always an acceptable surrogate for seasonality for many ecosystem functions. For instance, vegetative uptake in temperate climates is maximum during the spring period, at moderate temperatures; but release via decomposition is maximum during fall, also at moderate temperatures.
vii. Rate constants are sometimes derived from synoptic input–output data, which ignores the important transport delays inherent in long-detention systems.
viii. Rate constants are not constant for a specific wetland. Unavoidably, they vary with depth, pollutant concentrations, and hydraulic loading (Kadlec, 2000).
ix. Rate constants are not constant across all wetland types. Removal-rate constants depend on vegetation type and density, and on substrate size, type, and condition.
x. The first-order concept is valid only within an operating region for which there are no constraints on supplies of critical ingredients for the removal process. For instance, nitrification can be first order in ammonium only when there is sufficient oxygen available.
xi. Removal equations give deterministic predictions. The stochastic component of treatment wetland performance is often equally important and effectively masks deterministic effects over a short-operational period. Results for several years need to be overlain to establish the seasonal trend, if any.

Despite these short-comings, rate-constant models have advantages. They separate the effects of hydraulic loading and inlet concentration. They allow prediction of the gradient in pollutant concentration along the flow direction. They allow for background concentrations. They are easily applied over all water temperatures and seasons. They allow for extrapolation from small data sets, such as pilot studies. Therefore, it is desirable to keep the rate-constant approach as one of the tools of design.

The cure for the ills i–xi is theoretically straightforward: develop more-detailed wetland performance models that are capable of characterizing the several important effects noted in the preceding text. Some very detailed models have been proposed, such as an adaptation of the activated sludge model for treatment wetlands (McBride and Tanner, 2000), and spatially distributed, transient, biologically compartmentalized box-and-arrow constructs (Fitz and Sklar, 1999). However, very rarely does any wetland project measure and record all of the important variables that are required to calibrate such complex models—in fact, it has not yet been done satisfactorily.

The strategy advocated here is to utilize the existing (inadequate) data sets to maximum advantage, while retaining as much simplicity as possible.

Items i–v may be accommodated by relatively minor modifications of the first-order deterministic modeling technique. A distribution of detention times can be implemented by altering the spatial water-mass balance that is the foundation for pollutant removal. Area-based rate constants are used. Internal pollutant loadings can be added to the pollutant mass balance. Dynamic mass balances can be easily constructed. Flow nonuniformity can also be added to the water-mass

balance—but not by averaging flows, which can lead to serious errors (Kadlec, 1997b).

The effects of temperature and/or season vi can be deduced from operating data sets, provided a long enough period of record is available to allow for sorting out the competing deterministic and stochastic effects. The synoptic error vii may be avoided by again using a sufficiently long period of operating data and suitable averaging.

Items viii and ix pose greater difficulty because it seems necessary to include more of the unavailable details in the design model. A less satisfying but more practical approach is to acknowledge rate-constant variability and use a large amount of available data to establish the envelope of expectations, and the density of results within it. To that end, a robust graphical display is suggested in the form of a graph of outlet concentrations versus inlet pollutant loadings.

Item x can be dealt with by examination of the details of each specific suite of removal processes with a view to identification of possible mechanistic supply limitations.

Finally, item xi is incorporated by understanding the probability distributions that have been identified for similar wetland systems. This requires detrending the design data sets for effects of water mass balances, season, and temperature.

The foregoing procedures are more complicated than plugging numbers into a single recipe equation, but the risks of straying from the available knowledge envelope are greatly reduced.

## 15.8 A PERFORMANCE-BASED SIZING ALGORITHM

The methods that are used in treatment wetland sizing vary from type-to-type. Rules of thumb are valuable in preliminary assessment, but final decisions must be predicated either on performance requirements or upon regulatory prescriptions. Here, the performance-based approach is outlined, and that is the primary approach for FWS wetlands, which is detailed in Chapter 17. In Chapter 20, the relationship to existing codes and statutes for HSSF system sizing will be analyzed.

Given the shortcomings of previous methods, a strategy is here proposed that attempts to salvage the good features of past work and to improve upon past methods as much as data can support. The strategy described here is not simpler than previous methods because it includes more pieces; that is, it is an integrated rational approach to sizing. It is assumed that the practitioner will implement the steps via spreadsheet on a desktop computer because that has been standard in the engineering profession for many years.

The key features of the procedure are

*Basis:*

1. Set inlet concentrations
2. Set target concentrations (regulatory limits and exceedance factors)
3. Set inflow and seepage
4. Determine rain, *ET,* and temperature
5. Select wetland type (FWS or SSF)

*Parameter selection and calculation:*

6. Select rate constants and seasonality (Select plant community type)
7. Select DTD (hydraulic) efficiency = *P*-value (Select compartmentalization)
8. Adjust area till goals are all met

*Constraint checks and iteration:*

9. Set estimated growth cycle
10. Check biogeochemical cycles for consistency
11. Check constraints (hydraulic and chemical)
12. Check loading graph for risk assessment
13. Adjust for seasonality

*Iterate on procedure as needed*

The first four of these steps involve identifying the basis for sizing. These are fundamental to all wetland types and are the subject of Chapter 16. The next four involve parameter selection and area calculations. The last five are cross-checks followed by adjustments to parameter selection as necessary to preserve a desired degree of risk. These nine steps are the subject of Chapter 17.

## 15.9 GUIDELINES AND MANUALS

In the early years of treatment wetland technology, the designer had access to only a limited number of guidance documents. The amount of information available to, or utilized by, the various authors was extremely limited. The histories of the various pioneer systems were short, and guesses were too frequently made based upon the judgment of the writers. Although some elements of the early guidance and evaluation documents have survived the test of time, a good many were on very shaky ground. Treatment wetland data and analysis has grown, and continues to grow, at a double-exponential rate. The number of applications has increased greatly to include many new areas such as remediation, animal waste treatment, and many specialized industries. But that has been compounded by the rapidly expanding number of systems in each of those application areas. As a result, the knowledge base has been expanding with an estimated doubling time of perhaps three to five years. The arithmetic then tells us that about 95% of what is known today resulted from studies of the past 15 years. With no criticism intended (although some is perhaps deserved), any summaries or guidelines older than 15 years cannot be regarded as reliable and should no longer be used. In the United States, such early documents include:

### Obsolete Manuals

Chan E., Bursztynsky T.A., Hantshe N., Litwin Y.J. (1982) *The Use of Wetlands for Water Pollution Control*, EPA/600/2-82/036.

Cooper P.F. (Ed.) (1990) *European design and operations guidelines for reed bed treatment systems*, WRc: Swindon, United Kingdom.

Hyde H.C., Ross R.S., Demgen F. (Eds.) (1984) *Technology Assessment of Wetlands for Municipal Wastewater Treatment*, EPA/600/2-84/154; NTIS No. 85-106896.

Reed S.C., Crites R., Middlebrooks E.J. (1988) *Natural Systems for Waste Management and Treatment.* 1st ed., McGraw-Hill, New York.

U.S. EPA (1983b) *Design principles for wetland treatment systems*, EPA 600/2-83/026, Hammer D.A., Kadlec R.H. (Eds.), National Technical Information Service.

U.S. EPA (1985) *Freshwater Wetlands for Wastewater Management Handbook*, EPA/904/9-85/135, Atlanta, Georgia.

U.S. EPA (1987) *Report on the use of wetlands for municipal wastewater treatment and disposal*, EPA 430/09-88/005, U.S. EPA Office of Water.

U.S. EPA (1988b) *Design manual: Constructed wetlands and aquatic plant systems for municipal wastewater treatment*, EPA 625/1-88/022, U.S. EPA Office of Water: Cincinnati, Ohio.

U.S. EPA (1993f) *Subsurface flow constructed wetlands for wastewater treatment: A technology assessment*, EPA 832/R-93/001, U.S. EPA Office of Water.

Water Environment Federation (1990) *Natural Systems for Wastewater Treatment (WEF Manual of Practice FD-16). Chapter 13: Wetland Systems.* 1st ed., Water Environment Federation, Alexandria, Virginia.

With the proliferation of knowledge has come a comparable proliferation of guidelines and manuals. These have originated at all levels of government in North America—cities, states and provinces, and national. This situation is comparable around the globe, with many countries having promulgated design guidelines. To compound the complexity, there are now manuals for individual application areas such as animal wastewater, urban stormwater, and onsite treatment of domestic wastewater. These guidance documents are separate from the various prescriptions for treatment wetland design that have been issued by various levels of government, which will be discussed further in Chapter 20. In general, these documents bear later publication dates than the "old" sources, but the reader is cautioned that, in a number of cases, these "new" documents, in fact, simply reiterate the "old" material, and may therefore be regarded as obsolete.

The reader is encouraged to consult those manuals that are pertinent to the application and geographic region for any specific project, but to be very careful about accepting the pronouncements therein. It is important to separate factual bases from opinions, for which purpose the material in Part I may be used. The listing of manuals that follows is inclusive of the majority of national documents but only illustrative of regional, state, and city publications.

## Current Manuals

### National and International

Bendoricchio G., Dal Cin L., Persson J. (2000) *Guidelines for Free Water Surface Wetland Design*, EcoSys Bd. 8, University of Padua: Padua, Italy.

Cooper P.F., Job G.D., Green B., Shutes R.B.E. (1996) *Reed beds and constructed wetlands for wastewater treatment*. WRc Publications. 184 p. plus data diskette: Swindon, United Kingdom.

DLWC (1998) *The Constructed Wetlands Manual (two volumes)*, Department of Land and Water Conservation (DLWC) Information Centre: Sydney, New South Wales, Australia.

Interstate Technology and Regulatory Council (2003) Technical and Regulatory Guidance Document for Constructed Treatment Wetlands. http://www.itrcweb.org/WTLND-1.pdf.

IWA Specialist Group on Use of Macrophytes in Water Pollution Control (2000) *Constructed Wetlands for Pollution Control: Processes, Performance, Design and Operation.* Kadlec R.H., Knight R.L., Vymazal J., Brix H., Cooper P.F., Haberl R. (Eds.), IWA Publishing, London.

TVA (1993) *General design, construction, and operation guidelines: Constructed wetlands wastewater treatment systems for small users including individual residences;* 2nd ed., TVA/WM-93/10, Steiner G.R., Watson J.T. (Eds.), Tennessee Valley Authority (TVA) Resource Group Water Management: Chattanooga, Tennessee.

U.S. Army Corps of Engineers (2000) *Wetlands Engineering Handbook*, Report No. ERDC/EL TR-WRP-RE-21, Hayes D.F., Olin T.J., Fischenich J.C., Palermo M.R. (Eds.), U.S. Army Engineer Research and Development Center: Washington, D.C.

U.S. EPA (2000a) *Constructed wetlands treatment of municipal wastewaters*, EPA 625/R-99/010, U.S. EPA Office of Research and Development: Washington, D.C.

U.S. EPA (2000c) *Guiding principles for constructed treatment wetlands: Providing water quality and wildlife habitat*, EPA 843/B-00/003, U.S. EPA Office of Wetlands, Oceans, and Watersheds.

Wallace S.D., Knight R.L. (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements*, Final Report, Project 01-CTS-5, Water Environment Research Foundation: Alexandria, Virginia.

Water Environment Federation (2001) *Natural Systems for Wastewater Treatment (WEF Manual of Practice FD-16).* 2nd ed., Water Environment Federation, Alexandria, Virginia.

### City, State, and Regional

Alberta Environment (2000) *Guidelines for the approval and design of natural and constructed treatment wetlands for water quality improvement*, Alberta Environment Municipal Program Development Branch: Edmonton, Alberta.

Amell B., Chu A., White J. (2004) *Constructed Wetlands for Water Quality Improvement: A Design Primer for the Development Industry*, Report to the City of Calgary, June 2004.

Arizona Department of Environmental Quality (1995) *Arizona guidance manual for constructed wetlands for water quality improvement*, ADEQ TM 95-1, Knight R.L., Randall R., Girts M., Tress J.A., Wilhelm M., Kadlec R.H. (Eds.).

Indiana Department of Environmental Management (1997) *Constructed wetland wastewater treatment facilities guidance*, Water-001-NPD.

Korkusuz E.A. (2005) *Manual of Practice on Constructed Wetlands for Wastewater Treatment and Reuse in Mediterranean Countries*, Added Value Knowledge Report No. 5 (INCO-CT-2003-502453), Agbar Foundation and MED-REUNET (Mediterranean Network on Wastewater Reclamation and Reuse).

LEC (2000) *Draft guidelines for constructed stormwater wetlands*, Report by Lilley Environmental Consulting (LEC) for the City of Edmonton Drainage Service.

Pineywoods RC&D (2001) *Constructed Wetlands in East Texas (4 volumes).* Pineywoods Resource Conservation and Development Council (RC&D), Nagodoches, Texas.

SDDENR (1991) *Recommended Design Criteria Manual, Wastewater Collection and Treatment Facilities*, South Dakota Department of Environment and Natural Resources (SDDENR): Pierre, South Dakota.

Sinclair Knight Merz (2000) *Guidelines for using free water surface constructed wetlands to treat municipal sewage*, DNRQ00047, Queensland Department of Natural Resources: Brisbane, Australia.

### Application Areas

Center for Watershed Protection (2001) *New York State Stormwater Management Design Manual*, Report by the Center for Watershed Protection (CWP) for the New York State Department of Environmental Conservation, Albany, New York.

CH2M Hill, Payne Engineering (1997) *Constructed Wetlands for Animal Waste Treatment*, Report to the Gulf of Mexico Program Nutrient Enrichment Committee, U.S. EPA: Stennis Space Center, Mississippi.

Knight R.L. (2004) *Use of constructed wetland effluent treatment systems in the pulp and paper industry*, Technical Bulletin No. 876, National Council for Air and Stream Improvement, Inc. (NCASI): Research Triangle Park, North Carolina.

Lawrence I., Breen P.F. (1998) *Design Guidelines: Stormwater Pollution Control Ponds and Wetlands*, The Cooperative Research Centre for Freshwater Ecology: Canberra, Australia.

McBrien M. (2000) *The Wisconsin Stormwater Manual: Artificial Wetland Storm Water Management*, University of Wisconsin Cooperative Extension Publishing: Madison, Wisconsin.

PIRAMID Consortium (2003a) *Engineering guidelines for the passive remediation of acidic and/or metalliferous mine drainage and similar wastewaters*, European Commission 5th Framework RTD Project no. EVK-CT-1999-000021, "Passive *In-Situ* Remediation of Acidic Mine/Industrial Drainage" (PIRAMID), University of Newcastle Upon Tyne: Newcastle Upon Tyne, United Kingdom.

Schueler T.R. (1992) *Design of Stormwater Wetland Systems: Guidelines for Creating Diverse and Effective Stormwater Wetlands in the Mid-Atlantic Region*, Metropolitan Washington Council of Governments: Washington, D.C.

Tanner C.C., Kloosterman V.C. (1997) *Guidelines for Constructed Wetland Treatment of Farm Dairy Wastewaters in New Zealand*, NIWA Science and Technology Series No. 48, National Institute of Water and Atmospheric Research (NIWA): Hamilton, New Zealand.

Wong T.H.F., Breen P.F., Somes N.L.G., Lloyd S. (1999) *Managing Urban Stormwater Using Constructed Wetlands*, Cooperative Research Centre for Catchment Hydrology, Monash University: Clayton, Victoria, Australia.

## SUMMARY

Treatment wetlands have taken their place among accepted technologies for water quality management. Many thousands of engineered wetlands are currently operational worldwide to treat many types of wastewaters. These treatment wetlands range in size from a few square meters to thousands of hectares. They treat flows from single-family residences up to entire stormwater volumes from large watersheds. They have been engineered for effective control of a diversity of pollutants, ranging from domestic to agricultural to industrial waste streams. They have also been proven, in many cases, to be the most economical method for achieving water quality treatment goals. In spite of their widespread acceptance for water quality improvement, treatment wetlands are still designed using dozens of different methods. Some designers use rule-of-thumb approaches to size wetlands to meet permitted effluent limits. Others have favorite equations that are based on nonwetland technologies. Many designers build one size to fit all circumstances with no regard to sizing the wetland for an anticipated level of treatment. In addition to the nonconformity of sizing methods, wetland cell configuration varies widely between different designers. Some designers make long, narrow wetland cells. Others use many cells in place of a smaller number of larger cells. Some engineers design treatment wetland cells to operate with deep water, whereas others design with shallow water flows. Other areas of diversity in treatment wetland design include levee construction, types of water control structures, and plant selection.

All of these design approaches are not equal. Some treatment wetlands are too small and do not meet their water quality goals. Many that do meet goals are larger than necessary, and therefore do not meet goals of economy. Some treatment wetland applications (e.g., surface versus subsurface flow) may not be optimal for an existing application. Because designers certainly learn from their mistakes, the impacts of their errors frequently ripple out to other projects and applications. Regulatory officials who have been confronted by a series of failed projects are leery of permitting additional systems, even if proposed by different design engineers. A better, more defensible approach for practitioners embarking in this field is to learn from the operational history of the technology and not to repeat mistakes. The strategy advocated in this book is to utilize the best existing (admittedly incomplete) data sets to maximum advantage, while retaining as much simplicity as possible. The procedures are more complicated than plugging numbers into a single-design recipe equation, but the risks of straying from the available knowledge envelope are greatly reduced. Design may either be performance based or technology based, depending upon whether the performance expectations are explicit in an operating permit or implicit in a set of statutory design specifications. Large systems are often in the former group, whereas small on-site and urban stormwater systems are in the latter.

# 16 Design Basis

The preliminary step in the design of a treatment wetland is to acquire a fundamental understanding of the site of the wetland. Site conditions dictate the physical, chemical, and biological environment of a wetland treatment system. Conditions that should be evaluated during planning of a wetland treatment system include climate, geography, groundwater and its chemistry, soils and geology, rainfall and runoff water chemistry, biology, and socioeconomic factors. The importance of each of these conditions may vary, but all should be investigated to some extent. Detailed studies may be needed to determine the importance of those site conditions that affect technical feasibility.

This book primarily considers performance-based design algorithms. The first steps in the process require the assembly of the basis of the design, which includes the following steps:

1. Determine inlet concentrations and flows.
2. Determine target concentrations (regulatory limits and allowable exceedance factors).
3. Determine allowable inflow and seepage rates.
4. Determine rain, *ET*, and temperature ranges for the project site.
5. Select wetland type (FWS or SSF).

Often, the establishment of flows and concentrations will require the acquisition of data on flows and concentrations, at least to confirm estimates based on prior operations or knowledge of the technology of the source.

There are, unfortunately, numerous examples of inappropriate selection of the design basis for treatment wetlands. The difficulty is often not the misunderstanding of current conditions, but rather incorrect assumptions about future conditions. This may involve actions outside the control of the designer.

For instance the "failed" Gustine, California, system added wetlands to existing lagoons (Walker and Walker, 1990). The source water was a combination of municipal and milk processing wastewaters, the latter having very high biochemical oxygen demand (BOD). The design presumption was that milk wastewater would be discontinued, but this did not occur. Instead of the design influent BOD of 60 mg/L, the wetland actually experienced BOD of approximately 600 mg/L. The wetland could not meet design targets for the unplanned tenfold-higher inlet concentrations. Another example is the "successful" wetland system treating potato wastewater at Connell, Washington (Kadlec *et al*., 1997; Burgoon *et al*., 1999). The system was built according to a design based on operating data from a fixed-capacity processing plant. However, coincident with wetland start-up, the plant implemented water conservation. The loads of pollutants remained the same, but concentrations went up considerably as flows decreased. Fortunately, in this case the wetland system was robust enough to accommodate the change and still achieve goals. These anecdotes serve notice that the basis of design must be carefully set forth, and reasonable changes anticipated in influent flows and loads.

## 16.1 PROJECT SETTING

### Space Considerations: Limited versus Unlimited Space

Free water surface (FWS) treatment wetlands are in the category of land-extensive technologies. At the end of this chapter, it will be seen that horizontal subsurface flow (HSSF) wetlands for the same purpose are not much different in size. However, the site conditions—primarily property boundaries and topography—can limit the potential size of a treatment wetland for a particular source-water volume. This is particularly true of urban stormwater wetlands, which need to be sited in built-up areas and which often utilize high-value lands. Siting of wetlands inside the boundaries of major cities, such as Orlando, Florida (Palmer and Hunt, 1989), or Toronto, Ontario (Helfield and Diamond, 2004), means that the size of the wetland is dictated by existing streets, highways, and buildings, not by the size needed to achieve a particular performance goal.

Topography may also limit the potential size of a treatment wetland. The presence of steep slopes adjacent to the site can preclude construction beyond a certain limit, defined by the practicality of earth moving (Figure 16.1). When the site demands, the treatment wetland may be established in terraces, with elevation drops occurring between the successive cells of the system (Navarra, 1992; Inman *et al*., 2003).

Land ownership can also constrain opportunities for wetland construction. Above and beyond questions of acquisition costs, there is the issue of the willingness of the owner to sell property. Building a treatment wetland rarely falls into the category of eminent domain acquisition, although that has happened in connection with the phosphorus removal wetlands of South Florida. For large wetlands, suitable parcels are often already in agricultural use. Wetlands are frequently viewed as valuable landforms across the regional landscape, but aquatic and terrestrial landforms are also valuable. The construction of a treatment wetland implies the removal of other types of plant, animal, and human communities. Therefore, competing uses may block the construction of a wetland on a particular plot of land.

Perhaps the most serious potential constraint of preexisting landform is the presence of naturally occurring wetlands on property under consideration for a treatment wetland. In



**FIGURE 16.1** The treatment wetlands in the Tucush valley of the high Andes Mountains of Peru (4,100 masl) are constrained to a fixed area by extremely steep slopes. They treat the drainage coming from the wasterock dump of a mine operation. (Photo courtesy Compañía Minera Antamina. Reprinted with permission.)

the United States, it is generally not allowed to build any project that destroys existing wetlands. But what if the project is a constructed wetland? That situation is obviously confusing and unclear, and it is therefore not surprising that a variety of rules and regulations apply in various states.

Sometimes the constructed treatment wetland may be viewed as self-mitigating; that is, it inherently compensates for the loss of preexisting wetlands. That situation has occurred at the West Jackson County, Mississippi, constructed wetland site. It is probably most acceptable when the preexisting wetland is degraded, and of low regional value. However, in many other circumstances, construction in wetlands must be avoided. For instance, HSSF and vertical flow (VF) wetlands do not offer the same type of habitat that occurs in natural wetlands, and construction of these systems in natural wetlands is often blocked by regulatory constraints. Of course, the extreme circumstance is the use of natural wetlands for wastewater treatment, which is outside the scope of this book.

For these area-constrained situations, the design methods described herein are not used to select wetland area, but rather are used to forecast performance of the available wetland area. This predictive mode is readily accommodated in a rate coefficient approach, but is very awkward, if not impossible, for a loading design approach due to the data scatter inherent in loading charts.

## Soils and Geology

For planning purposes, site soils in the United States can be characterized by using USDA Soil Conservation Service soil surveys, which are generally available for most counties within the United States. Other countries often have similar mapping resources. Soil surveys typically include maps of soil types as well as summaries of soil properties, groundwater conditions, climatic information, and plant community information.

Soils are classified by soil scientists based on a complex array of physical and chemical characteristics. Soil information that might be important during project planning includes the presence of hydric soils, which occur in natural wetlands (even if formerly drained) and could be a potential regulatory constraint for a constructed wetland site; soil texture and composition as a suitable medium for berm construction or for impeding leakage to the groundwater; depth to seasonal high groundwater; and depth to confining layers of clays or rock horizons. On-site soils are typically preferred for the rooting media in FWS wetlands. In some cases, the sorption potential of these rooting soils will be a design variable, such as for metal removal.

The construction of wetlands entails the excavation of the wetland basin, including any deep zones, possibly together with conveyance and seepage interception canals. Therefore, the soil thickness above bedrock is an important piece of design information, because that material is movable without blasting. The characteristics of the bedrock are important if such blasting is required (Figure 16.2). Construction in rock is extremely expensive, and is to be avoided if possible.

At the other extreme, on-site materials may be unsuitable for the construction of embankments, because they cannot withstand exposure to the water (Figure 16.3).

## Groundwater

Infiltration of wastewater to the groundwater is important because infiltration affects the wetland water balance and could pose regulatory problems under some conditions. Soil infiltration rates published in soil surveys typically overestimate the actual infiltration rates under sustained, saturated soil conditions and are not reliable for project planning or design. Surface infiltrometer tests or well slug tests provide better estimates of the groundwater leakage that can be



**FIGURE 16.2** Deep zones and canals for South Florida's stormwater treatment areas require blasting of the limestone bedrock. A thin veneer (0.3–1.0 m) of peat overlies the limestone.



**FIGURE 16.3** This collection canal in the Lakeland, Florida, FWS system was built using unstable materials from on site. Despite the attempt to reinforce the embankment with concrete matting, erosion caused the discharge structure to drop into the water.

expected from a full-scale wetland treatment system. Methods for measuring infiltration rates are described by the Soil Conservation Service (SCS) (Hansen, 1980; U.S. Bureau of Reclamation, 1993). Field tests are the most reliable method of estimating groundwater infiltration rates. For constructed wetlands, it may be necessary to construct pilot wetland basins on a proposed site and then instrument inflows and outflows to develop an accurate water balance.

Wetlands can be built on leaky soils as long as regulatory requirements can be met and adequate hydroperiods can be maintained with the wastewater addition and net rainfall. In fact, wetlands have been designed with groundwater recharge as a specific project goal (Ewel and Odum, 1984; Knight and Ferda, 1989). Groundwater infiltration can be eliminated as a project concern for constructed wetlands by using a clay or synthetic impervious liner. Although this approach may not be necessary if the wastewater has received secondary pretreatment, it is recommended when wastewater is less than secondary quality, or is known to contain contaminants of concern for the regional groundwater and its intended uses.

Percolation tests are often used as the basis of sizing infiltration fields for septic tank effluent disposal, although such tests are probably insufficient for ensuring adequate performance of the field (Crites and Tchobanoglous, 1998). The allowable hydraulic loading for the infiltration field is set according to a published table or curve, relating the allowable loading to the time for the water level in a test pit to drop a specified amount (usually 2.5 cm). Data collected from percolation tests are then typically related to a prescriptive hydraulic loading that is usually much less than the observed percolation rate. The reduction in hydraulic loading is to account for the long-term accumulation of microbial biomass and particulate matter in the soil, which substantially reduces the infiltration rate (Tyler and Converse, 1994). Allowable hydraulic loadings are usually in the range of 1–5 cm/d. Additionally, there must be a specified vertical travel distance to the groundwater table, typically about 1 m of unsaturated soil (to allow for the removal of pathogens). These requirements are commonly set forth in local codes and rules, and are enforced as a condition for acceptability of new on-site (septic) systems. These codes are typically intended for single-home treatment systems, but are often extrapolated to larger systems due to a lack of more appropriate regulatory guidelines.

The focus of on-site (septic) system codes is the disposal of primary effluent into the soil matrix. When water is pretreated, organic and pathogen loads are substantially reduced, and soil-based treatment is less critical for regulatory compliance.

Given this basis, constructed wetlands are frequently viewed by the on-site regulatory community as a means for justifying higher loadings or lesser unsaturated travel distances in the infiltration bed, or both. For example, the state of Indiana allows reduction in the size of the absorption field associated with a subsurface-constructed wetland based on the soil loading rate (Indiana Department of Environmental Management, 1997). For soil loading rates less than or equal to 5 cm/d but greater than 2 cm/d, the allowable reduction in field size is 50%. For soil loading rates of less than 2 cm/d but greater than or equal to 1 cm/d, the allowable reduction in the field is 33%. Similar reductions in infiltration area are allowed in other states.

In general, it is beneficial to understand the directions and flows of regional groundwater under the project site. Different levels of hydrogeological surveys may be performed, depending on the requirements of the specific project. Considerable detail is necessary for groundwater remediation wetlands that intercept a plume of contamination, because those studies provide the flows and concentrations needed to determine wetland size or performance. For instance, the design of the Hillsdale, Michigan, project involved multiple monitoring wells, studied over several years, and three-dimensional computational fluid mechanics (Ecology and Environment Engineering, 2004). Modeling at a similar level was necessitated at the Columbia, Missouri, project, because of proximity to the city's potable water well field (Brunner and Kadlec, 1993). If the water leaving the system is transported by unsaturated flow, more complex models will be required (Langergraber, 2001; Davis, 2007).

### Altitude

As the use of treatment wetland technology has grown across the planet, the site conditions have broadened to include a wider range of conditions, among which is the altitude of the project. A few experiences have identified special issues, such as the types of wetland plants that are adapted to high-altitude conditions: *Phragmites* is not a mountain plant! (Navarra, 1992). Other concerns have yet to be explored. For instance, treatment wetlands have now been built at up to 4,000 m above sea level (see Figure 16.1), at which altitude the atmosphere is approximately at half sea-level density. Therefore, the partial pressure of oxygen is half that at sea-level, with potential consequences on the ability of the wetlands to process reactions that require dissolved oxygen, such as nitrification.

### Biological Conditions

The addition of any type of water or wastewater will alter biological conditions at a site. Constructed wetlands frequently replace upland habitats with wetland vegetation. The upland habitats that are lost might include plant communities such as grassland, forest, scrub, desert, or agriculture. The environmental values of these upland habitats should be assessed during project planning. Likewise, wastewater discharge to natural wetlands can cause biological changes of varying magnitudes (see Chapter 3). Existing plant and animal communities in natural wetlands will change depending on the degree of changes to surface water quality and hydrology. Construction-related impacts will result in replacement of part of the existing vegetation by distribution pipes, boardwalks, and monitoring structures. For most constructed wetland projects, site-specific biological conditions do not represent a major technical constraint.

## 16.2 CHARACTERIZATION OF DOMESTIC AND MUNICIPAL WASTEWATER

Wastewater quality varies widely among domestic, municipal, industrial, agricultural, and stormwater categories. Different wastewater sources have unique mixtures of potential pollutants, so that even a single wastewater source category, such as municipal wastewater or urban runoff, may vary considerably depending on local, site-specific circumstances. However, for some chemical constituents, the qualitative and quantitative composition of wastewaters from different sources varies less. In general, any summary of "typical" wastewater concentrations and loads must be considered cautiously.

Site-specific wastewater data showing historical flows and mass loads provide the best information for wetland treatment system design. However, because many treatment systems are designed for new facilities or because historical monitoring may be nonexistent or insufficient, it is useful to know the typical concentrations of major constituents in similar wastewaters. This section summarizes information from a number of sources on the typical pollutant composition of wastewater applied to engineered wetlands. These "typical" concentrations and loads should only be used when site-specific information is not available.

The total municipal wastewater flows from municipal sources undergoing treatment in the United States is $45 \times 10^9$ m$^3$ per year, serving approximately 72% of the population (U.S. EPA, 2007). In addition to industrial and municipal wastewaters, nonpoint source pollution contributes about two thirds of the total pollution load to U.S. inland surface waters (U.S. EPA, 1989). Sources of nonpoint flows include urban and suburban runoff, diffuse agricultural runoff, forestry activities, runoff from concentrated agricultural activities such as feedlots, mine drainage, and runoff from undisturbed areas. However, in certain areas urban runoff or other stormwater sources provide the greatest percentage of uncontrolled pollutants. Wetlands are often used in conjunction with other treatment devices, including septic tanks, lagoons, and mechanical treatment plants (Figure 16.4). In those circumstances, the water quality of interest for the wetland design is that exiting a pretreatment step.

The amount and timing of the water to be treated is the first and foremost item of the design basis. This information should include the possible seasonality of flows and the anticipated progression of flows over the life of the design. This is more important for treatment wetland design than for conventional concrete and steel treatment plants, because of the implied life cycle of the process and the nature of urban and industrial growth. It is customary to plan for a 20-year life expectancy for conventional wastewater treatment plants, because mechanical equipment often wears out during this period. But wetlands clearly can continue to function for far longer periods than two decades; for example, there are receiving wetlands that have been in operation for periods of 70 years (Great Meadows; Yonika *et al.*, 1979) and 90 years

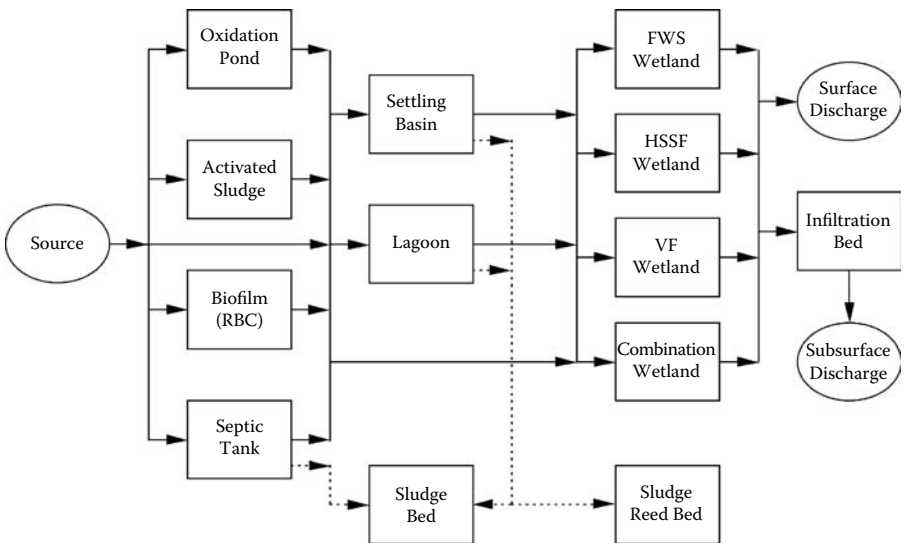



**FIGURE 16.4** Simplified options for treatment trains involving treatment wetlands.

(Brillion; Spangler *et al*., 1976). Projecting flow estimates far into the future is risky, so it is necessary to be explicit about flow capacity at the time of design.

Most of the pollutants that are common to many of these wastewater sources can be effectively treated by wetland systems. The normal concentration range of these pollutants is an important consideration in evaluating wetland treatment system options. This section compares and contrasts these wastewater sources to facilitate initial alternative evaluation.

### Water Quantity

The information on water quantities and timing is assembled into the annual and monthly water budgets for the design, including any seasonal or event storage that may be necessary. Such water budgets are easily prepared within the framework of a spreadsheet program on a personal computer. This information is later linked to the computation of the expected reductions in pollutant concentrations. Interestingly, the addition of a wetland to any of the several forms of pretreatment provides dampening of flow pulses. Although it is necessary to account for the diurnal cycles in the inflows for hydraulic purposes, the wetland will typically "hold" several such daily pulses, because of the extended detention time used in the wetland.

### Small Domestic Systems

Most design information in engineering textbooks is based on large-scale sewer networks that have a continuous base flow. Small-scale wastewater treatment systems often do not have a continuous base flow. On the contrary, low flows are zero (no flow), and peak flows are many times larger than the average flow. These differences in water use patterns raise issues that are not encountered in the design of larger sewage treatment works.

### Small Flows

For design of single-family home treatment systems, the accepted practice in the United States is to base the design flow on the number of bedrooms within the home. These prescriptive flow determinations (typically ranging from 455 to 568 L/d per bedroom) are used to provide a sufficient factor of safety for soil infiltration of septic tank effluent. They do not represent actual water use. Prescriptive flow determinations are commonly interpreted as representing the maximum expected occupancy of the home (two occupants per bedroom) and a corresponding peak flow rate. As a result, peaking factor determinations and infiltration/inflow allowances are typically not necessary when using prescriptive flows based on a bedroom count. Special provisions may apply in some circumstances (Minnesota Pollution Control Agency, 1999).

Flow projections may be based on population for small communities. A prescriptive criterion of 379 L/d per person is commonly used in North America (Great Lakes UMRB, 1997). This per-person flow guideline is intended to represent an average dry weather flow from domestic wastewater sources plus a "normal" amount of infiltration for gravity sewers built with modern construction techniques. If the only available information is the number of homes, an average number of people per household may be used to approximate the total population. The average household size in the United States is 2.7 people (American Housing Survey, 2003), although this varies by geographic location. An appropriate peaking factor must be applied to determine peak flows.

### Patterns of Small Flows

Wastewater flow from individual residences is delivered to a small-scale treatment system via a series of discrete pulses triggered by flush toilets, washing machines, dishwashers, etc. Low flows in small systems will be zero (no flow). Most water use occurs in the morning, evening, and at mealtimes. In the United States, water use from single-family homes has been idealized for design purposes, as indicated in Figure 16.5. As more and more homes are added to the system, flow pulses overlap. If there are enough homes in the collection network, flow pulses overlap to form a continuous base flow, and flow peaks start to attenuate.

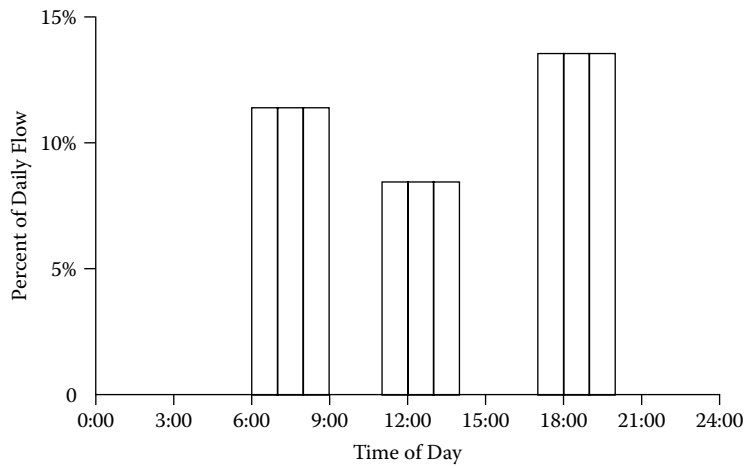



**FIGURE 16.5** Idealized water use pattern for an individual home. (Adapted from NSF International (2000) *Residential wastewater treatment systems*. NSF/ANSI 4–2000, NSF International: Ann Arbor, Michigan. Reprinted with permission.)

For single-family homes, the ratio of the peak flow to the average flow (peaking factor) can be five or higher. Larger treatment systems will experience lower peaking factors due to overlapping flow pulses and the presence of a continuous base flow. In the United States, Recommended Standards for Wastewater Facilities (Great Lakes UMRB, 1997) suggest a formula based on population to determine the ratio of the peak hourly flow to the average daily flow (Equation 16.1). For small populations (less than 100 people), this relationship results in a peaking factor of approximately 4.5.

$$\frac{Q_{\text{peak hourly}}}{Q_{\text{average day}}} = \frac{18 + \sqrt{P}}{4 + \sqrt{P}} \tag{16.1}$$

where

$Q$ = Flow rate, any units
$P$ = Population in thousands

The pattern of peak flow events can be altered dramatically if wastewater is collected and pumped into the treatment system, as might be the case for septic tank pretreatment. Sources of inflow and infiltration from homes (sump pumps, footing drains, roof leaders, and furnace drains) can easily produce much higher flows unless they are identified during the design process with a home plumbing survey program and subsequently separated from the wastewater collection system.

### Actual Water Use

Water use studies in the United States estimate an average daily water use of 189 to 265 L/d per person for homes built before 1994; implementation of standards for water-efficient appliances since then has reduced water use in newer homes to approximately 161 to 227 L/d per person (U.S. EPA, 2002c). Water use is strongly influenced by cultural practices and varies widely from country to country. Across Europe, typical flow rates in small communities (less than 500 people) range from 80 to 120 L/d per person (IWA Specialist Group on Use of Macrophytes in Water Pollution Control, 2000). In Germany, water use rates are much lower than in the United States (Gesellschaft zur Förderung der Abwassertechnik d.V (GFA), 1998), at 100–150 L/d per person. In urban areas of developing countries, water use is approximately 60 L/d per person (Nhapi *et al*., 2003).

## Lagoons

There are many variants on the concept of aquatic units for wastewater treatment, ranging from single-pond units (Water Environment Federation, 2001) to complex arrays of multiple units (Craggs, 2005). Often, other treatment process units are added to complement the pond itself (Middlebrooks *et al*., 2005). The combination of a pond followed by a wetland has been explored at a number of locations (Horne, 1995; Steinmann *et al*., 2003; Tanner and Sukias, 2003; Kadlec, 2003d; Polprasert *et al*., 2005; Wang *et al*., 2005; Kadlec, 2005e). Because the wetland is often an add-on, the flow of the water exiting the pond is often known from performance data.

The prescription for lagoon operation may be continuous discharge, typical of warm climates, or episodic discharge, typical of cold climates. Lagoon systems often discharge to surface waters, for which the goal is to minimize water quality impacts. Maximum dilution occurs at high flow of the recipient, which in turn occurs during freshets, i.e., the spring thaw and the autumn wet season. Therefore, lagoon discharges are traditionally scheduled for those times of maximum dilution. When a wetland is added to the system, there are more options for scheduling the discharge. For instance, the system may be designed for discharges that avoid ammonia toxicity in the recipient (Kadlec and Pries, 2004). Winter storage may be contemplated, provided capacity is present or designed. Therefore, the designer has an added degree of freedom: the total annual volume may be managed to optimize treatment, perhaps at the expense of more pond volume. This design feature is discussed in more detail subsequently.

## Mechanical Plants

Pretreatment systems, such as activated sludge plants, are small-retention devices, which do not typically have much capacity to dampen the incoming flow pulses. Hour-to-hour, day-to-day, and month-to-month flow variations are likely to be passed through the pretreatment system, and thus affect what is entering a follow-on treatment wetland. These pulses will then be partially evened out by an add-on wetland. Flows, whether municipal or industrial, are often seasonal in character. It is necessary to anticipate those patterns, because the wetland must function appropriately under these variable hydraulic conditions. Monthly flow estimates will be required for most point-source projects.

### Infiltration and Inflow

Infiltration is defined as groundwater that seeps into a wastewater collection system. It invariably introduces additional flow into the collection network. Infiltration is strongly influenced by groundwater elevation, workmanship of sewer construction, quality of construction materials, and fraction of the overall collection network that relies on gravity flow. Typical sources of infiltration include poorly installed service laterals, leaking joints on sewer pipes, cracked sewer pipes, and leaking manholes. Exfiltration (movement of water out of the collection system) can also occur. Portions of collection systems that are pumped (such as pressure sewers) have positive internal pressures and are often pressure-tested during construction. As a result, pressure sewer collection systems have a much lower potential for infiltration.

Inflow is defined as extraneous water that is directly discharged to the wastewater collection system. In combined sewer systems, stormwater is a major source of inflow. It is driven by rainfall intensity and amount of impervious surface present within the catchment area. In newer collection networks, stormwater is almost always excluded. In these

situations, major sources of inflow are generally limited to roof leaders, sump pumps, and foundation drains. Because most inflow sources are driven by rainfall, these tend to be high-flow, short-duration events. These "surge" events can have major impacts on treatment systems. The combined effect of infiltration and inflow depend on a number of factors, including the integrity of the sewer system, size of the collection pipes, the presence of high groundwater, and other factors. Typical allowances for combined infiltration/inflow range from 0.09 to 0.9 $m^3$/d/cm/km (Metcalf and Eddy, 1998).

## Water Quality

The concentrations of the pollutants in the water to be treated are critical to the sizing process, and to the prediction of the wetland performance in the face of unknown future variations. A clear definition of the incoming water quality, including the anticipated temporal distribution of concentrations, is essential. There are often seasonal fluctuations for point sources, as well as diurnal fluctuations. Incoming patterns of chemical composition propagate through the wetland and undergo modification, resulting in a spectrum of output compositions. Some of this output variability may be predicted by the design models, namely, those variations that represent responses to moderately slow input changes (those which occur on monthly or less). Faster events involve ecosystem processes that are not included in the design models available at the present time, and therefore will give the appearance of generating stochastic variations.

In domestic and municipal wastewater collection systems, the following components contribute to sewage flow:

- Human excreta (feces and urine)
- Wastewater generated by personal use, including washing, laundry, food preparation, etc. (graywater), and water used as the carrier media for human bodily wastes (blackwater)
- Water that inadvertently leaks into the collection system (infiltration and inflow)
- Wastewater from commercial or industrial sources

Component concentrations of sewage will depend on the individual circumstances of the community. In communities with hand-carried water supplies or limited water resources, graywater is often looked at as an irrigation resource and is not commingled with excreta. In industrial societies, mixing graywater and blackwater to produce "combined sewage" is the norm (Günther, 2000). As a result of these differences in water usage, and hence composition, wetland designers need to consider how the community is utilizing water.

### Small Domestic Systems

There is often no composition data to be used for the design of treatment systems for small systems. It is necessary to resort to estimating methods that consider water use and population in the source community. Untreated human urine and fecal material can introduce a variety of pollutants into the environment. Typical per-person generation rates are summarized in Table 16.1 (Del Porto and Steinfeld, 2000). Graywater includes spent water from bathtubs, showers, washbasins, washing machines, laundry tubs, kitchen sinks, and dishwashers. In developed countries, graywater accounts for 50 to 82% of household water use and represents about half of the organic waste solids produced in the home. When conventional flush toilets are used in a waterborne sewer system, graywater is often combined with blackwater. Relative contributions of pollutants by source (for a combined sewer system) are summarized in Table 16.2 (U.S. EPA, 2002c). Typical constituent concentrations for residential septic tank systems are given Table 16.3.

### Lagoons

Another source of treatment wetland influents arises from pond treatment as the initial component of the treatment train. One or more facultative, anaerobic or aerated ponds or lagoons may be used (Shilton, 2005). Because the wetland is often an add-on, the quality of the water exiting the pond is often known from performance data. If the entire system is constructed at the same time, the lagoon elements should be designed according to the currently accepted

**TABLE 16.1**
**Typical Per-Person Waste Generation Rates**

| Parameter | Urine | Feces | Combined |
|---|---|---|---|
| Volume (L per capita·day) | 1.2 | 0.15 | 1.4 |
| Dry solids (g per capita·day) | 60 | 45 | 105 |
| Moisture content (g per capita·day) | 95% | 70% | — |
| Organic carbon (g per capita·day) | 8.5 | 22 | 30 |
| $BOD_5$ (g per capita·day) | 7.5 | 11 | 18.5 |
| Nitrogen (g per capita·day) | 11 | 2 | 13 |
| Phosphorus (g per capita·day) | 1 | 0.6 | 1.6 |

*Source:* From Del Porto and Steinfeld (2000) *The Composting Toilet System Book.* Center for Ecological Pollution Prevention, Concord, Massachusetts. Reprinted with permission.

**TABLE 16.2**
**Typical Per-Person Combined Sewage Generation Rates**

| Parameter (Mean Values) | $BOD_5$ (g per capita·day) | Suspended Solids (g per capita·day) | Nitrogen (g per capita·day) | Phosphorus (g per capita·day) |
|---|---|---|---|---|
| Garbage disposal | 18.0 | 26.5 | 0.6 | 0.1 |
| Toilet | 16.7 | 27.0 | 8.7 | 1.6 |
| Bathtubs, sinks, appliances | 28.5 | 17.2 | 1.9 | 1.0 |
| Approximate total | 63.2 | 70.7 | 11.2 | 2.7 |

*Source:* Adapted from U.S. EPA (2002c) *Onsite wastewater treatment systems manual*. EPA 625/R-00/008. U.S. EPA Office of Research and Development: Washington, D.C.

methods (Shilton, 2005). Lagoon systems are typically designed for reduction of BOD and total suspended solids (TSS), and occasionally ammonia. Data and older models exist for pond pathogen reduction, but have not been recently updated and synthesized (Davies-Colley, 2005). Phosphorus data for lagoon systems are not voluminous, because it is not frequently regulated in lagoon discharges. Some approximations of the effluent characteristics of several types of lagoons are shown in Table 16.4.

## Mechanical Plants

Table 16.5 summarizes the typical quality of medium-strength, raw, municipal wastewater in the United States and provides a range of values for commonly observed constituents. Municipal wastewater is composed of a variable array of components characterized by the presence of biodegradable organic matter (paper, feces, and food), particulate and dissolved solids, and nutrients. Many municipal wastewaters also receive some component of industrial waste. These flows and residential sources may add trace metals and pesticides to typical municipal wastewater.

Table 16.5 also provides a range of estimated treatment efficiencies for conventional primary and secondary treatment processes, and summarizes the typical quality of secondarily treated municipal wastewaters. These removal efficiencies vary widely depending on the types of treatment processes. However, it is generally observed that at least 70% of the BOD and TSS are removed from municipal wastewater during primary and secondary treatment. Treatment requirements have generally increased over the past decades, and many treatment plants now include at least partial nitrification, perhaps denitrification, and phosphorus removal. These blur the terminology, because they range from "advanced secondary" to "tertiary" and beyond. The follow-on treatment wetland may therefore be termed "tertiary" or, as might be supposed, "quaternary."

This summary can be used as a rough estimate of the influent water quality to be applied to a wetland system designed for primary, secondary, or advanced wastewater treatment.

**TABLE 16.3**
**Typical Wastewater Component Concentrations Entering and Leaving a Residential Septic Tank**

| | Raw Waste | | Septic Tank Effluent | | |
|---|---|---|---|---|---|
| Parameter | Central Estimate | Range | Without Kitchen Solids (Central Estimate) | With Kitchen Solids (Central Estimate) | Range |
| $BOD_5$ (mg/L) | 450 | 210–530 | 180 | 190 | 140–200 |
| TSS (mg/L) | 500 | 230–600 | 80 | 85 | 50–90 |
| $NH_4$-N (mg/L) | 40 | 7–40 | 40 | 45 | 40–60 |
| TKN (mg/L) | 70 | 50–90 | 70 | 75 | 50–90 |
| $NO_x$-N (mg/L) | <1 | 0–1 | <1 | <1 | 0–1 |
| Total phosphorus (mg/L) | 17 | 10–30 | 16 | 16 | 12–20 |
| Fecal coliforms (CFU/mL) | — | $10^6$–$10^{10}$ | — | — | $10^3$–$10^6$ |
| Viruses (PFU/mL) | — | — | — | — | $10^5$–$10^7$ |

*Source:* Data from Metcalf and Eddy Inc. (1991) *Wastewater Engineering, Treatment, Disposal, and Reuse.* Tchobanoglous and Burton (Eds.), Third Edition, McGraw-Hill, New York; Crites and Tchobanoglous (1998) *Small and Decentralized Wastewater Management Systems.* McGraw-Hill, New York.

## TABLE 16.4
## Typical Composition of Lagoon Discharge Water and Percent Removals at Various Levels of Treatment

| Parameter | | Primary Anaerobic | Secondary Aerobic | Facultative | Aerated Facultative | Aerated Partial Mix |
|---|---|---|---|---|---|---|
| BOD | *Reduction*, % | 50–85 | — | 75–95 | — | Up to 95 |
| | Median effluent, mg/L | — | — | — | — | 25 |
| | Effluent range, mg/L | — | 20–40 | 15–35 | 1–45 | 15–40 |
| TSS | *Reduction*, % | — | — | — | — | — |
| | Median effluent, mg/L | — | — | — | — | 40 |
| | Effluent range, mg/L | 80–160 | 80–140 | 10–90 | 1–90 | 20–60 |
| $NH_4$- N | *Reduction*, % | — | — | 23–97 | — | — |
| | Median effluent, mg/L | — | — | 20 | — | 4 |
| | Effluent range, mg/L | — | — | 0.2–25 | 0–12 | — |
| TP | *Reduction*, % | — | — | 50 | — | 15–25 |
| | Median effluent, mg/L | — | — | — | — | 5 |
| | Effluent range, mg/L | — | — | — | 3–4 | — |
| FC | *Reduction* ($log_{10}$) | — | — | 1–5 | 1–5 | — |
| | Median effluent, CFU/100 mL | — | — | 200 | 100 | 100 |

*Source:* Shilton (2005) In *Pond Treatment Technology.* Shilton (Ed.), IWA Publishing, London; Metcalf and Eddy Inc. (1991) *Wastewater Engineering, Treatment, Disposal, and Reuse.* Tchobanoglous and Burton (Eds.), Third Edition, McGraw-Hill, New York; Crites and Tchobanoglous (1998) *Small and Decentralized Wastewater Management Systems.* McGraw-Hill, New York; Crites *et al.* (2006) *Natural Wastewater Treatment Systems.* Meyer (Ed.), CRC Press, Boca Raton, Florida; U.S. EPA (1983a) *Design manual: Municipal wastewater stabilization ponds*. EPA 625/1-83/015, U.S. EPA Office of Water: Cincinnati, Ohio; U.S. EPA (1983c) *Wastewater stabilization ponds: Nitrogen removal*. U.S. EPA Office of Water: Washington, D.C.; Rich (1999) *High Performance Aerated Lagoons.* American Academy of Environmental Engineers, Annapolis, Maryland.

## TABLE 16.5
## Typical Composition of Municipal Wastewater and Percent Removals at Various Levels of Treatment

| | Raw Wastewater (mg/L) | | Percent Removal | | Secondary Effluent (mg/L) | |
|---|---|---|---|---|---|---|
| Constituent | Typical | Range | Primary | Secondary | Typical | Range |
| $BOD_5$ | 220 | 110–400 | 0–45 | 65–95 | 20 | 10–45 |
| COD | 500 | 250–1,000 | 0–40 | 60–85 | 75 | 35–75 |
| TSS | 220 | 100–350 | 0–65 | 60–90 | 30 | 15–60 |
| VSS | 165 | 80–275 | — | — | — | — |
| $NH_4$-N | 25 | 12–50 | 0–20 | 8–15 | 10 | <1–20 |
| $NO_x$-N | 0 | 0 | — | — | 6 | <1–20 |
| Org-N | 15 | 8–35 | 0–20 | 15–50 | 4 | 2–6 |
| TKN | 40 | 20–85 | 0–20 | 20–60 | 14 | 10–20 |
| Total N | 40 | 20–85 | 5–10 | 10–20 | 20 | 10–30 |
| Inorg P | 5 | 4–15 | — | — | 4 | 2–8 |
| Org P | 3 | 2–5 | — | — | 2 | 0–4 |
| Total P | 8 | 6–20 | 0–30 | 10–20 | 6 | 4–8 |
| Arsenic | 0.007 | 0.002–0.02 | 34 | 28 | 0.002 | — |
| Cadmium | 0.008 | <0.005–0.02 | 38 | 33–54 | 0.01 | <0.005–6.4 |
| Chromium | 0.2 | <0.05–3.6 | 44 | 58–74 | 0.09 | <0.05–6.8 |
| Copper | 0.1 | <0.02–0.4 | 49 | 28–76 | 0.05 | <0.02–5.9 |
| Iron | 0.9 | 0.10–1.9 | 43 | 47–72 | 0.36 | 0.10–4.3 |
| Lead | 0.1 | <0.02–0.2 | 52 | 44–69 | 0.05 | <0.02–6.0 |
| Manganese | 0.14 | 0–0.3 | 20 | 13–33 | 0.05 | — |
| Mercury | 0.001 | <0.0001–0.0045 | 11 | 13–83 | 0.001 | <0.0001–0.125 |
| Nickel | 0.2 | — | — | 33 | 0.02 | <0.02–5.4 |
| Silver | 0.022 | 0.004–0.044 | 55 | 79 | 0.002 | — |
| Zinc | 1.0 | — | 36 | 47–50 | 0.15 | <0.02–20 |

*Source:* WPCF (1983) *Nutrient Control. Manual of Practice FD-7*, Water Pollution Control Federation: Washington, D.C.; Metcalf and Eddy Inc. (1991) *Wastewater Engineering, Treatment, Disposal, and Reuse.* Tchobanoglous and Burton (Eds.), Third Edition, McGraw-Hill, New York; Richardson and Nichols (1985) In *Ecological Considerations in Wetlands Treatment of Municipal Wastewaters*. Godfrey (Ed.), Van Nostrand Reinhold Company, New York, pp. 351–391; Krishnan and Smith (1987) In *Aquatic Plants for Water Treatment and Resource Recovery*. Reddy and Smith (Eds.), Magnolia Publishing, Orlando, Florida, pp. 855–878; Williams (1982) In *Water Reuse*. Middlebrooks (Ed.), Ann Arbor Science, Ann Arbor, Michigan, pp. 87–136.

## 16.3 CHARACTERIZATION OF OTHER WASTEWATERS

### Industrial Wastewaters

Although industrial wastewater quality varies among industries, it has a fairly consistent intrasystem effluent quality. Table 16.6 summarizes the typical quality of raw wastewater from a number of industries that have used wetlands treatment technology. Raw industrial wastewater usually receives some level of pretreatment before discharge to a wetlands treatment system. If total concentrations of BOD, suspended solids, and ammonia nitrogen in untreated industrial wastewater are in the concentration range of hundreds to thousands milligrams per liter, it is generally not acceptable for wetlands discharge without additional pretreatment.

### Landfill Leachates

Treatment and disposal of liquid leachates is one of the most difficult problems associated with the use of sanitary landfills for disposal of solid waste. Leachates are produced when rainfall and percolated groundwater combine with inorganic and organic degraded waste. In unlined landfills, leachates

**TABLE 16.6**
**Typical Pollutant Concentrations in a Variety of Untreated Industrial Wastewaters**

| Constituent | Units | Pulp and Paper[a] | Landfill Leachate[b] | Coal Mine Drainage[d] | Petroleum Refinery[e] | Electroplating[f] | Breweries[g] |
|---|---|---|---|---|---|---|---|
| $BOD_5$ | mg/L | 100–500 | 42–10,900 | — | 10–800 | — | 1,500–3,000 |
| COD | mg/L | 600–1,000 | 40–90,000 | — | 50–600 | — | 800–1,400 |
| TSS | mg/L | 500–1,200 | 100–700 | — | 10–300 | 4–600 | 100–500 |
| VSS | mg/L | 100–250 | 60–280 | — | — | 30–100 | 50–500 |
| TDS | mg/L | — | — | — | 1,500–3,000 | 800–5,800 | 50–350 |
| $NH_4$-N | mg/L | — | 0.01–1,000 | — | 0.05–300 | — | — |
| TN | mg/L | — | 70–1,900 | — | — | 10–120 | 25–45 |
| TP | mg/L | — | <0.01–2.7 | — | 1–10 | 20–50 | – |
| pH | S.U. | 6–8 | 3–7.9 | 3–5.5 | 8.5–9.5 | 4–10.5 | 5–7 |
| Sulfate | mg/L | — | 10–260 | 20–2,000 | ND–400 | 30–120 | — |
| Conductance | µS | — | 1,200–16,000 | — | — | — | — |
| TOC | mg/L | — | 11–8,700 | — | 10–500 | — | — |
| Aluminum | mg/L | — | 0.5 | 50 | — | — | — |
| Arsenic | mg/L | — | 0.011–10,000 | — | — | — | — |
| Barium | mg/L | — | 0.1–2,000 | — | — | — | — |
| Cadmium | µg/L | — | 5–8,200 | — | — | 10,000–50,000 | — |
| Chromium | mg/L | — | 0.001–208 | — | ND–3 | 10–120 | — |
| Iron | mg/L | — | 0.09–678 | 50–300 | — | 2–20 | — |
| Lead | µg/L | — | 1–19,000 | — | — | — | — |
| Manganese | mg/L | — | 0.01–550 | 20–300 | — | — | — |
| Selenium | µg/L | — | 3–590 | — | — | — | — |
| Oil and Grease | mg/L | — | — | — | 10–700 | — | — |
| Phenols | mg/L | — | <0.003–17 | — | 0.5–100 | — | — |
| Cyanide | mg/L | — | — | — | — | 1–50 | — |

*Note:* ND = not detected.

[a] From Jorgensen (1979) *Studies in Environmental Science* 5. Elsevier, New York.

[b] From Staubitz *et al*. (1989) *Constructed Wetlands for Wastewater Treatment: Municipal, Industrial, and Agricultural.* Hammer (Ed.), Lewis Publishers, Chelsea, Michigan, pp. 735–742; Lema *et al*. (1988) *Water, Air, and Soil Pollution* 40: 223–250; Bolton and Evans (1991) *Water, Air, and Soil Pollution* 60: 43–53.

[c] From Wildeman and Laudon (1989) In *Constructed Wetlands for Wastewater Treatment: Municipal, Industrial, and Agricultural*. Hammer (Ed.), Lewis Publishers, Chelsea, Michigan.

[d] From Girts and Kleinmann (1986) *National Symposium on Mining, Hydrology, Sedimentology, and Reclamation.* University of Kentucky Press, Louisville, Kentucky, pp. 165–171.

[e] From Adams *et al*. (1981) *Development of Design and Operational Criteria for Wastewater Treatment*. Enviro Press, Nashville, Tennessee; ANL (1990) *Environmental consequences of, and control processes for, energy technologies*. Argonne National Laboratory (ANL) and Noyes Data Corporation: Park Ridge, New Jersey.

[f] From OECD (1983) *Emission Control Costs in the Metal Plating Industry*. Organization for Economic Cooperation and Development (OECD).

[g] From Cooper (1978) *The textile industry. Environmental control and energy conservation*. Noyes Data Corporation. Park Ridge, New Jersey; Wildeman *et al*. (1993a) *Wetland Design for Mining Operations*. Bitech Publishers, Vancouver, British Columbia.

frequently discharge to groundwater or appear as surficial drainage around the base of the landfill. In modern lined landfills, leachates are collected from the lined cells and routed to treatment units. The use of constructed wetlands to treat these landfill leachates is a well-developed technology, currently undergoing rapid expansion in application. This application is discussed further in Chapter 25.

The highly variable nature of solid waste, differences in age and decomposition, and the diversity of chemical and biological reactions that take place in landfills result in a wide range of chemical quality of leachates (McBean and Rovers, 1999). Reviews of "average" landfill concentrations of COD, volatile acids, and nitrogenous compounds show increases during the first few years of operation and then decline over ten or more years. Table 16.6 provides typical ranges encountered in landfill leachates. Flows are generally low, but vary depending on management and minimization of percolation from rainfall. Clearly, the expected volume and chemical quality of a landfill leachate is highly site-specific, may change over time, and must be estimated on a case-by-case basis for wetland treatment system design. A detailed discussion of the leachate quality that may reach a treatment wetland is found in McBean and Rovers (1999).

### Pulp and Paper Wastewater

The pulp and paper industry converts wood products including pines, spruce, poplar, beech, birch, and aspen, as well as recycled paper, into liquefied cellulose pulp and paper. Raw wood and wood chips are converted to pulp (cellulose fibers) by mechanical grinding (ground wood) or through chemical degradation and leaching (sulfite and Kraft processes). At an increasing number of pulp and paper mills, this pulp is bleached to delignify and decolorize the cellulose fibers before paper manufacture. About 29–34 $m^3$ of raw wastewater is produced for each metric ton of pulp and paper produced (Britt, 1970). Total wastewater flow for the U.S. pulp and paper industry is about $20 \times 10^6$ $m^3/d$ (Greyson, 1990). Table 16.6 summarizes the typical composition of this wastewater, although different manufacturing processes result in different wastewater qualities. This flow is equivalent to a raw organic matter ($BOD_5$) load of about $15 \times 10^6$ metric tons per day (Greyson, 1990).

Raw wastewater from pulp and paper mills typically receives primary treatment through settling, either in ponds or in primary clarifiers. When required to meet discharge limitations, secondary treatment at most pulp and paper mills includes biological conversion of $BOD_5$ and additional solids settling in aerated lagoons or in conventional activated sludge treatment systems. To meet reduced effluent limitations, some pulp and paper mills are being required to provide treatment beyond the secondary level. The goals of additional treatment depend on site conditions, such as the quality of the effluent after secondary treatment and water quality permit limits in the receiving water. One goal may be to further reduce $BOD_5$, TSS, nitrogen, phosphorus, color, chlorinated organics (such as adsorbable organic halides or dioxin), and whole effluent toxicity. Constructed and natural wetland treatment systems have been used at a number of pulp and paper mills to provide this advanced secondary or tertiary treatment (Knight *et al*., 1994; Knight, 2004).

### Mine Drainage

During and following mining operations, runoff and leachate from tailings and from abandoned tunnels and shafts dissolve trace metals, contaminating nearby surface waters. Leachates from coal and metal mines contain residual trace metals, notably iron, manganese, and aluminum from coal mines (Table 16.6). Seeps from abandoned mines typically received no treatment in the past, but there is an increasing emphasis on corrective measures (Younger *et al*., 2002). Constructed wetlands are used as a technology to reduce metal concentrations in mining wastewater (Younger *et al*., 2002; PIRAMID Consortium, 2003a, 2003b). As discussed in Chapter 11, metals are precipitated and sequestered in sediments, and taken up by plants in wetlands. Wetland design for metals removal is sometimes limited by the need to avoid toxic concentrations in tissues that could subsequently accumulate in the food chain. Information necessary to evaluate the ability of wetlands to provide treatment of these waste products is summarized in Chapter 11 and in the references listed previously. Both removal rate specification and rate constants have been used in design, but the available level of detailed design data support is not as large as for municipal wastewater treatment.

During coal mining, iron pyrite and other metal-bearing minerals are exposed to percolating water, which leads to the release of acidic leachates to surface water. These drainages typically have low pH and elevated concentrations of dissolved iron, sulfate, calcium, and magnesium. In addition, the drainages have variable and somewhat elevated concentrations of aluminum, copper, manganese, nickel, and zinc (Table 16.6). Many of the streams and impoundments in the Appalachian coal mining region of the United States are affected by acid mine drainage. Conventional treatment of leachates at these sites includes surface grading and recontouring to reduce or divert flows and chemical buffering and precipitation with mechanical treatment plants to improve water quality. Because these processes have relatively high capital and lifetime costs, there has been considerable interest in developing more cost-effective alternatives. Beginning in the early 1980s, research focused on the potential of aerobic wetlands for precipitation of ferric sulfate to neutralize pH and reduce dissolved ferrous iron concentrations. Constructed wetlands are now used at many sites in the United States and Europe to increase the pH and reduce concentrations of iron and manganese at coal mine sites (Kleinman and Hedin, 1989; Younger *et al*., 2002; PIRAMID Consortium, 2003a, 2003b) of coal mine drainage.

### Petroleum Industry Wastewater

Because of the diverse processes at refineries and associated transportation facilities, and the storage of flammable

liquids, land area requirements are large and include many kilometers of piping and hundreds of tanks and storage areas. Wastewater is generated by manufacturing processes, cooling tower blowdown, water and sludge drainage from tanks, and stormwater drainage and runoff (UNEP, 1987). Typical wastewater pollutants at petroleum refineries include $BOD_5$, COD, oil and grease, TSS, $NH_4$-N, phenolics, $H_2S$, trace organics, and heavy metals. Concentrations of many of these pollutants are reduced through source control and preliminary treatments such as sour water stripping, oxidation and neutralization of spent caustics, and cooling tower blowdown treatment. Table 16.6 lists some examples of pollutant concentrations remaining in refinery wastewater.

Raw wastewater from petroleum refineries typically receives additional treatment including gravity separation of oils and greases, primary clarification, dissolved air flotation, and secondary treatment, including oxidation ponds, aerated lagoons, activated sludge, trickling filters, and activated carbon. The API separator process typically removes 60 to 99% of the oil and grease, and smaller proportions of other pollutants. Primary treatment removes 20 to 70% of the $BOD_5$ and TSS and 10 to 60% of the COD. Secondary treatment will reduce 40 to 99% of the $BOD_5$, 30 to 95% of the COD, 40 to 90% of the TOC, 20 to 85% of the TSS, 60 to 99% of the oil and grease, 60 to 99% of the phenol, 9 to 99% of the $NH_4$-N, and 70 to 100% of the sulfide (ANL, 1990).

As described in Chapter 13, constructed wetlands are providing advanced secondary and tertiary treatment of process water and stormwater at a large number of refineries (Knight *et al*., 1999; API, 1999). Constructed wetlands typically will reduce remaining concentrations of $BOD_5$, COD, TSS, $NH_4$-N, oils and grease, phenols, and metals to advanced treatment levels.

### Animal Industry Wastewaters

Animal industry wastewater contains high $BOD_5$, COD, TSS, and nutrients and is qualitatively similar to municipal wastewater. Mass loadings from animal feed lots and other concentrated agricultural activities require intensive treatment systems to provide environmental protection. Traditional treatment methods such as anaerobic lagoons and spray irrigation are not always adequate to provide high-quality water for off-site discharge. Constructed wetlands are being used in a growing number of cases to receive pretreated dairy and swine wastes (NADB database, 1998; Knight *et al*., 2000). These wetland treatment systems must be designed with reasonable organic loadings to prevent plant mortality, odors, and poor treatment efficiencies. Treatment wetlands are a compatible component of on-farm, total waste management. Their land intensiveness is not a serious limitation in most instances. Farmers typically have the equipment and skills necessary to build their own wetlands and operate them successfully. Table 16.7 summarizes the composition of wastes from animal operations, both entering and leaving treatment wetlands.

### Stormwater Runoff

Concentrations of most parameters in stormwater are time dependent. Stormwater concentrations and loads are cyclic

**TABLE 16.7**
**Average Wetland Influent and Effluent Concentrations of Selected Animal Facilities (mg/L)**

| Wastewater Type: | Dairy | Poultry | Cattle | Swine |
|---|---|---|---|---|
| **Number of Systems:** | **60** | **5** | **9** | **58** |
| **$BOD_5$** | | | | |
| In | 442 | 153 | 137 | 104 |
| Out | 141 | 115 | 24 | 44 |
| % Reduction | 68 | 25 | 83 | 58 |
| **TSS** | | | | |
| In | 1,111 | — | 291 | 128 |
| Out | 592 | — | 55 | 62 |
| % Reduction | 47 | — | 81 | 52 |
| **$NH_4$-N** | | | | |
| In | 105 | 74 | 5.1 | 366 |
| Out | 42 | 59 | 2.2 | 221 |
| % Reduction | 60 | 20 | 57 | 40 |
| **TN** | | | | |
| In | 103 | 89 | — | 407 |
| Out | 51 | 70 | — | 248 |
| % Reduction | 51 | 22 | — | 39 |

*Source:* Data from NADB database (1998) *North American Treatment Wetland Database (NADB), Version 2.0*. Compiled by CH2M Hill. Gainesville, Florida; and Knight *et al*. (2000) *Ecological Engineering* 15(1–2): 41–55.

with periods of dry fall and deposition, then the first flush of runoff after rain, followed by exponential decreases in runoff constituent concentrations as storages rinse from the landscape, and finally dry conditions and deposition until the next storm event. Chapter 14 provides a more complete description of the expected flows and concentrations for such event-driven systems.

Table 16.8 provides typical mean concentrations for constituents. The averages are flow-weighted to provide realistic estimates of the total constituent load that escapes during multiple storm events. Instantaneous concentrations will be considerably higher than these averages. Pollutant concentrations and loads generally range from low levels, from undeveloped and park lands to low-density residential and commercial, to agricultural, to higher-density residential and commercial, and finally to high-density commercial, industrial, and agricultural land uses. Mean concentrations per event for $BOD_5$ vary from 1.45 mg/L for undeveloped lands to 20 mg/L for high-density urban areas. TSS concentrations vary from 11 mg/L for undeveloped areas to 150 mg/L for high-density urban areas.

The mass loading rates provided in Table 16.8 represent normalized pollutant loads that are somewhat independent of local rainfall amounts. Because pollutant loads per area per time are relatively constant between similar land use areas, variable local rainfall washes these loads off the land in a few large events or over many smaller events. Urban pollutant loads increase with the imperviousness of the watershed. Although 20 to 40% of the material on street surfaces is organic, it does not biodegrade easily because

**TABLE 16.8**
**Composition and Mass Loading Rates for Stormwaters**

| | Urban | | Industrial | | Residential/Commercial | | Agricultural | |
|---|---|---|---|---|---|---|---|---|
| Constituent | Concentration (mg/L) | Load (kg/ha·yr) | Concentration (mg/L) | Load (kg/ha·yr) | Concentration (mg/L) | Load (kg/ha·yr) | Concentration (mg/L) | Load (kg/ha·yr) |
| $BOD_5$ | 20 (7–56) | 90 | 9.6 | 34–98 | 3.6–20 | 31.59–135.2 | 3.8 | 11.59 |
| COD | 75 (20–275) | — | — | — | — | — | — | — |
| TSS | 150 (20–2890) | 360 | 93.9 | 672–954.5 | 18–140 | 84.28–797 | 55.3 | 24.14 |
| VSS | 88 (53–122) | — | — | — | — | — | — | — |
| $NH_3N$ | 0.582 | — | — | — | — | — | 0.33–0.48 | — |
| TKN | 1.4 (0.57–4.2) | — | — | — | — | — | 2.16–2.27 | — |
| TN | 2.0 (0.7–20) | 11.2 | 1.79 | 7.8–18.06 | 1.1–2.8 | 9.144–32.18 | 2.32 | 10.61 |
| Ortho-P | 0.12 | — | 0.13 | 1.321 | 0.05–0.40 | 0.568–3.302 | 0.13–0.227 | 0.942 |
| TP | 0.36 (0.02–4.3) | 3.4 | 0.31 | 2.2–3.151 | 0.14–0.51 | 1.412–4.85 | 0.344 | 1.362 |
| Copper | 0.05 (0.01–0.40) | 0.049 | — | 0.077 | — | 0.045 | — | — |
| Lead | 0.18 (0.01–1.20) | 0.174 | 0.202 | 0.269–2.053 | 0.065–0.214 | 0.157–2.431 | — | — |
| Zinc | 0.20 (0.01–2.9) | 0.630 | 0.122 | 0.98–1.240 | 0.046–0.170 | 0.218–1.88 | — | — |
| Chromium | — | 0.28 | — | 0.044 | — | 0.026 | — | — |
| Cadmium | 0.0015 | 0.16 | — | 0.024 | — | 0.013 | — | — |
| Iron | 8.7 | — | — | — | — | — | — | — |
| Mercury | 0.00005 | 0.043 | — | 0.065 | — | 0.038 | — | — |
| Nickel | 0.022 | 0.032 | — | 0.030 | — | 0.029 | — | — |
| Oil and Grease | 2.6 | — | — | — | — | — | — | — |

*Source:* Dames and Moore (1990) *Lakeland Comprehensive Stormwater Management and Lake Pollution Study*, Volume I. Report to the City of Lakeland, Florida (May 1990); U.S. EPA (1983b) *Design principles for wetland treatment systems*. EPA 600/2 83/026, Hammer and Kadlec (Eds.), National Technical Information Service; Marsalek and Schroeter (1989) *Water Pollution Research Journal of Canada* 23: 360–378; Bastian (1986) *Potential Impacts on Receiving Water*. Urbonas and Roesner (Eds.). Proceedings of the ASCE Engineering Foundation Conference: Urban Runoff Quality—Impact and Quality Enhancement Technology, 23–27 June 1986. American Society of Civil Engineers: Henniker, New Hampshire, pp. 157–160; Lager *et al.* (1977) *Urban stormwater management and technology: Update and user's guide*. EPA 600/8–77/014, U.S. EPA Office of Research and Development, Municipal Environmental Research Laboratory: Cincinnati, Ohio; Marsalek (1990) *Water Science and Technology* 22: 23–30; Driscoll (1986) *Lognormality of Point and Non-Point Source Pollutant Concentrations*. Urbonas and Roesner (Eds.), Proceedings of the ASCE Engineering Foundation Conference: Urban Runoff Quality—Impact and Quality Enhancement Technology, 23–27 June 1986. American Society of Civil Engineers: Henniker, New Hampshire, pp. 438–458; Shelley and Gaboury (1986) *Estimation of Pollution from Highway Runoff—Initial Results*. Urbonas and Roesner (Eds.). Proceedings of the ASCE Engineering Foundation Conference: Urban Runoff Quality—Impact and Quality Enhancement Technology, 23–27 June 1986. American Society of Civil Engineers: Henniker, New Hampshire, pp. 459–473; Novotny (1992) *Water Environment Technology* January: 40–43.

it comes from leaf and wood litter, rubber, and road surface material (Novotny, 1992). The high metal content of highway solids comes from vehicle emissions. Novotny (1992) reported that the average total nitrogen load from urban lands is 5 kg/ha·yr (1 to 38.5 kg/ha·yr), and the total phosphorus load averages 1 kg/ha·yr (0.5 to 6.25 kg/ha·yr). Urban and residential runoff is being treated with wetland detention basins (Kehoe, 1993) and constructed wetlands (Carleton *et al*., 2001).

## 16.4 TREATMENT GOALS

Wastewater treatment and disposal are regulated by an ever-increasing number of federal, state (provincial), and local laws, rules, ordinances, and standards. In some cases, the most challenging part of implementing a wetland treatment project is complying with regulations through the permitting process. A detailed knowledge of the pertinent regulations is essential to evaluate the feasibility of a wetland treatment project, and to design it properly. An up-to-date, detailed survey of federal, state, and local ordinances should be conducted to determine those that might be relevant to specific projects.

Treated water may be destined for one of three primary receivers: surface water, groundwater, or irrigation (reuse). There are often stringent specifications of quality that must be met to allow discharges to these recipients, and they are quite different. The intent of specifications for surface water discharges is the preservation or improvement of the designated uses of those waters. No matter what the receiving ecosystem or post-wetland treatment element, proper design requires a clear statement of the required water quality leaving the treatment wetland.

### Receiving Water Standards

In the United States, the Clean Water Act created the National Pollutant Discharge Elimination System (NPDES) permitting program. An NPDES permit is required for nearly all point discharges of water or wastewater into waters of the United States, including municipal and industrial wastewater. NPDES permits specify allowable flows and chemical quality of discharges into waters of the United States based on established water quality standards for those receiving waters. The Clean Water Act guides water quality standards, which are promulgated individually by the states. Water quality standards vary among water bodies within a state and among states, depending on specific receiving water resources. Many wetland treatment systems discharge to surface waters, and therefore must meet the conditions of a discharge permit. The conditions of the permit dictate the required performance of the wetland, and therefore govern its sizing.

Traditionally, permits have been developed to control both flows and loads of pollutants. There are typically annual averages and monthly and weekly maxima, perhaps adjusted seasonally. The relation between averages and maximum allowable concentrations may have been determined from other technologies, and may be inappropriate for a wetland system. Similar procedures are in place for other countries.

Discharge in some circumstances is directed to downstream wetlands, which are often a combination of surface and subsurface waters. These wetlands are often federally regulated waters, and subject to appropriate regulations. However, the upstream treatment wetland is a treatment system, typically regulated according to a different set of rules. The design goals for the treatment wetland therefore become the water quality and quantity desired for the management of the downstream, jurisdictional wetland. These are likely to be stricter than for discharge to a large river, for example (due to lower dilution factors in the natural wetland environment).

### Groundwater Discharges

Groundwater discharges are regulated via either the codes for single-home on-site (septic) drainfields or the rules for infiltration of treated wastewaters from municipal treatment plants. The septic drainfield codes are not based explicitly on water quality, and typically use a prescriptive approach based on a presumed reduction in pathogen counts. Larger systems typically target specific water quality parameters, such as nitrate, and hydrogeologic modeling is often required to determine the fate and transport of these parameters in the subsurface environment.

Groundwater discharges of treated water are feasible in a number of circumstances. The problems of avoidance of eutrophication of surface receiving waters are replaced by problems of ensuring proper quality for the aquifer to be recharged. If the groundwater beneath the wetland, or a follow-on infiltration bed, is a drinking water source, then attention must be paid to nitrate, pathogens, and metals, as well as to trace organic chemicals. However, many aquifers are not, and will not be, used for potable water supply. Wetlands, therefore, have a role in pretreatment for conventional rapid infiltration basins (RIBs) and in posttreatment for nitrate removal from underdrained RIBs.

The primary concepts of regulation of groundwater discharges concern nitrates, pathogens, and perhaps salts. The limit of 10 mg/L nitrate-nitrogen as a drinking water standard in the United States results in specifications of nitrate, or by implication, total nitrogen in such discharges. The need to regulate pathogens if any drinking water use is present is obvious. Salt content is of concern if it is a perceived threat to potable water supplies. Groundwater discharges may be a preferred alternative because of the phosphorus-binding potential of many soils. Land application has a long tradition as a means of wastewater disposal. But it is often plagued by a surplus of nitrogen, which escapes crop utilization and nitrifies during transport to the groundwater. Wetlands have the potential to strip excess wastewater nitrogen before land application. In this application, wetland design targets the requirements of the subsequent treatment process (i.e., land application).

The design of the treatment wetland interfaces with the design of the infiltration system. Several sources discuss the design of rapid infiltration systems (Crites and Tchobanoglous, 1998; Water Environment Federation, 2001; Crites *et al*., 2006).

### Interfacing to Reuse

In many parts of the globe, water is in short supply. As treatment technologies improve, it has become possible to consider the treated water as a resource, with a variety of potential beneficial uses. The irrigation of agricultural crops is the leading consumer of treated water. Crops include trees, pastures, and fodder crops in North America, but food crops are irrigated with treated water in other parts of the world. The key consideration is the potential for passing of pathogens to consumers of the crop. Treatment wetlands have been used to help condition the water for these applications (Crites and Tchobanoglous, 1998). Landscape irrigation is another reuse candidate, with applications for ornamentals and golf courses. Again, treatment wetlands have been employed as conditioners in this application (Wallace and Kadlec, 2005).

The reuse water quality standards to be met vary from state to state in the United States. U.S. EPA has guidelines for various categories of reuse (U.S. EPA, 2004). State regulations or guidelines are in place in virtually all states, and typically apply to several categories: (a) unrestricted urban use, (b) restricted urban use, (c) nonfood crops, and (d) food crops. Some of these regulations contain extremely low requirements for solids and pathogens. For instance, the state of California requirement is for turbidity less than 2 NTU, and total coliforms less than 2.2 MPN/100 mL, for the highest reuse category (which incidentally includes use for flushing toilets) (California Code of Regulations, 2001). Other states have less stringent requirements; for instance, Colorado allows unrestricted irrigation of water with a BOD less than 20 mg/L, fecal coliform bacteria less than 25 MPN/100 mL, and TSS less than 40 mg/L (Colorado Department of Health Water Quality Control Division, 2005). States that are driven primarily by liability concerns tend to have very stringent reuse limits, whereas states driven by water scarcity often have less strict limits to reduce the economic barriers to water reuse.

In water-scarce areas such as Mediterranean countries, water scarcity may be a driving factor for reuse, and lower levels of treatment may be acceptable (e.g., control of parasites but not bacterial or viral contaminants). In these cases, alternate treatment guidelines will apply (Korkusuz, 2005).

Clearly, there is a lower limit to the treatment achievable in constructed wetlands, because natural processes create background concentrations that may be in excess of local regulatory requirements for water reuse.

### Excursion Containment and Safety Factors

All treatment technologies possess a spectrum of effluent concentrations, which is predictable only in the probabilistic sense. Therefore, in addition to the mean effluent concentration (which may vary in a deterministic way with temperature and loading), there is an associated bandwidth of concentration. Regulations may constrain both the mean and the maximum of the band, via specification of a limit on the maximum daily, weekly, or monthly value, together with a limit on the average annual value. In design, care must be taken to accommodate the most restrictive of multiple averaging tests given by the regulation.

The average system performance will depend on either the season of the year or the water temperature, or both. These are deterministic variations; that is, they may be predicted from the seasonality of the removal rate coefficient or, more directly, from information on observed trends in treatment wetland outlet concentrations. Theta factors and trend properties (given in Part I) allow the designer to forecast seasonal deterministic trends. There remains the variability not predicted by such seasonality (see Chapter 9, Figure 9.48, for example).

Probabilistic effects are important in the utilization of design models for predicting removal performance of treatment wetlands. Regulatory requirements often employ a standard other than a long-term average. There may be a maximum monthly concentration not to be exceeded, or a specified concentration not be exceeded for more than a certain percentage of samples. To illustrate, consider the representation of Figure 16.6, showing the hypothetical reduction curve for a typical wetland. The information in Part I is unequivocal; as wetland size increases, there is a downward trend in pollutant concentration.

However, there is also a scatter in the individual measurements that make up the trend. During any specified part of the year, the concentration of a pollutant follows a decreasing curve, with wetland size (detention time), and has an associated bandwidth of scatter in expected values (Figure 16.6). In this hypothetical example, that scatter is shown as a uniform distribution about the trend line, with a bandwidth proportional to the trend value. It is supposed that the regulatory limit is a concentration of 30 mg/L, as a maximum allowable. The trend model tells us what size wetland is needed to meet 30 mg/L as a long-term average, which is a detention time of 6.1 days. However, the scatter is such that half the time the measured values will be higher, up to 42 mg/L. The exceedance frequency is expected to be 50% because the design is based on the mean performance. As this level of excursions is likely to be quite unacceptable from a regulatory standpoint, it would be necessary to increase the size of the wetland.

At a detention time of 9.1 days (almost a 50% increase in wetland size), a large majority of excursions are contained (95% this hypothetical example) below the regulatory limit, and the system would experience exceedances only 5% of the time. In this case, the 50% increase in size is needed for excursion containment of an expected, quantified scatter. At the size that contains excursions (9.1 days' detention), the trend value is 21 mg/L, or 70% of the limit value for the design. This fraction is called the *coefficient of reliability*

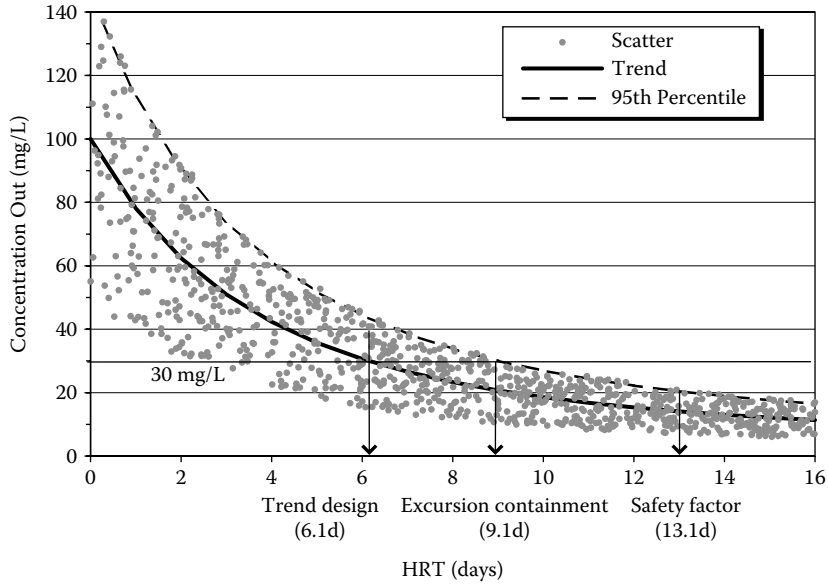



**FIGURE 16.6** A hypothetical example of design for excursion containment. The trend represents $P = 3$, $C^* = 5$ mg/L, and $k = 30$ m/yr. The scatter is a uniform distribution with a ±50% bandwidth. On average, a goal of 30 mg/L can be met with 6.1 days' detention. To avoid exceedances at the 95% level (one time in 20), the detention time should be 9.1 days. More wetland area (more detention time) may be added as a safety factor; i.e., 13.1 days should contain outlet concentrations to less than 20 mg/L most of the time.

(COR). Crites and Tchobanoglous (1998) present a method for its estimation from the coefficient of variation (variance/mean) of one or more datasets, adapted from activated sludge technology. In this book, an exceedance multiplier is used, which is just the reciprocal of the COR:

$$\textit{Exceedance multiplier} = 1 + \Psi = \frac{1}{\text{COR}} \qquad (16.2)$$

where

$\Psi$ = additive fraction of trend of a given exceedance frequency, dimensionless

Values of such multipliers were determined for many pollutants for many wetlands, and the average values are tabulated in Part I for various exceedance frequencies. The relationship between monthly trend average effluent concentration and the 90th percentile monthly concentration for typically regulated constituents is given in Table 16.9, which is extracted from the various results in Part I.

Exit concentrations fluctuate with an amplitude of about 1.75 times the mean for the 90th percentile, meaning that this percentile is about 75% higher than the mean. (Fecal coliform bacteria are a separate case; the multiplier for FC is on $\log_{10}$).

Beyond excursion containment, standard engineering practice is to allow a margin for the unexpected. There are no fixed rules for the selection of a safety factor for wetlands. Water Environment Federation (2001) suggests that an extra 15–25% area is required for treatment wetlands. Equivalently, the designer may elect to design to a more stringent standard than that specified in the regulation. For instance, in the example in Figure 16.6, it could be decided to contain excursions below 20 mg/L instead of 30 mg/L. That leads to a yet larger area requirement, corresponding to 13.1 days' detention in this example.

**TABLE 16.9**
**Trend Multipliers Required to Contain the 90th Percentile of Excursions around Trend Means for Various Pollutants**

| Parameter | FWS | HSSF | VF |
|---|---|---|---|
| BOD | 1.80 | 1.78 | — |
| TSS | 2.21 | 1.84 | — |
| $NH_4$-N | 1.89 | 1.76 | 3.03 |
| $NO_x$-N | 1.74 | — | — |
| TN | 1.53 | — | — |
| TP | 1.94 | 1.66 | — |
| FC ($\log_{10}$) | 1.23 | 1.48 | — |

*Note:* Other percentiles can be found in the pollutant chapters of Part I.

Historically, Kadlec and Knight (1996) determined multipliers corresponding to the 100th percentile of monthly means from the NADB. These were relative to the long-term mean value for a particular wetland, and therefore seasonal variations, whether temperature driven or not, were included in the multiplier. In this book, an annual trend is computed as the basis of the multiplier, thus excluding seasonal phenomena from this measure of random scatter. Further, wetland data may sometimes contain sufficient intrasystem variability to place the 100th percentile above the median inlet concentration. For wetlands with very low inlet concentrations, it may not be possible to design a wetland to totally avoid the possibility of monthly exceedances. Consequently, multipliers for various frequencies of occurrence are tabulated in Part I, and are used in Part II for design purposes.

Curiously, the use of a COR has been ascribed as an attribute of areal first-order models but not an attribute of volumetric first-order models (Water Environment Federation, 2001; Crites *et al*., 2006). Incredibly, the absence of a COR in a volumetric model has been described as an advantage, because then it has "no limiting impact on the mathematical results of design models" (Crites *et al*., 2006). Conversely, Crites *et al*. (2006) portray the use of a COR to adjust the design goal as a disadvantage of the areal model, because it "may result in excessive wetland sizes to achieve low concentrations." Of course, the use of a COR has nothing to do with how one determines the trend values, as is apparent from its use with the volumetric model (Crites and Tchobanoglous, 1998). And there is no doubt that the larger wetland sizes needed to contain excursions are required to achieve an acceptable level of regulatory compliance.

## Other Design Parameters

Some of the specifications of regulatory permits or licenses are outside the commonly encountered groups of rational design parameters, which include BOD, TSS, nitrogen compounds, phosphorus pathogens, metals, organics, and temperature. These do not have loading charts, nor do they have *k*-values.

### pH

There is often a specified range of allowable pH for discharges to surface waters, typically 6.0–9.0. Most treatment wetland applications are not likely to exceed such ranges, as detailed in Chapter 5. Exceptions are the acid mine drainage wetlands, in which the design goal includes raising the pH of the incoming water. Treatment wetlands are not particularly effective at neutralizing strong acid, whereas they are quite good at creating circumneutral pH for more benign influents, such as food wastewaters. Other exceptions include industrial processes, for which a neutralization step is included as part of pretreatment.

### Toxicity

The U.S. Clean Water Act prohibits the discharge of toxic substances to waters of the United States. For this reason, whole effluent toxicity (WET) monitoring is included in the NPDES permits for many municipal treatment plants. WET tests are the standardized procedure to detect levels of acute and chronic toxicity in municipal and industrial effluents. Individual pollutants that contribute to toxicity may be monitored in some instances, such as for nitrite and ammonia, but biomonitoring is required to assess the overall net potential for acute and chronic toxicity to receiving water biota or surrogates.

The short-term chronic toxicity tests that were eventually developed by the U.S. EPA are a relatively inexpensive method of assessing WET (U.S. EPA, 1994). Freshwater chronic toxicity tests utilize three organisms:

- The water flea (*Ceriodaphnia dubia*)
- The fathead minnow (*Pimephales promelus*)
- The green alga (*Selenastrum capricornutum*)

Test methods require a seven-day (96 hours for the algal test) testing period, with renewals of the testing solution using the tested effluent three times during that seven-day period. For the water flea, the test encompasses three reproductive cycles, involving three broods of young (neonates). The adult water fleas are typically fed three times per day during the testing period. Acute toxicity is assessed via mortality. Chronic toxicity is assessed via the number of young produced per female. Tests may be conducted using just control water and 100% effluent, or one or more diluted effluent concentrations.

Fathead minnow testing utilizes larval fish, tested over a seven-day growth period with test water renewals. Acute toxicity is assessed through observed mortality of the fish. Chronic toxicity is assessed by measurement of the final dry weight of the tiny fish at the end of the test period. Laboratory controls are utilized and one or more effluent concentrations are tested to assess the lethal and sublethal effects of the effluent on the fish. Some states have specific protocols that vary from the federal WET guidelines, because of the use of other vertebrate and invertebrate species. Two alternate species are rainbow trout (*Salmo gairdneri*) and a different water flea, *Daphnia magna*. Fathead minnows may be replaced with indigenous fish species such as the bluegill (*Lepomis macrochirus*) or the bannerfin shiner (*Cyprinella leedsi*).

## Toxicity Reduction in FWS Wetlands

A number of studies have shown that constructed wetlands can be effective in reducing toxicity (Knight *et al*., 1997; U.S. EPA, 1999; Wetland Solutions, Inc., 2003). U.S. EPA-sponsored synoptic studies at six constructed treatment wetlands in the United States (McAllister, 1992; McAllister, 1993a; McAllister, 1993b), including standardized toxicity tests. The Collins, Mississippi, wetland had significant acute and chronic toxicity at the wetland inflow, probably due to high unionized ammonia concentrations, but acute and chronic toxicity were absent in the wetland outflow. West Jackson County, Mississippi, had slight acute and chronic toxicity to the water flea at the wetland inflow but no toxicity

at the outflow. Survival and reproduction of the water fleas were increased by the Lakeland, Florida, constructed wetland. No chronic toxicity was present at either the inflow or the outflow of the Orlando Easterly, Florida, wetland.

No toxicity was detected at the inflow of the Incline Village, Nevada, wetland, but the water in the terminal cell of this near-total evaporative wetland was acutely and chronically toxic to the water flea. A similar reduction in reproduction was observed with passage through the Show Low, Nevada, constructed wetland, another near-zero discharge (evaporative) system. It is likely that toxicity was increased in these wetlands (Incline Village and Show Low) as a result of evaporative concentration of salts and other constituents.

Toxicity test results are reported for four treatment wetlands in the NADB v. 2.0 (U.S. EPA, 1999): Santa Rosa, California; Minot, North Dakota; American Crystal Sugar Company in Hillsboro, North Dakota; and Arcata, California (Wetland Solutions, Inc., 2003). An average of 9% mortality in rainbow trout was observed in samples collected at three points in the Santa Rosa wetland system, but there were no influent data. At the Minot constructed wetland outlet, no significant mortality was recorded for water fleas and fathead minnows for test dilutions up to 87.5% effluent. The Hillsboro sugar processing wastewater treatment wetland produced effluents slightly toxic to fathead minnows (15% mortality) and to water fleas (16% mortality). No wetland influent data were available from this site. At the Arcata wetland, data indicated little change in survival between the wetland inlet and outlet (94 to 96%).

Whole water toxicity was measured with water fleas and fathead minnows in quarterly tests of operation in six wetlands treating paper mill wastewaters (Knight *et al*., 1994). No acute toxicity to either organism was found. Influent samples were chronically toxic to the water flea in all cases, but reduced by the wetlands in 10 of 11 comparisons. No chronic toxicity to fathead minnows was observed in any of the wetland outflow samples.

A number of petroleum facilities have reported the reduction of toxicity by treatment wetlands (API, 1999). These include refineries at Richmond, California (Duda, 1992), and St. Charles, Louisiana (Hawkins *et al*., 1995), as well as oil transfer terminals and oil sand processing facilities (Bishay *et al*., 1995).

The Tres Rios constructed wetlands at Phoenix, Arizona, have been extensively studied for toxicity reduction (Wetland Solutions, Inc., 2003). Acute and chronic toxicity in both incoming pretreated water and the wetland outflows was nearly nonexistent or difficult to detect. It was concluded that use of full-scale constructed wetlands for final effluent polishing of the 91st Avenue WWTP final effluent reduced the frequency and magnitude of WET. The wetlands typically reduced low levels of toxicity when they occurred in the WWTP effluent, and did not add to toxicity. However, high levels of toxicity passed through the wetlands with minor alteration, probably due to the very short hydraulic residence times in this system.

In summary, there exist studies indicating that treatment wetlands generally reduce levels of acute and chronic toxicity if present in the inflow. However, the limited amount of wetland data, variation in influent toxicity, and the variability among test protocols preclude forecast modeling of wetland effects on WET. It is not currently possible to calculate the required size of a treatment wetland for a specified level of toxicity reduction.

## 16.5 CLIMATE AND THE WATER BUDGET

Performance-based design rests upon the water budget for the system. The amount of water to be treated is only one part of the hydrologic setting for the project, with precipitation, evapotranspiration, and seepage being the other principal components. These variables are in turn components of the site climate. Another critical component of climate is temperature, which is a driving force for the rates of some wetland processes.

### Climate

Climate cannot be controlled when selecting a specific site for a constructed or natural wetland project. However, climate is important during project planning and alternative selection because it affects the type and size of wetland that will be used. Latitude is the most critical determinant of climate because it determines seasonal temperature ranges. Other climatic factors that are important during project planning include rainfall, insolation, wind direction and velocities, and evapotranspiration. In the United States, climatic information can be obtained from the National Aeronautic and Atmospheric Administration (NOAA) at the following Web address:

National Climatic Data Center
U.S. Department of Commerce
http://www.ncdc.noaa.gov/oa/climate/climateproducts.html

For the entire world, a summary overview of annual patterns of temperature and precipitation may be obtained (free of charge) at:

Climates of the World
National Oceanic and Atmospheric Administration
National Climatic Data Center
Asheville, North Carolina
http://www.ncdc.noaa.gov/oa/documentlibrary/pdf/climatesoftheworld.pdf

Weather and climate data for specific sites across the planet can also be obtained through Web sites such as Weather Underground (http://www.wunderground.com/).

Wetland hydrology is affected not only by the volume of applied wastewater but also by the net balance between rainfall and evapotranspiration. Wetland designers in areas with high net precipitation must take this incremental added water volume into account when deciding on wetland volume, flow control structures, and freeboard of berms. In

areas of high evapotranspiration, wetland design calculations need to consider water loss to the atmosphere. In some cases, the decision may be made to allow the wetland to leak, and seepage can become an important feature of the water budget and have a strong effect on treatment calculations. Chapter 2 presents examples of water balances for FWS wetland treatment systems.

### Rainfall

Figure 2.2 provides a map of the annual average precipitation for the United States. More detailed information concerning the average monthly net precipitation values for the project location should be used to estimate the typical monthly water balance for a continuous flow wetland under extreme seasonal conditions. Historical values of monthly mean precipitation may be found, for 1–10 divisions per state, in the following free Internet publication:

Climatology of the United States No. 85
Division Normals and Standard Deviations of Temperature, Precipitation, and Heating and Cooling Degree Days
National Oceanic and Atmospheric Administration
National Climatic Data Center
Asheville, North Carolina
http://hurricane.ncdc.noaa.gov/cgi-bin/climatenormals/climatenormals.pl

In cold climates, snowfall occurs in the winter months, and accumulates on the wetland during the early winter. The spring melting period causes this stored water to enter the wetland, often as a fairly brief pulse lasting a few days only. Snowfall and snow depth records are available from the National Climatic Data Center (NCDC) for the United States.

As discussed in Chapter 14, rainfall is the primary driving force for stormwater wetlands, and yet more detailed data is therefore desirable. Site-specific information on the intensity and duration of rains, and the interevent spacing, is needed to perform design calculations.

### Evapotranspiration

The mechanisms by which a wetland loses water to the atmosphere have been detailed in Chapter 2. Methods of estimation and data sources have also been detailed in that chapter.

As an example of the kind of information that wetland designers have available, Figure 16.7 shows the various climate zones of the state of California. Table 16.10 lists the monthly *ET* for each zone. Note that arid regions have an annual *ET* loss of 1,819 mm/yr, whereas wet, coastal regions have only 838 mm/yr. The peak *ET* rates are in midsummer, and range up to 9 mm/d for the warm arid zones. For states that do not have irrigation information services, the designer can obtain pan evaporation data for locations in the United States from the NCDC.

The transpiration component of *ET* moves downward into the root zone of a FWS wetland, and thus represents a withdrawal of water and pollutants from the surface water body. In that regard, this transpiration flux resembles infiltration, and can be treated as such in FWS design. In SSF systems, water passes through the rhizosphere, and consequently both evaporation and transpiration losses are withdrawn from the flowing water.

As discussed in Chapter 2, evaporation affects both the flow rate (detention time) and the concentrations of pollutants (evaporative concentration). Therefore, it is inaccurate to merely average the inflows and outflows to account for effects, because that misses the evaporative concentration effect. The flow is slowed, and pollutants become more concentrated. The reverse is true for rainfall: velocities are increased, and pollutants are diluted. The mass balances used in this book for design correctly account for both velocity and dilution effects.

Extreme examples of systems in which *ET* is a dominant factor include lightly loaded wetlands in arid climates. The Incline Village, Nevada, system evaporates nearly all the incoming water, and *ET* was the key factor in project design (Kadlec *et al*., 1990). The Saginaw, Michigan, project operates only in the summer, during which there is an excess of *ET* over rainfall even for this moderately wet climate (Kadlec, 2003c). In another example, on the order of half the water entering the Roblin, Manitoba, FWS wetland system is, by design, lost to net evapotranspiration (PFRA and Town of Roblin, 2003).

### Seepage

Although many treatment wetlands are contained, either with clay or synthetic liners, there are a number of situations in which containment is not needed or not feasible. Any water that leaks vertically downward, or horizontally outward under and through containment berms, represents a water loss that slows flow and contributes to pollutant loss from the wetland water body. Infiltrated water moves into and with the regional groundwater, and may reemerge at nearby locations as groundwater discharge. During passage through the soils under and around the wetland, these seepage waters may receive some additional treatment, by microbial degradation or adsorption, for example. However, not all potential contaminants are subject to soil and aquifer treatment.

The design procedures described herein pertain to the wetland water body and the planned outflows that occur as discharges from structures. This is not to say that infiltration flows can be ignored; instead, the projects with significant seepage are to be viewed as dual-discharge projects. The seepage should meet groundwater requirements, and the structure outflows should meet the appropriate goals, which may be for surface water recipients or infiltration.

Examples of systems in which *ET* is a nontrivial factor include the stormwater treatment areas of South Florida (Nungesser and Chimney, 2006), which lose and/or gain up to about 20% of the flow from groundwater. The New River



**FIGURE 16.7** Reference evapotranspiration for the state of California. See Table 16.10 for the annual pattern for each zone. (Adapted from California Irrigation Management Information System; http://www.cimis.water.ca.gov/cimis/info.jsp.)

wetlands in Southern California infiltrated 47% of the inflow at the Brawley site, and 39% at the Imperial site, over a four-year period of operation (TTI and WMS, 2006).

The wetlands at the Sacramento, California, site (Nolte and Associates, 1998) infiltrated 12.4 ± 4.0% of the inflow for four cells in 1998. The infiltration rates for the site soils were determined prior to the project to be less than 0.15 cm/h for two loams, and 0.15–0.50 cm/h for two clays. For the four wetland cells, in their fifth year of operation, the rates were 0.028 ± 0.008 cm/h, considerably less than those of the parent soils.

Three of the four wetlands at the Tres Rios, Arizona, site had minimal infiltration, but the fourth was built on very coarse river gravel bed materials (Wass, Gerke and Associates, 2001), and the majority of the water leaked vertically downward. Of interest in this project are the trends in the infiltration rates over the first few years of the project history (Figure 16.8). Wetlands H1 and H2 were built on former agricultural soils, and possessed modest infiltration losses of 1.5 and 1.9 cm/d, respectively. Wetland C2 was built on coarse river gravel with a topping of agricultural soil, and the agricultural soils sealed the wetland to about the same leakage rate as H1 and H2: 1.7 cm/d. Wetland C1 was built on coarse river gravel, and leaked at an average rate of 19.7 cm/d. There was a decrease in the infiltration in the cobble wetland C1, from 25.3 to 14.0 cm/d based on the trend line. However, the other three cells maintained a relatively constant infiltration rate. Based on these limited results, only minimal self-sealing should be anticipated in treatment wetlands. If infiltration is not an allowable operating condition, the use of clay or synthetic liners should be considered.

## Temperature

As shown in the design equations provided in Chapter 9, water temperature is important when sizing wetland treatment systems for removal of all forms of nitrogen. Figure 16.9 provides a map of mean annual temperatures in the United States. However, more detail is desirable, and monthly mean temperature is a good estimator of monthly mean water temperature (see Chapter 4), which allows the interpretation of thermal effects on some microbial components of treatment.

**TABLE 16.10**
**Monthly Reference Evapotranspiration by Zone for the State of California**

| Zone | Jan (mm/mo) | Feb (mm/mo) | Mar (mm/mo) | Apr (mm/mo) | May (mm/mo) | Jun (mm/mo) | Jul (mm/mo) | Aug (mm/mo) | Sep (mm/mo) | Oct (mm/mo) | Nov (mm/mo) | Dec (mm/mo) | Total (mm/yr) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 | 24 | 36 | 63 | 84 | 102 | 114 | 118 | 102 | 84 | 63 | 30 | 16 | 836 |
| 2 | 31 | 43 | 79 | 99 | 118 | 130 | 126 | 118 | 99 | 71 | 46 | 31 | 991 |
| 3 | 47 | 57 | 94 | 122 | 134 | 145 | 142 | 134 | 107 | 87 | 61 | 47 | 1,176 |
| 4 | 47 | 57 | 87 | 114 | 134 | 145 | 150 | 142 | 114 | 87 | 61 | 47 | 1,184 |
| 5 | 24 | 43 | 71 | 107 | 142 | 160 | 165 | 150 | 114 | 79 | 38 | 24 | 1,115 |
| 6 | 47 | 57 | 87 | 122 | 142 | 160 | 165 | 157 | 122 | 94 | 61 | 47 | 1,262 |
| 7 | 16 | 36 | 63 | 99 | 134 | 160 | 189 | 165 | 122 | 71 | 30 | 16 | 1,101 |
| 8 | 31 | 43 | 87 | 122 | 157 | 175 | 189 | 165 | 130 | 87 | 46 | 24 | 1,255 |
| 9 | 55 | 71 | 102 | 130 | 150 | 168 | 189 | 173 | 145 | 102 | 69 | 47 | 1,401 |
| 10 | 24 | 43 | 79 | 114 | 150 | 183 | 205 | 181 | 130 | 79 | 38 | 24 | 1,248 |
| 11 | 39 | 57 | 79 | 114 | 150 | 183 | 205 | 189 | 145 | 94 | 53 | 39 | 1,347 |
| 12 | 31 | 50 | 87 | 130 | 173 | 198 | 205 | 181 | 137 | 94 | 46 | 24 | 1,356 |
| 13 | 31 | 50 | 79 | 122 | 165 | 198 | 228 | 197 | 145 | 94 | 46 | 24 | 1,379 |
| 14 | 39 | 57 | 94 | 130 | 173 | 198 | 220 | 197 | 145 | 102 | 53 | 39 | 1,449 |
| 15 | 31 | 57 | 94 | 145 | 189 | 206 | 220 | 197 | 145 | 102 | 53 | 31 | 1,472 |
| 16 | 39 | 64 | 102 | 221 | 197 | 221 | 236 | 213 | 160 | 110 | 61 | 39 | 1,664 |
| 17 | 47 | 71 | 118 | 152 | 205 | 229 | 252 | 220 | 168 | 110 | 69 | 47 | 1,688 |
| 18 | 63 | 85 | 134 | 175 | 220 | 244 | 244 | 220 | 175 | 126 | 76 | 55 | 1,819 |

*Note:* See Figure 16.7 for the locations of the zones.

*Source:* California Irrigation Management Information System; http://wwwcimis.water.ca.gov/cimis/info.jsp.

Historical values of monthly mean air temperatures may be found, for 1–10 divisions per state, in Climatology of the United States No. 85 (previously discussed in the subsection titled "Rainfall" in this chapter).

For planning purposes, the long-term average temperature during the coldest month of the year has been found to be a good estimator of the critical low water temperature that will be experienced in a wetland treatment system. For areas

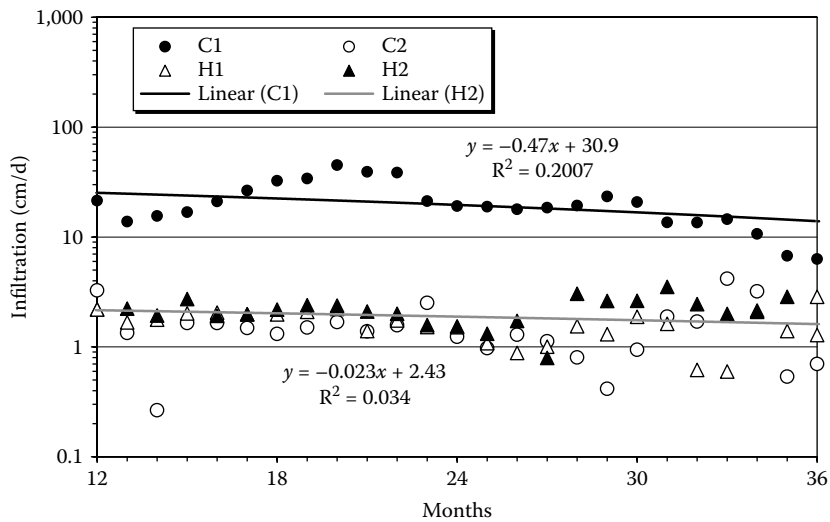



**FIGURE 16.8** Infiltration rates at the Tres Rios FWS wetlands.

| Cell | Soil | Infiltration Rate (cm/d) | Percent of Inflow |
|---|---|---|---|
| C1 | Cobble | 19.7 | 74 |
| C2 | Silty loam over cobble | 1.7 | 12 |
| H1 | Silty loam | 1.5 | 12 |
| H2 | Silty loam | 1.9 | 14 |

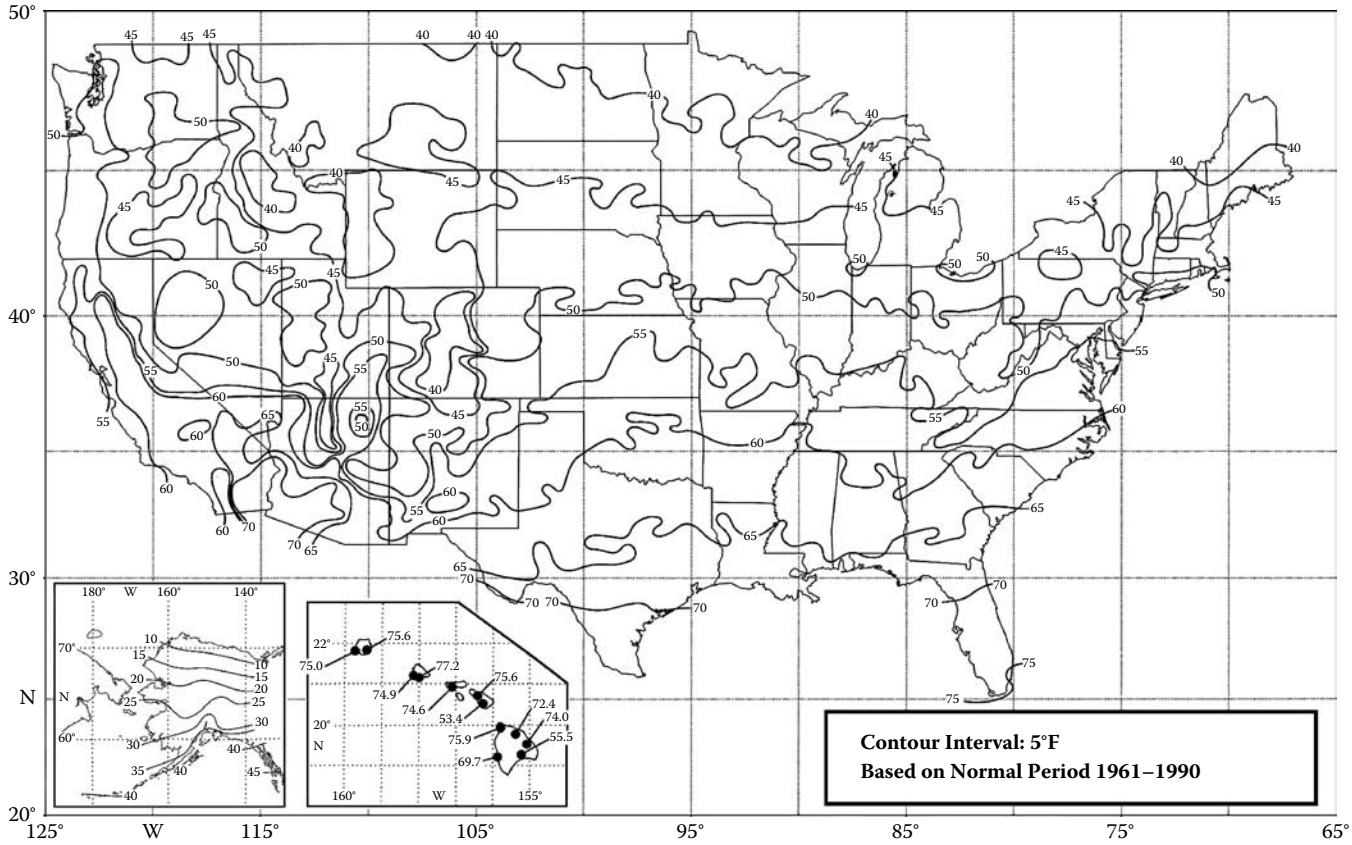



**FIGURE 16.9** Mean annual air temperatures in the United States (°C = (°F – 32)/1.8). (From National Climatic Data Center (2007) at http://www.ncdc.noaa.gov/oa/climate/climateproducts.html.)

in which the average monthly temperature is less than zero, it can be assumed that the minimum wetland operational temperature will be slightly above zero under an ice cover (see Chapter 4 for more information).

## 16.6 SELECTION OF WETLAND TYPE

The type of wetland selected varies with a number of factors, and the best choice may require alternatives evaluation. There are some over-arching concepts that can assist in the selection.

### "Natural" versus "Engineered" Systems

Treatment wetlands can be designed to be "au natural" or complex. Examples of designs along a full spectrum from almost zero fossil-fuel input to highly engineered and managed systems can be found around the world. In this book we term these design and operation strategies as Type *A* and Type *B* wetlands. Type *A* systems are at the most natural, low-energy end of the scale. They have the minimum number of structures, and may often be gravity fed. Vegetation may be allowed to proceed by natural recruitment (Figure 16.10). Type *B* wetlands are higher energy and more highly engineered systems. Pumps are likely to be present, for recycle as well as other water transfers, and piping can be complex. Special media may be involved, and the system may be mechanically aerated in some cases. Vegetation is likely to be highly specific, such as a monoculture of planted *Phragmites* (Figure 16.11). There are good reasons to slide on this scale of design and management complexity, depending on project goals.

Type *A* wetlands are the preferred solution when:

- There is sufficient treatment potential with a Type *A* approach.
- Relatively inexpensive land area is abundant and land is not a limiting factor.
- There is insignificant potential for unauthorized human interaction with the wastewater in the wetland.
- Habitat and aesthetic benefits outweigh the potential for nuisance conditions related to mosquitoes or other wildlife.

Type *A* wetlands are typically constructed FWS marshes, relatively low-tech HSSF wetlands, and natural treatment wetlands. They are always less costly to operate than Type *B* treatment wetlands, and may be less costly to construct, depending on the local situation.

The Type *B* treatment wetland design and management strategy is preferred when one or more of the factors listed previously are limiting the design. Specific examples of Type *B* treatment wetlands include:



**FIGURE 16.10** Type *A* constructed wetlands may be built by berming off a plot of ground and allowing natural regrowth to vegetate the system. This wetland at Onaway, Michigan, is used for ultra-polishing of a lagoon effluent that has had post-sand filtration and phosphorus removal by chemical precipitation.

- HSSF wetlands augmented with saturable media intended for phosphorus or metals removal
- Addition of aerators in constructed HSSF wetlands for improvement of nitrification
- Addition of VF components to HSSF constructed wetlands to improve nitrification (potentially with flow recirculation)
- FWS or HSSF wetlands and aquatic plant treatment systems with plant harvesting
- High solids treatment wetlands with frequently cleaned forebays, or frequent refurbishment and restarting, as for sludge reed beds

Type *B* wetlands are more expensive to operate, and may be more expensive to construct, compared to Type *A* wetlands, and consequently there needs to be a clearly demonstrable need for the added complexity and energy consumption.

## FWS or HSSF?

The two most prevalent types of treatment wetland, especially during the early history of the technology, are FWS and HSSF wetlands. The mind-set in North America was often to emulate the natural habitat wetlands (duck marshes) for ecological reasons. In Europe, there prevailed, and still does, the concept that HSSF flow systems are more efficient, and will require a smaller footprint for an equivalent level of treatment performance.

An analysis of performance data suggests that HSSF wetlands are not more efficient than FWS systems. Earlier




**FIGURE 16.11** This small-scale vertical flow wetland system, near Aarhus, Denmark, consists of a below-ground sedimentation tank, two vertical flow *Phragmites* beds, three phosphorus removal beds, four pumps in pump wells, and complex interconnected piping. It qualifies as a Type *B* constructed wetland system. (Photo courtesy H. Brix.)

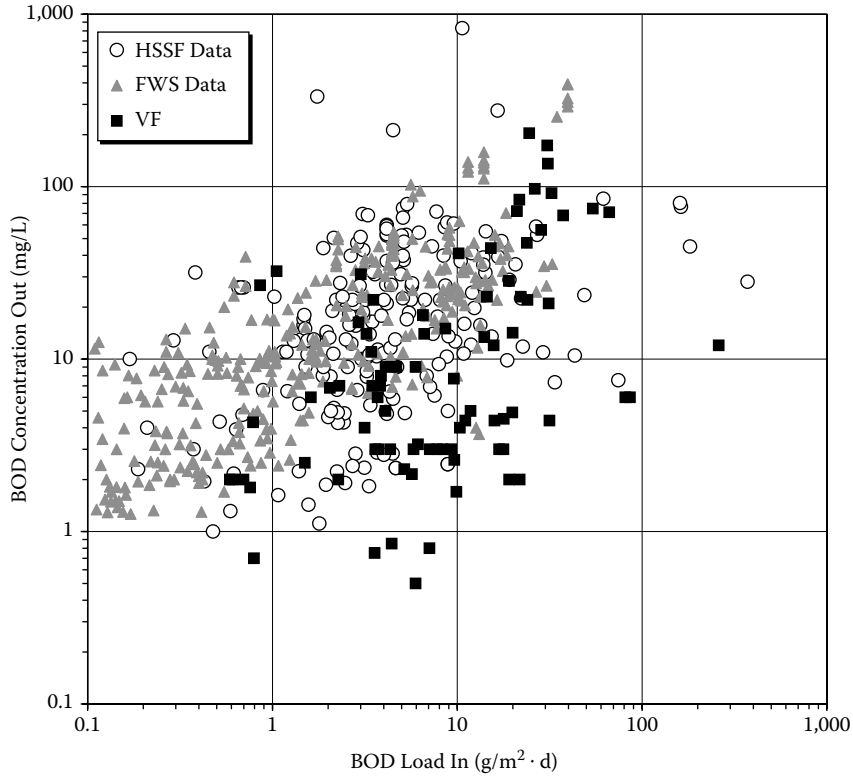



**FIGURE 16.12** Overlay of FWS, HSSF, and VF BOD loadings and effluent concentrations.

analyses, such as that in Kadlec and Knight (1996), suffer from two difficulties: the North American experience of 15 years ago was very limited in numbers of systems and the length of their operations, and the calibration ranges of FWS and HSSF systems were quite different. To compare types, FWS calibrations had to be extrapolated to higher loadings, and HSSF, to lower loadings. As is now known, such extrapolations are not warranted, and can be greatly in error (Kadlec, 2000).

At the time of this writing, there are much more extensive and numerous data sets available, so that comparisons need not rely on extrapolations. The foundation for accomplishing this comparison has been set down in Part I for the various common pollutants. There are two obvious avenues of comparison: loading charts and *k*-values, which were both used in Kadlec and Knight (1996). However, both can now be updated based on much more information. In the case of loading charts, the (desirable) overlap in loading ranges has increased. In the case of *k*-values, the underlying hydraulics can now be better accounted.

Superpositions of the loading data for HSSF, FWS, and VF wetlands are shown in Figure 16.12, 16.13, and 16.14 for BOD, ammonia, and fecal coliforms, respectively. Other pollutants behave similarly. It is clear from these scatter plots that the respective data clouds overlap virtually entirely for HSSF and FWS wetlands. There is no difference when they are compared on this areal basis. At the level of this broad perspective, this means that a square meter of FWS will do the same job as a square meter of HSSF. However, VF wetland data are set apart on these graphs, with VF systems typically providing lower outlet concentrations at equivalent loadings. This has, of course, been well documented in the literature for reduction of ammonia.

The loading chart method does not separately account for changes in inlet concentrations and hydraulic loading (or detention time); but first-order areal rate coefficients do. Therefore, it is useful to examine the various calibrations that have been set forth in Part I. Table 16.11 provides the central percentiles of the distributions of *k*-values for the various common contaminant targets. For BOD, FWS rate coefficients are higher, except for the tertiary application zone. FWS is also better for $NH_4$-N, TKN, and TN, but slightly worse for $NO_x$-N. The horizontal subsurface systems appear slightly better for fecal coliform reduction. Overall, the message is that there is not a large difference between *k*-values for FWS and HSSF wetlands, and the advantage may be to either type, depending on which contaminant is under consideration. From the standpoint of treatment performance per unit area, as embodied in the *k*-values, there is little discrimination. This is supportive of the same conclusion reached from the loading chart approach.

VF wetlands appear to be slightly better for removal of BOD and ammonia, but because the nitrate reduction is not much different, the reduction of total nitrogen is not superior. This is also supportive of the same conclusion reached from the loading chart approach.

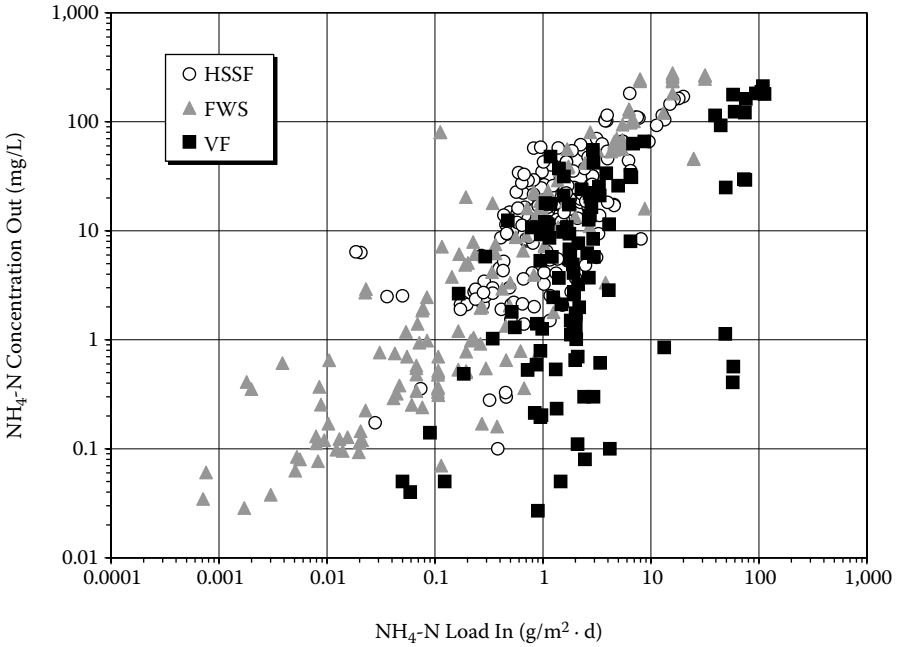



**FIGURE 16.13** Overlay of FWS, HSSF, and VF ammonia loadings and effluent concentrations.

As discussed in Chapter 23, it is not just economics alone that favor the choice of HSSF wetlands. Factors other than reduction performance are also important in the selection process. The principal reasons for selecting the HSSF option over the FWS option are concerns about:

- Human health via contact with untreated waste-water
- Mosquito control
- Odor control
- Minimizing wildlife interactions within the wetland

In the United States, human health regulations do not allow the potential for contact between humans and raw wastewater. The perceived risk of contact with pathogens is too great. This in turn means that no treatment system can allow such contact. The raw sewage facilities at mechanical plants and lagoons are strongly fenced, with required setbacks from human activities. If wetlands are to be used for raw wastewater (or primary effluent), and are unprotected from human access, then the HSSF option is necessary to ensure that there will be no human contact with the water being treated.

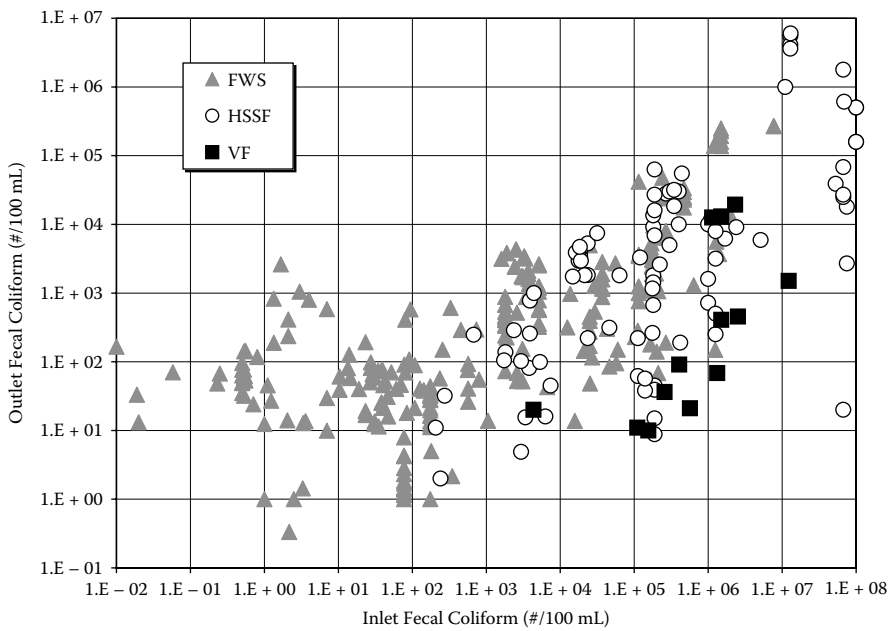



**FIGURE 16.14** Overlay of FWS, HSSF, and VF influent–effluent fecal coliform concentrations.

**TABLE 16.11**
**Summary of First-Order Rate Constants for Selected Parameters**

| $C_i$ (mg/L) | BOD *Tertiary* 0–30 mg/L | BOD *Secondary* 30–100 mg/L | BOD *Primary* 100–200 mg/L | BOD *Super* >200 mg/L | Ammonification ORG-N | Nitrification $NH_4$-N | Denitrification $NO_x$-N | TKN | TN | TP | FC |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **FWS** | | | | | | | | | | | |
| *P* | 1 | 1 | 1 | 1 | 3 | 3 | 3 | 3 | 3 | 3.4 | 3 |
| *C**, mg/L | 2 | 5 | 10 | 20 | 1.5 | 0 | 0 | 1.5 | 1.5 | 0.002 | 40 |
| 30th percentile *k*, m/yr | 16 | 16 | 23 | 54 | 10.7 | 8.7 | 18.5 | 6.1 | 6.6 | 4.5 | 49 |
| 50th percentile *k*, m/yr | 33 | **41** | 36 | 189 | **17.3** | **14.7** | **26.5** | **9.8** | **12.6** | **10.0** | **83** |
| 70th percentile *k*, m/yr | 79 | 67 | 112 | 439 | 27.4 | 45.1 | 33.6 | 13.6 | 24.2 | 16.7 | 177 |
| **HSSF** | | | | | | | | | | | |
| *P* | 3 | 3 | 3 | 3 | 6 | 6 | 8 | 6 | 6 | * | 6 |
| *C**, mg/L | 1 | 5 | 10 | 15 | 1 | 0 | 0 | 1 | 1 | * | 0 |
| 30th percentile *k*, m/yr | 36 | 24 | 15 | 21 | 8.8 | 5.2 | 32 | 4.8 | 4.7 | * | 56 |
| 50th percentile *k*, m/yr | 86 | **37** | 25 | 66 | **19.6** | **11.4** | **42** | **9.1** | **8.4** | * | **103** |
| 70th percentile *k*, m/yr | 224 | 44 | 44 | 114 | 38.0 | 18.8 | 73 | 14.6 | 14.2 | * | 181 |
| **VF** | | | | | | | | | | | |
| *P* | 6 | 6 | 6 | 6 | 6 | 6 | 6 | 6 | 6 | * | 6 |
| *C**, mg/L | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | * | 0 |
| 30th percentile *k*, m/yr | 22 | 40 | 53 | 48 | 24.0 | 23.9 | 20.3 | 14.2 | 4.1 | * | 76 |
| 50th percentile *k*, m/yr | 63 | **56** | 76 | 71 | **27.2** | **52.8** | **40.8** | **33.1** | **5.2** | * | **112** |
| 70th percentile *k*, m/yr | 105 | 79 | 122 | 93 | 37.5 | 97.3 | 85.0 | 47.0 | 10.3 | * | 201 |

* = insufficient data to determine parameters.

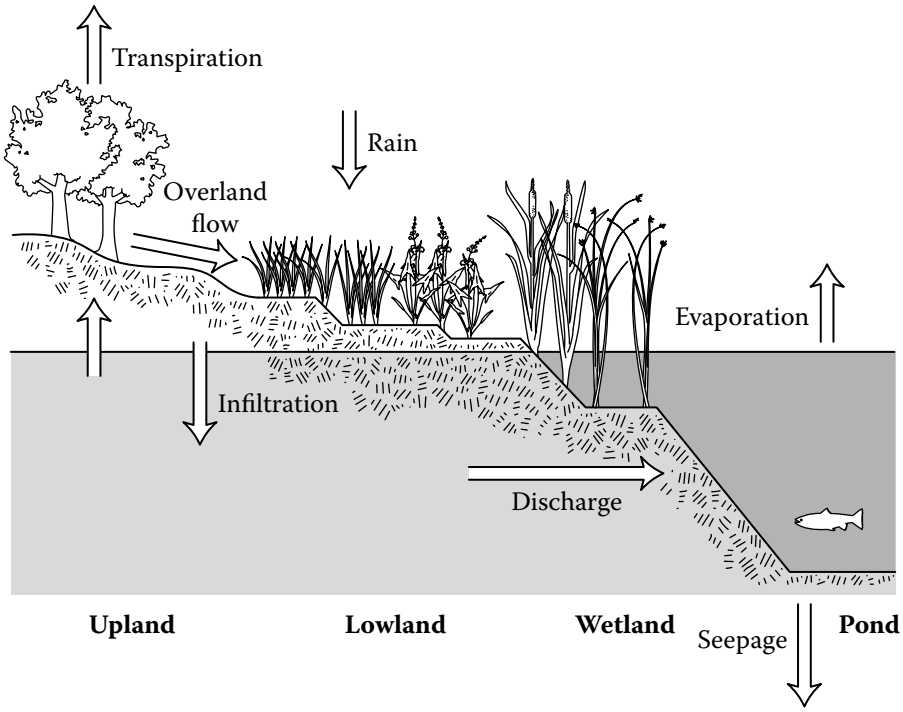



**FIGURE 16.15** Natural systems for wastewater treatment borrow from natural gradient and hydrologic regimes.

Although insect management is possible to a degree in FWS wetlands (see Chapter 22), there are potential nuisance and health problems, particularly from mosquitoes. Mosquitoes require surface water of some sort to breed and develop, and hence FWS wetlands may provide suitable habitat. The degree to which this added habitat exacerbates the regional milieu is very site specific, with greater possibilities for real and perceived problems in warm arid climates where there are few natural wetlands. The choice of HSSF wetlands in these situations can provide an acceptable alternative to costly control measures in FWS systems, such as expensive larvacide and adulticide applications. In a similar way, the choice of a HSSF system avoids gross oversizing (as has sometimes occurred in the southwestern United States) to allow for hydraulic and vegetation patterns that are inimical to mosquito activity.

Raw wastes also can emit significant odors, usually due to reduced sulfur compounds or decomposition of organic matter. Because wetlands function with a vegetative cycle that includes decomposition of organic matter, they are a potential source of odors under most circumstances, even with no wastewater added. However, natural wetland odors are typically not viewed as objectionable, whereas the sulfurous emanations of wastewater are universally disliked. Odor emission is also present for wastes other than sewage. For example, potato and other food processing wastewaters can create extreme odors, not only aesthetically objectionable, but also in some cases intolerable for human activity (Burgoon *et al*., 1999). Wetlands are no worse in this regard than the pretreatment lagoons that may precede them, but nonetheless odor situations are to be avoided. Setbacks and other measures can help in control, but keeping the water below ground, and under a vegetative canopy, is probably the easiest odor control measure.

Wildlife invariably interacts with treatment wetlands, and to a greater degree for FWS systems (see Chapter 19). Although such interactions are usually favorably looked upon, that is not always the case. For instance, flying waterfowl are a distinct threat to aviation if they occur in large numbers near airports (Naval Facilities Engineering Service Center and Wetland Solutions Inc., 2004). In other applications, wildlife represents a potential threat to system operation, and may require oversizing of FWS systems to accommodate their impacts (Kadlec *et al*., 2007). SSF wetlands are less attractive to many sorts of birds and animals, and choosing HSSF or VF wetlands will minimize animal interactions in most cases.

## Other Natural Systems

Wetlands of various kinds form a subset of natural systems technologies. A spectrum of constructed ecosystems have been devised, all of which emulate some portion of the gradients and hydrologies found in nature (Figure 16.15). In this book, the focus is on those ecosystems that contain emergent plants that are obligate to the definition of an inundated or periodically saturated wetland environment. However, it has been suggested that on the dry end of the spectrum, overland flow systems resemble FWS wetlands, and in fact may be described by the same sorts of models (Wong *et al*., 2006). At the wet end of the spectrum, wetlands transition into ponds, and there are not large differences in observed performances

**TABLE 16.12**
**Unit Ecosystems that May Be Used, Usually in Combination, for Treatment of Various Wastewaters**

| Unit Ecosystem | Recommended Information Sources |
|---|---|
| **Constructed Wetlands** | |
| FWS | This book |
| HSSF | This book |
| VF | This book |
| Alkalinity producing systems | Younger *et al*., 2000; PIRAMID, 2003 |
| **Related Natural Technologies** | |
| Slow-rate land application | Crites and Tchobanoglous, 1998; WEF, 2001; Crites *et al*., 2006 |
| Overland flow | Crites and Tchobanoglous, 1998; WEF, 2001; Crites *et al*., 2006 |
| Rapid infiltration | Crites and Tchobanoglous, 1998; WEF, 2001; Crites *et al*., 2006 |
| On-site drainfield filtration | Crites and Tchobanoglous, 1998; WEF, 2001; Crites *et al*., 2006; Wallace and Knight, 2006 |
| Sand filters | Metcalf and Eddy, 1991; Crites and Tchobanoglous, 1998 |
| Anaerobic ponds | WEF 2001; Shilton, 2005; Crites *et al*., 2006 |
| Facultative ponds | WEF 2001; Shilton, 2005; Crites *et al*., 2006 |
| Aerated ponds | WEF 2001; Shilton, 2005; Crites *et al*., 2006 |
| High-rate algal ponds | WEF 2001; Shilton, 2005 |
| Floating aquatic plant | Crites and Tchobanoglous, 1998; WEF, 2001; Shilton, 2005; Crites *et al*., 2006 |
| **Emerging Natural Technologies** | |
| Algal turf scrubbers | Current scientific literature |
| FWS algal dominated | Current scientific literature |
| **Natural Wetlands** | |
| FWS marsh | Kadlec and Knight, 1996; this book |
| FWS woody swamp | Kadlec and Knight, 1996; this book |

in pollutant reduction for many contaminants between ponds and wetlands (Kadlec, 2005e). Consequently, the designer is faced with a decision process that may be heavily conditioned by the regulatory political "climate." Alternatives often need to be compared, and a combination of unit ecosystems given consideration.

There are many literature sources that may be consulted for information on the performance of the various unit ecosystems (Table 16.12). Many of these sources have been updated within the past few years. Infiltration systems may be variously identified as slow-rate, rapid, on-site or soil-aquifer treatment. All rely on processes associated with vertical movement of water in the unsaturated or saturated zones of the soil. Slow-rate systems may also rely on plant uptake and harvest. Overland flow consists of vegetated fields with very shallow (a few centimeters), and perhaps intermittent, flow at moderately steep slopes (1–8%). Plants need not be wetland obligates, and most often are not; rather, upland grasses are employed. The design of such systems is usually on a rudimentary loading basis.

Ponds are typically deep enough to preclude the presence of emergent plants, often on the order of one to two meters. If the water is on the order of half a meter deep, and the system has no emergent plants, it may be termed either a shallow lagoon or an unvegetated FWS wetland. Design guidelines abound, with the most recent compilation in Shilton (2005).

There are new unit ecosystems that show promise for appropriate niches in treatment trains. These include intensively managed and harvested algal systems, termed algal turf scrubbers (Craggs *et al*., 1996a), and periphyton dominated systems for phosphorus reduction (Kadlec and Walker, 2004).

Finally, we separate natural wetlands from the category of constructed natural systems. This is done for two reasons: current wetland protection regulations preclude the use of natural wetlands in the large majority of situations in North America, and natural wetlands in general fit nicely into the performance projections for constructed FWS systems. Natural wetlands are impacted by the addition of either more water or more contaminants than they have historically been exposed to. Thus, the use of natural wetlands typically requires an assessment of the impacts of the potential wastewater addition on the ecology of that wetland. A sizeable performance database exists, because of past projects that have used existing wetlands as receiving waters for partially treated discharges. In the past, this has often been a matter of convenience, rather than design, because of the proximity of a project to wetland receiving waters (for example, Brillion, Wisconsin [Spangler *et al*., 1976], or Kinross, Michigan [Kadlec and Bevis, 1990]).

It is usually not optimal to utilize a wetland as the sole element of a natural system to meet final requirements. The integration of the wetland into an engineered natural system often makes better economic sense. The phosphorus-binding capacity of soils, and their nitrification potential, can complement the wetland's capabilities for anaerobic

processes and denitrification. Ponds can provide sedimentation and water storage in a more land-economical way. The techniques presented in this treatise are applicable to wetlands used as pieces of integrated natural systems. Various combinations of treatment wetlands of different types, as well as other unit ecosystems, are considered in Chapter 24. Here, the designs of the three common types of wetlands are considered as individual elements of the integrated treatment system.

## 16.7 PRE- AND POST-TREATMENT REQUIREMENTS

Wetlands are not capable of entertaining all levels of all pollutants, because they are biological systems that can only exist in a certain envelope of potential concentrations of certain contaminants. Therefore, the designer must adhere to the requirements of the selected ecology, or the wetland may suffer injury or destruction. Inflows may require pretreatment to allow a functioning ecosystem to exist. The discharge requirements of receiving waters may be outside the range that the wetland can produce, due primarily to wetland background concentrations. Thus, posttreatment may be needed to meet regulatory limits.

### Pretreatment

FWS wetlands in North America normally receive municipal water of approximately secondary quality or better. This is in contrast to the subsurface technology (HSSF and VF) in Europe, which typically treats settled or primary influents. Urban and agricultural runoff waters are also relatively dilute influents. Industrial wastewater treatment wetlands are usually added to the end of a "conventional" treatment process. Some fairly strong food processing wastewaters are being treated with integrated natural systems, but these systems usually have some form of primary treatment, such as a clarifier, lagoon, or filter. Pretreatment may also be required to ensure the survival of the treatment wetland.

### Wetland Survival Limits

As treatment wetlands are used in new and challenging applications, there is an increasing risk of inadvertently designing for conditions that the wetland cannot tolerate. A few of the basic considerations are listed here.

### Excessive Solids

Sediment accumulation is one of the few processes in wetlands treatment that has a foreseeable requirement for maintenance. Sooner or later, sediment accumulation will require some sort of removal activity or reconfiguration of the wetland. Based on present experience, such maintenance will be on the schedule of decades for FWS wetlands unless there is an unusually high content of settleable solids in the incoming water. In that event, a settling basin or pond is a logical pretreatment device. Stormwater wetlands typically embody such elements. For HSSF wetlands, there is an increasing body of knowledge that indicates the cross-sectional loading is a critical design parameter (see Chapter 21). For VF wetlands, it is apparent that resting intervals between loading periods are necessary to control solids accumulation and associated bed clogging problems (see Chapter 21).

Upsets in pretreatment elements of the overall treatment train can episodically pass large quantities of TSS to downstream treatment wetlands, which can pose difficulties for both FWS and HSSF wetlands. In the case of HSSF systems, the solids can form a blanket that smothers the vegetation, probably by overpowering the plants' abilities to modify their rhizosphere to avoid oxygen deficiencies. Figure 16.16 shows this phenomenon at the Hardin, Kentucky, HSSF facility, which has a package activated sludge plant for pretreatment. Sludge overflows blanketed the inlet, clogging the bed and killing the vegetation. This problem also occurs for FWS wetlands, as evidenced in Figure 16.17 for the Listowel, Ontario, facility.

Iron precipitates can form precipitates that can be problematic in treatment wetlands. Wastewaters may contain reduced iron compounds that can become oxidized in the wetland environment. Under aerobic conditions, oxyhydroxides precipitate to form ochreous sludge that can coat plants in a FWS wetland, or potentially clog the media in an SSF system (Figure 16.18).

Inlet deep zones may serve to protect a FWS wetland from TSS overloads, but not if such loads are sustained (see Figure 7.13). Solids overload protection is thus a necessary part of treatment wetland design.

### Phosphorus Precipitates

Phosphorus reduction is one of the least efficient processes in wetland treatment. Low TP concentrations can



**FIGURE 16.16** The Hardin, Kentucky, HSSF *Phragmites* and *Schoenoplectus* (*Scirpus*) wetlands received periodic sludge overflows from the pretreatment device (Imhoff tank). As a result, the inlet to the bed became a sludge bed with no live plants.



**FIGURE 16.17** The Listowel, Ontario, FWS *Typha* wetlands received episodic sludge overflows from the pretreatment device (primary lagoons). As a result, the inlet to the wetland became a sludge bed with no live plants. Incoming water forms a rivulet across the sludge.

be reduced to even lower ones, but large phosphorus load removal requires a large wetland area. Consequently, some pretreatment for the reduction of high TP concentrations is normally cost effective. Iron and alum addition are the most frequent choices, both of which directly or indirectly remove phosphorus, by precipitation and co-precipitation respectively. The settling of these solids is often increased by the addition of polymeric flocculating additives. In the case of FWS wetlands, the point of addition needs to be upstream of the wetland, because there is not an effective way to provide chemical contacting in the wetland itself. However, if the precipitates carry over into the wetland, they can create unacceptable conditions in the sediments, because of aluminum toxicity or because co-precipitation produces the ferric materials discussed previously.

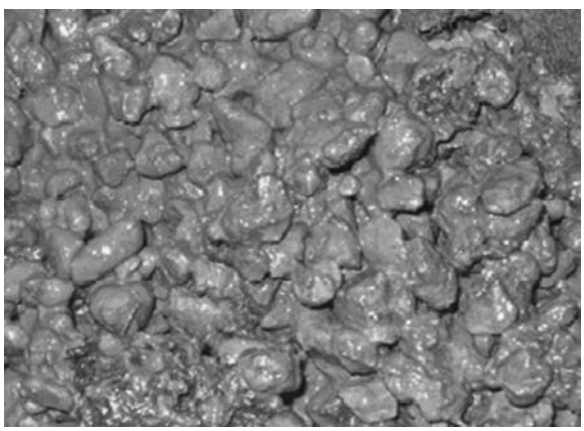


**FIGURE 16.18** The Anamosa, Iowa, HSSF wetland, vegetated by a mixture of local species, received landfill leachate with a moderately high iron content. As a result, the media became clogged with iron precipitates. (Photo courtesy J. Nivala.)

### Excessive Nitrogen

Nitrogen concentrations of wastewaters are among the most variable across various possible sources. For instance, nitrified secondary waters may have ammonia at less than 1.0 mg/L, but landfill leachate can reach 1,000 mg/L. At the extreme low end, there may not be enough nitrogen to sustain wetland vegetation after the developing ecosystem has "mined" the available preexisting soil nitrogen. At the extreme high end, inorganic nitrogen of over 100–200 mg/L may pose a threat to vegetation, because of fertilizer "burning" effects. Therefore, the introduction of strongly ammoniacal influents to treatment wetlands must be avoided. Fortunately, the fairly simple expedient of recycling effluent flows to dilute influent concentrations can usually solve this problem (Kadlec, 2003c). Other nutrients, phosphorus and potassium, have not been observed to create similar difficulties.

### Strong Specialty Chemicals

Treatment wetlands are increasingly used for treating waters that contain individual chemicals, which, if in excess, can threaten the wetland ecosystem. As an example, a chemical plant effluent at Estarreja, Portugal, is successfully treated using a HSSF *Phragmites* wetland. However, a high-concentration pulse of nitric acid found its way into the wetland, causing the destruction of vegetation in the inlet zone (Figure 16.19).

It is noteworthy that wetland plants have evolved to include varieties that can withstand extreme conditions. Low pH varieties can live in acidic conditions, down to pH = 2 to 3, as demonstrated by the acid bog vegetation of northern latitudes. Conversely, other wetland plants can live in very basic waters, such as those adapted to alkaline fens and some of the prairie potholes. Thus, it is necessary to consider the tolerances of the selected wetland plants to the conditions



**FIGURE 16.19** The Estarreja, Portugal, HSSF *Phragmites* wetland received an accidental discharge of nitric acid from pretreatment. As a result, the inlet to the wetland (right) became devoid of living plants. The outlet zone (left) maintained a healthy stand.

that they are to endure. Acid mine drainage wetlands are in the category of those that require consideration of strong chemical content.

### Excessive Sulfide

Freshwater wetland plants are intolerant of sulfide (Lamers *et al*., 2002; see the sulfur section of Chapter 11). Therefore, if the water entering a treatment wetland contains sulfate, there exists the possibility of reaching high levels of sulfide in anaerobic treatment wetland sediments. This has the potential to be devastating to FWS plants, and has been implicated as a possible cause of the total demise of vegetation (*Scirpus*) at the Tres Rios wetlands (Figure 16.20). Yet, other wetland plants can endure stronger sulfidic conditions, as evidenced by the ability of *Phragmites* to survive in waters nearly saturated with sulfate, and the attendant sulfide created by anaerobic conditions.

### Oxygenation of Wetland Influents

Intuitively, it seems that adding oxygen to waters entering treatment wetlands is a useful strategy. Several wetland processes, such as BOD reduction and nitrification, are oxygen dependent. The incoming water may be oxygen-poor because of pretreatment processes. Therefore, preoxygenation has been used in some applications (Figure 16.21). However, it is only possible to raise the oxygen content of the water to saturation by this technique, typically 6–10 mg/L (see Chapter 5). For a hydraulic loading rate of 6 cm/d (central tendency), the



**FIGURE 16.20** The Tres Rios, Arizona, FWS *Scirpus* wetlands received nitrified secondary wastewater from an activated sludge plant. After two years, the vegetation died, possibly due to sulfide toxicity.



**FIGURE 16.21** The Lakeland, Florida, FWS inlet cascade.

addition of 5 mg/L of dissolved oxygen (DO) only adds 0.3 $gO/m^2 \cdot d$. That is a small proportion of the DO needed for aerobic microbial processing in most treatment wetlands.

### Postwetland Requirements

A common regulatory requirement is the reduction of fecal coliform bacteria to low levels in the wetland effluent. Because FWS wetlands are frequented by warm-blooded animals and birds, the background levels of fecal coliforms often do not comply with typical receiving water standards. This particular bacterial indicator does not distinguish the animal of origin, and hence the conservative regulatory stance is to ascribe the total measure to human origins. Therefore, the attainment of low fecal coliform levels may need to be accomplished by postwetland disinfection, via chlorination, ozonation, or UV irradiation. In the event that chlorination is used, there remains an issue of the residual chloramines that are formed, as well as the levels of residual chlorine (see Chapter 12). Concerns for the effects of disinfectants, such as chlorine, upon the treatment wetland microbial consortia that conduct much of treatment have resulted in dechlorination facilities in some cases.

The low amounts of dissolved oxygen found in many FWS and HSSF treatment wetland effluents may also violate receiving water standards in many cases. In Europe, the DO content of receiving streams is not usually a restriction. Natural wetlands (meaning those not used for treatment) do not often meet the DO requirement of approximately 5 mg/L required for many receiving streams in the United States. For instance, the lightly loaded wetlands used for Everglades protection in Florida, which have essentially no BOD load nor any ammonia load, have a high exceedance rate for DO (SFWMD, 2006). These systems do not on average meet the 5 mg/L level required by the Class III water standards of the state of Florida, thus necessitating new rules and regulations for wetlands. In general, wetlands fail DO regulatory requirements, and some form of postwetland aeration may then be required. If the site topography permits, this can be a simple aeration cascade, as previously illustrated in Figure 16.21.

The peculiarities of discharge regulations can lead to other posttreatment steps. For instance, the Lakeland, Florida, system was subject to a very low TSS discharge requirement of 5 mg/L annual average, because of the Grizzle–Figg law in Florida. The last element in the train of wetland cells was a water hyacinth cell (*Eichhornia crassipes*), which lost full cover due to insect attack. The resulting open water allowed phytoplankton development, with resulting TSS of 7.6 mg/L. As an interim measure, a dye (Aquashade™) was added to block the radiation needed by the algae. Subsequently, the wetland cells were reconfigured to avoid the terminal open water, and compliance was regained.

## SUMMARY

In this chapter, the information-gathering phase of treatment wetland design is described. Design of wetland treatment systems or any other pollution control technology depends on the best possible knowledge of influent wastewater characteristics. Some general guidelines are presented for pollutant concentrations and loads from a variety of wastewater categories that might be applied to wetland treatment systems. Site-specific conditions, and their potential constraining effects on design, are collected and analyzed. The meteorology that the wetland will experience is determined. Importantly, the goals that need to be accomplished are to be carefully set down for use in subsequent design calculations. In the performance-based design of the wetland, all of this information plays a quantitative role.

Table 16.13, adapted from the Alberta 2000 guidelines, provides a sample checklist of the factors that affect the selection and sizing of the wetland (Alberta Environment, 2000). It documents the basic information required to determine the feasibility of using a treatment wetland for polishing of a wide variety of industrial, agricultural, and municipal wastewater discharges. It is intended to act as a general screening tool. General information including soil type, permeability, and site topography must be confirmed with testing, surveying, etc., prior to the final stage of design. Data include site location, type of wastewater pretreatment, pretreated water quality, hydraulic loading, soil conditions, land availability, effluent water quality objectives, current land use, topography, and climatic factors. The collected data must be reviewed, and contaminant loadings determined and compared to provincial guidelines. Further treatment of the wastewater may be required prior to inflow to the wetland system, either in the existing wastewater treatment plant or as a separate treatment unit. It must also be determined if portions of the treatment wetland system will be accessible to the public. This will have some bearing on the type of treatment wetland that will be most suited to the treated wastewater source.

## TABLE 16.13
## Predesign Checklist

**Potential Wetland Location Site Data**

Site name: _____

- City/community: _____
- Population: _____

Other anticipated wetland uses (nature study, hunting, aquaculture, other): _____
Key/sensitive wildlife habitat: _____
Site substrate material (e.g., sand, clay, muck, sandy clay, clayey sand, etc.): _____

- Substrate permeability: high ____ medium ____ low ____

% Vegetation Cover:

- Submergent _____
- Emergent _____
- Meadow _____
- Forest _____

Land area available: _____
Proximity to water/wastewater source: _____
Current site land use: _____

- Zoning: _____
- Ownership: _____

Adjacent land use: _____

- Zoning: _____
- Ownership: _____

Presence of existing or potential limiting land use (e.g., environmentally sensitive areas, etc.): _____
_____
Protected species historical or archeological resources on or near site: _____
_____
Aquifers, aquitards, or natural wetlands, note location:_____ _____
_____
% of Available land area covered by natural wetlands:_____
Type of natural wetland (marsh, open water, floating aquatic, etc.): _____
Dominant plant species: _____
Site topography (flat, rolling, steep, other): _____

- Is the wetland/site landlocked? Yes___ No ___
- If no, what water body will/does the treatment wetland discharge to? _____

Water body classification: _____
Wetland outlet location and description: _____
Upland buffer zone description: _____

(*Continued*)

**TABLE 16.13 (CONTINUED)**
**Predesign Checklist**

## Climatic Factors

Site elevation _____

Number of frost-free days _____

Annual and monthly temperature

| Jan | Feb | Mar | Apr | May | Jun | Jul | Sep | Oct | Nov | Dec | Annual |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | | | |

Annual and monthly snowfall

| Jan | Feb | Mar | Apr | May | Jun | Jul | Sep | Oct | Nov | Dec | Annual |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | | | |

Annual and monthly rainfall

| Jan | Feb | Mar | Apr | May | Jun | Jul | Sep | Oct | Nov | Dec | Annual |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | | | |

Annual and monthly evapotranspiration

| Jan | Feb | Mar | Apr | May | Jun | Jul | Sep | Oct | Nov | Dec | Annual |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | | | |

## Potential Influent Quantity and Quality

Design flows _____

Stormwater to watershed area: _____

Define watershed border and note watershed area:___ ha

Runoff coefficient: _____

Wastewaters (municipal, industrial, etc.) _____

Wastewater pretreatment: _____

Operating season (months): _____

Period of record

- Start _____ (year)
- End _____ (year)

Average flow ($m^3/d$): _____

Metals (list) _____

_____

Pesticides/herbicides (list) _____

_____

Organics (list) _____

_____

## Parameter Concentrations and Loadings

| | | | |
|---|---|---|---|
| $BOD_5$ | _____ | mg/L _____ | kg/d |
| TSS | _____ | mg/L _____ | kg/d |
| TDS | _____ | mg/L _____ | kg/d |
| $NH_4$-N | _____ | mg/L _____ | kg/d |
| $NO_3 + NO_2$ | _____ | mg/L _____ | kg/d |
| TN | _____ | mg/L _____ | kg/d |
| TKN | _____ | mg/L _____ | kg/d |
| Organic nitrogen | _____ | mg/L _____ | kg/d |
| TP | _____ | mg/L _____ | kg/d |
| $SO_4/S_2$ | _____ | mg/L _____ | kg/d |
| Alkalinity | _____ | mg/L _____ | kg/d |
| Chloride | _____ | mg/L _____ | kg/d |
| Dissolved oxygen | ____ | mg/L _____ | kg/d |

Conductivity_____

pH _____

Temperature _____ °C

Fecal coliform _____ CFU/100 mL

E. coli _____ CFU/100 mL

## Treatment Wetland System Outflow Targets

Approval permit limits to be met at

- wetland inflow _____
- wetland outflow _____

Approved flow (per day, week, month, year) _____

Approval duration (annual, seasonal, monthly, other) _____

$BOD_5$ _____ mg/L _____ kg/d

TSS _____ mg/L _____ kg/d

$NH_4$-N _____ mg/L _____ kg/d

TN _____ mg/L _____ kg/d

TP _____ mg/L _____ kg/d

Dissolved oxygen _____ mg/L

Fecal coliform _____ CFU/100 mL

pH _____

# 17 Sizing of FWS Wetlands

A performance-based procedure is unavoidable for FWS wetlands because they address too many applications with varying degrees of source strength and pretreatment. In general, the targets are also variable because of the differences in regulatory requirements among countries, states, and discharge recipients. Loading specifications, whether as pollutant kilograms per wetland hectare per year or as wetland hectares per person equivalent, have not, and, in general, cannot be developed to deal with the spectrum of situations created by FWS diversity of applications. It is not an exaggeration to say that each FWS wetland is unique.

Design of FWS wetlands may be roughly divided into two categories: sizing calculations and physical specifications. Sizing requires characterization of the incoming water and regional meteorology as well as the goals of wetland treatment, as discussed in Chapter 16. Here, a comprehensive sizing strategy is presented, based on the information assembled. Chapter 18 will deal with physical considerations, including the number of cells, layout, depth, bathymetry, soils and plants, structures, and lining. It is recognized that wetlands are almost never stand-alone treatment devices, but rather form part of a treatment train. Other components may be mechanical, such as clarifiers or filters, or more natural, such as settling basins or lagoons. More than one type of wetland may be involved at the same site—FWS, VF, and HSSF. This chapter focuses on only the FWS component of treatment systems.

Based on the performance-based sizing algorithm introduced in Chapter 15, the key features described in this chapter are the following:

Parameter selection and calculation

6. Select rate constants and seasonality. (Select plant community type.)
7. Select DTD (hydraulic) efficiency = *P*-value. (Select compartmentalization.)
8. Adjust area till goals are all met.

Constraint checks and iteration

9. Set estimated growth cycle.
10. Check biogeochemical cycles for consistency.
11. Check chemical constraints.
12. Check loading graph for risk assessment.
13. Adjust for seasonality.

Repeat this procedure as needed to meet design goals.

The complexity of wetland behavior and, hence, of the sizing calculation is such that a single equation for wetland area cannot be written. At this point in the evolution of the treatment of wetland design, the sizing procedure must move out of the realm of the pocket calculator and into the world of spreadsheet computations. The single formula, black-box approach will not suffice.

## 17.1 POLLUTANT REDUCTIONS AND PERFORMANCE COMPUTATIONS

The information collected is utilized as a basis to forecast the area needed to achieve the goals of the project. The procedure outlined here is for a steady-flow situation. Different flows may need to be considered to deal with daily or seasonal peaking. Different seasons may need to be considered, so that the "bottleneck" period of the year is identified and analyzed.

Many literature sources provide single equations into which anyone may insert numbers to compute a wetland area. Although such an option would be very convenient, it is not realistic except for a few highly specialized cases. Accordingly, the procedure given here explores the effect of changing wetland area (or equivalently, detention time) on the forecasted effluent concentrations of contaminants. It is presumed that the designer will conduct the design calculations via a computer spreadsheet so that design changes may be easily explored.

Wetland hydrology is first determined as a necessary precursor to area calculations. Annualized calculations are considered first; the effects of season will be considered later in the chapter.

### Water Budget

The annual water budget forms the basis for understanding pollutant reductions and the area required. Assuming a constant water level, the flows out of the wetland may be computed from the inflow and meteorological data. Some infiltration is also assumed here if leakage is present in the wetland. The relevant equations are

$$Q_o = Q_i + A \cdot (P - ET - I) \tag{17.1}$$

$$q_o = q_i + (P - ET - I) \tag{17.2}$$

where

$A$ = wetland area, $m^2$
$ET$ = evapotranspiration rate, m/d
$I$ = infiltration rate, m/d
$P$ = precipitation rate, m/d
$q$ = hydraulic loading rate, m/d
$Q$ = flow rate, $m^3/d$

The inlet hydraulic loading will be increased by rainfall (≈0.5–1.5 m/yr) and decreased by evapotranspiration (*ET*) (≈ 0.5–1.5 m/yr). However, there may be seasonal imbalances. These amounts are important if the wetland is to have a very low hydraulic loading or, correspondingly, a long detention time. The ratio $a = (P - ET)/q_i$ is the atmospheric augmentation.

Evapotranspiration (rain) has two effects: lengthening (shortening) of detention time and concentration (dilution) of dissolved constituents. The use of an average flow rate compensates for altered detention time but not for dilution or concentration. The fractional error in a first-order model prediction of concentration due to flow averaging is approximately equal to $a$, for $a > -0.5$. Thus, if 25% of the inflow evaporates, use of a first-order model with average flow predicts concentrations 25% lower than required by the mass balance. If rain adds 25% to the flow, use of a first-order model predicts concentrations 25% higher.

A possibly important feature of *ET* is that the transpiration carries water into the root zone (see Part I, Chapter 4). Therefore, if the pollutant mass balances are done on surface water, then the transpiration component is the same as infiltration: it carries materials into the soil. For a fully vegetated wetland, somewhat more than half of *ET* is attributable to transpiration. Transpiration is typically about one half to two thirds of *ET* (Kadlec, 2006c):

$$T = (ET - E) = \alpha ET \tag{17.3}$$

where
$E$ = evaporation, m/d
$T$ = transpiration, m/d
$\alpha$ = transpiration fraction of *ET*, dimensionless

The pollutant mass balances will be conducted on a cells-in-series basis (see Figure 17.1). Results of the overall water mass balance are apportioned to the cells according to the chosen number of TIS. For the first unit in the series:

$$Q_1 = Q_{in} + A_1(P - ET - I) \tag{17.4}$$

where
$A_1$ = area of tank number 1, $m^2$
$Q_1$ = flow rate out of tank 1, $m^3/d$

Flows are thus computed sequentially, from inlet to outlet, for the number of tanks chosen (*P*TIS).

The input data requirements for water mass balances are

1. Inflow ($Q_{in}$)
2. Rain ($P$)
3. Evapotranspiration ($ET$)
4. Infiltration ($I$)
5. Area ($A$)
6. Apparent number of tanks in series ($P$-value)

An example is detailed in Table 17.1. This hypothetical example is configured to include small rainfall, considerable *ET*, and some infiltration; in other words, a leaky arid region system. The net loss of incoming water is 41%, of which 44% is *ET* and 56% is infiltration.

## Pollutant Mass Balances

The TIS model is then carried forward via a sequential calculation of pollutant concentrations for each "tank" in the chosen hydraulic model (Figure 17.1).

A first-order areal model with rate constant $k$ is selected with necessary wetland background concentration $C^*$. A volumetric first-order model may also be chosen for which $k = \varepsilon h k_V$. The pollutant mass balance for the first of the wetland segments, designated by subscript "1" for steady-state,

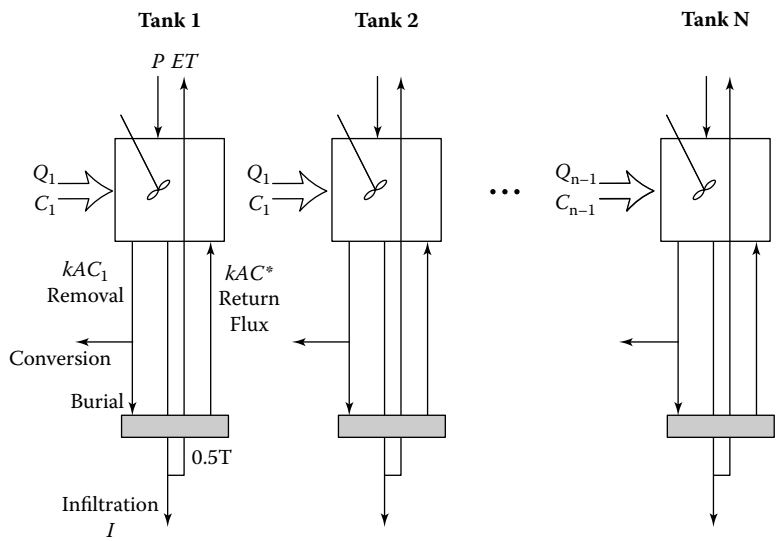



**FIGURE 17.1** Conceptual TIS model for pollutant reduction.

**TABLE 17.1**
**Water Budgets for a FWS Design Example**

**Parameters**

**Flow rate = 5,000 m³/d**
**Rain = 0.05 cm/d**
***ET* = 0.40 cm/d**
**Infiltration = 0.50**
***P*TIS = 3**
**Area = 24 ha**
Area/tank = 8 ha
Depth = 0.3 m
Porosity = 0.95
Volume/tank = 22,800 m³

| | Inflow | Tank 1 | Tank 2 | Tank 3 = Outflow | Total |
|---|---|---|---|---|---|
| Flow rate (m³/d) | 5,000 | 4,320 | 3,640 | 2,960 | — |
| Rain (m³/d) | — | 40 | 40 | 40 | 120 |
| ET (m³/d) | — | 320 | 320 | 320 | 960 |
| Infiltration (m³/d) | — | 400 | 400 | 400 | 1,200 |
| Nominal detention (d) | 13.7 | 5.3 | 6.3 | 7.7 | 19.2 |
| HLR (cm/d) | 2.08 | 5.40 | 4.55 | 3.70 | — |

*Note:* Sequential tank-to-tank calculations are based on Equation 17.4; this example has a combined detention time of 19.2 days, whereas averaging the flows leads to 17.2 days; the required input data are shown in bold.

nonuniform flow is

$$Q_1C_1 = (Q_{in}C_{in}) - (I \cdot A_1C_1) - (\alpha ET \cdot A_1C_1) - \left(k \cdot A_1 \cdot (C_1 - C^*)\right) \quad (17.5)$$

In this simple version, rainfall has been assumed to have zero pollutant concentration, but it is easy to add an atmospheric input of the pollutant if it exists. Infiltration is assumed to occur at the outlet concentration. Transpiration flow of the contaminant has been included. Combining Equation 17.4 with Equation 17.5 gives the concentration exiting the hypothetical segment number one:

$$C_1 = \frac{Q_iC_{in} + (k \cdot A_1 \cdot C^*)}{Q_1 + \left((\alpha ET)A_1\right) + (I \cdot A_1) + (k \cdot A_1)} \quad (17.6)$$

Or

$$C_1 = \frac{q_{in}C_{in} + (k \cdot C^*)}{q_1 + (\alpha ET) + I + k} \quad (17.7)$$

Note that the hydraulic loading rates in Equation 17.7 are the individual tank loading rates, not the overall system loading rates. This computation is then repeated sequentially for the remaining segments, using, in each case, the outlet concentrations and flows from the preceding unit. The wetland outlet concentration is that exiting the final hypothetical segment.

Outgoing pollutant loads are calculated as the product of the volumetric outflow (m³/d) and the outflow concentration (kg/m³).

The additional input data requirements for the pollutant mass balances are

7. Input concentration ($C_{in}$)
8. Background concentration ($C^*$)
9. Rate coefficient ($k$)
10. Transpiration fraction ($\alpha$)

The hypothetical example presented for the water budget is extended to illustrate these calculations, continuing the choice of $P = 3$ TIS. Phosphorus is chosen as the pollutant of interest, entering the wetland at 2.00 mg/L. The value of $C^*$ is chosen to be low, 0.01 mg/L. The rate constant is selected to be the median shown for phosphorus in FWS systems in Table 10.11, $k$ = 10 m/yr. Because the transpiration flux is presumed to draw phosphorus into the root zone, the fraction of *ET*, that is, transpiration, must be selected, and in the example it is picked as $\alpha = 0.5$.

The computed phosphorus concentration in the surface outflow water is $C_o = 0.62$ mg/L, or a concentration reduction of 69% (Table 17.2). However, the existence of *ET* losses and infiltration creates a different result for mass removal: 82% of the phosphorus entering does not leave with surface water. If infiltration water departs vertically downward without further treatment, then 15% of the phosphorus mass removal is due to infiltration. Thus, it is seen that load reduction and concentration reduction are two different goals, which may lead to different designs. Both these types of design sizing require mass balance computations for water and rate calculations for pollutants.

**TABLE 17.2**
**Phosphorus Budgets for a FWS Design Example**

**Parameters**

**$C_i$ = 2.00 mg/L**
**$C^*$ = 0.01 mg/L**
**$k$ = 10 m/yr**
$k$ = 0.0274 m/d
Area/tank = 80,000 m²
**Theta factor = 0.5**

| | Inflow | Tank 1 | Tank 2 | Tank 3 = Outflow | Removed | Reduction |
|---|---|---|---|---|---|---|
| Concentration (mg/L) | 2.00 | 1.42 | 0.96 | 0.62 | — | 69% |
| Load in surface water (kgP/yr) | 3,650 | — | — | 666 | 2,984 | 82% |
| Load infiltrated (kgP/yr) | — | 207 | 140 | 90 | 437 | 15% |
| Load stored (kgP/yr) | — | — | — | — | 2,547 | 70% |

*Note:* This example is based on the water budget shown in Table 17.1; sequential tank-to-tank calculations are based on Equation 17.6; the required input data are shown in bold; the shaded cells represent potential design targets.

## Interconnected Pollutants: The Case of Nitrogen

Nitrogen species interconvert, thereby linking the mass balances for organic, ammonia, and oxidized nitrogen. It is sometimes possible to disconnect these species, as, for instance, in the case of wetlands that receive nitrate but little or no organic or ammonia nitrogen. However, in many cases, it is necessary to account for the (internal) production of ammonia from organic sources, i.e., from either the incoming water or the decomposition of wetland necromass, and the internal production of oxidized nitrogen. A simple, presumed chemistry is

$$\text{ORG-N} \rightarrow \text{NH}_4\text{-N} \rightarrow \text{NO}_x\text{-N} \rightarrow \text{N}_2 \qquad (17.8)$$

In a simplified version of analysis, uptake and return from biomass is not included. The effects of the biogeochemical cycle on nitrogen will be explored in a subsequent part of the design process. The three mass balances then become linked, and the tank equations are:

$$Q_1 C_{O1} = (Q_{in} C_{O,in}) - (I \cdot A_1 C_{O1}) - (\alpha ET A_1 C_{O1}) - k_O A_1 (C_{O1} - C_O^*) \qquad (17.9)$$

$$Q_1 C_{A1} = (Q_{in} C_{A,in}) - (I \cdot A_1 C_{A1}) - (\alpha ET A_1 C_{A1}) - k_A A_1 (C_{A1} - C_A^*) + k_O A_1 (C_{O1} - C_O^*) \qquad (17.10)$$

$$Q_1 C_{N1} = (Q_{in} C_{N,in}) - (I \cdot A_1 C_{N1}) - (\alpha ET A_1 C_{N1}) - k_N A_1 (C_{N1} - C_N^*) + k_A A_1 (C_{A1} - C_A^*) \qquad (17.11)$$

where
$C_O$ = organic N concentration, mg/L
$C_A$ = ammonia N concentration, mg/L
$C_N$ = oxidized N concentration, mg/L
$k_O$ = organic N rate coefficient, m/d
$k_A$ = ammonia N rate coefficient, m/d
$k_N$ = oxidized N rate coefficient, m/d

and the subscript "in" denotes parameters associated with the influent flow for each nitrogen form.

These may be rearranged to solve the outlet concentrations for each tank:

$$C_{O1} = \left( \frac{Q_{in} C_{O,in} + k A_1 C_O^*}{Q_1 + (\alpha ET) A_1 + I A_1 + k_O A_1} \right) \qquad (17.12)$$

$$C_{A1} = \left( \frac{Q_{in} C_{A,in} + k_A A_1 C_A^* + k_O (C_{O1} - C_O^*)}{Q_1 + (\alpha ET) A_1 + I A_1 + k_A A_1} \right) \qquad (17.13)$$

$$C_{N1} = \left( \frac{Q_{in} C_{N,in} + k_N A_1 C_N^* + k_A (C_{A1} - C_A^*)}{Q_1 + (\alpha ET) A_1 + I A_1 + k_N A_1} \right) \qquad (17.14)$$

Equation 17.12 is a direct analog of Equation 17.7 for an unspecified generic pollutant. Equations 17.13 and 17.14 contain extra production terms from ammonification in the ammonia balance—and nitrification in the oxidized nitrogen balance. The three must be solved sequentially—Equation 17.12 followed by Equations 17.13 and 17.14.

## Design Parameters: Sources of Information

The *P-k-C** design model is the basis for this sizing analysis; it is therefore necessary to select values of these three parameters for all pollutants of concern.

### Background Concentrations

Wetland systems are dominated by plants (autotrophs), which act as primary producers of biomass. However, wetlands also include communities of microbes (heterotrophs) and higher animals, which act as grazers and reduce plant biomass. Most wetlands support more producers than consumers, resulting in a net surplus of plant biomass. This excess material is

typically buried as peat or exported from the wetland (Mitsch and Gosselink, 1993). The net export results in an internal release of particulate and dissolved biomass to the water column, which is measured as nonzero levels of biochemical oxygen demand (BOD), total suspended solids (TSS), total nitrogen (TN), and TP. These wetland background concentrations are typically denoted by the term $C^*$. Enriched wetland ecosystems (such as those treating wastewater) are likely to produce higher background concentrations than oligotrophic wetlands. These elevated background concentrations are largely due to increased biomass cycling resulting from the higher levels of nutrients and organic carbon in the wastewater. Even land-locked wetland basins, which only receive water inputs through precipitation, will have nonzero background concentrations.

Consequently, many pollutants are not reduced to zero in treatment wetlands—including BOD, TSS, organic nitrogen, and phosphorus. However, it is important to distinguish between artifacts of data fitting and real wetland processes. Short-circuiting can lead to high values of data-fit $C^*$ for heavily loaded systems, but these high $C^*$ can be dealt with by improving the hydraulics. Independent of hydraulics, the wetland can manufacture water-phase organics and solids, and cycle nutrients into and out of the water body. The chapters of Part I contain estimates of the $C^*$-values for the common pollutants, and many literature sources provide ranges of background concentrations (U.S. EPA, 1999; IWA Specialist Group on Use of Macrophytes in Water Pollution Control, 2000; U.S. EPA, 2000a; Wallace and Knight, 2006; Crites *et al*., 2006). A summary is given in Table 17.3.

Some individual exotic chemicals are foreign to typical wetland environments and are not expected to exhibit background concentrations. Examples include halogenated hydrocarbons and pesticides.

### Number of Tanks to Be Used in the Model

The performance of the wetland depends on the number of tanks (TIS) selected—very strongly if the design is to approach wetland background concentrations or high degrees of removal (see Chapter 6). If a high degree of removal is required, it will necessitate a very large wetland with poor hydraulics, or a smaller wetland with good hydraulics. Figure 17.2 illustrates this high degree of sensitivity in the region of low $P$ for large reductions. This chart quantifies the fact that a tiny fraction of the water following a fast short-circuit will carry enough unreacted material to the outlet to make 99+% reduction almost impossible to achieve. Low $P$ numbers represent flow patterns that, by virtue of either fast forward mixing or velocity profiles with high-speed elements, carry fractions of unreacted material directly to the outlet.

The $P$-value is somewhat at the discretion of the designer. More cells and greater length-to-width ratios can increase the $P$-value. As an illustration of the design decision to be made, consider a hypothetical case relative to Figure 17.2. Suppose the wetland is to achieve a 90% reduction. It is possible to consider a one-cell wetland, with a presumptive $P = 3$ that needs 84 $m^2/(m^3/d)$. Or, the designer can opt for two cells in a series, each with a presumptive $P = 3$ that needs 68 $m^2/(m^3/d)$. The question might be resolved based on the economics, i.e., does the cost savings of 20% area reduction outweigh the added cost of the divider berm and structures? This illustration carries a zero background concentration, but the concepts are applicable to any pollutant, provided the reduction fraction to background is used instead of percentage removal.

It should be remembered that the designer can control the $N$-value for the wetland (inert tracer tanks in series) but cannot entirely control the $P$-value. Pollutants that are mixtures, which may undergo weathering in the wetland, act to reduce the applicable $P$-value. Table 6.3 provides some guidance on the apparent $P$-values to be selected relative to tracer $N$-values.

**TABLE 17.3**
**Summary of Background Concentrations for FWS Wetlands**

| Parameter | Lightly Loaded | Heavily Loaded |
|---|---|---|
| $BOD_5$ (mg/L) | 2 | 10 |
| TSS (mg/L) | 2 | 15 |
| Organic N (mg/L) | 1 | 3 |
| Ammonia N (mg/L) | <0.1 | <0.1 |
| Oxidized N (mg/L) | <0.1 | <0.1 |
| Total phosphorus (mg/L) | <0.01 | 0.04 |
| Fecal coliforms (CFU/100 mL) | 10–50 | 100–500 |

*Note:* In the case of fecal coliforms, lightly/heavily loaded refers to animal use.

### Rate Coefficients

In this initial, annualized analysis, the appropriate rate coefficients are those (shown in the chapters of Part I) as the result of the fitting of annual data from existing wetlands. Those are variable across wetlands and years, thus producing frequency distributions of the fitted $k$-values. These are to be found as follows:

- $BOD_5$ — Chapter 8, Table 8.2
- Organic N — Chapter 9, Table 9.11
- Ammonia N — Chapter 9, Tables 9.17 and 9.20
- Total Kjeldahl nitrogen (TKN) — Chapter 9, Table 9.12
- Oxidized N — Chapter 9, Table 9.23
- Total N — Chapter 9, Table 9.14
- Total P — Chapter 10, Table 10.11
- Fecal coliforms — Chapter 12, Table 12.3

Conspicuously absent from this list is the common constituent TSS. Some individual system data were presented in

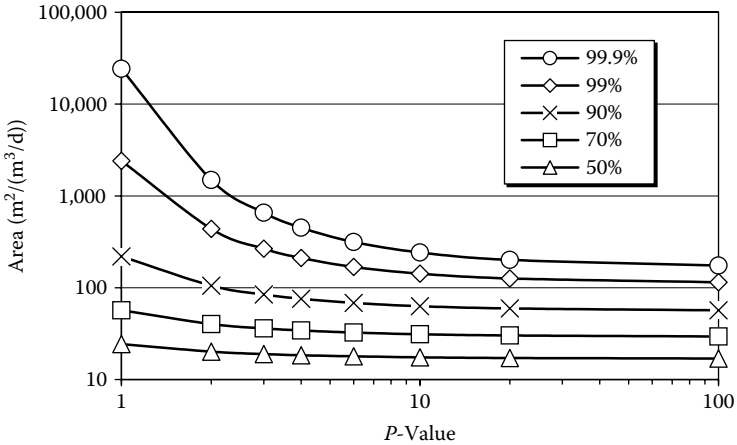



**FIGURE 17.2** The effect of the number of tanks on the area required for different degrees of removal. The calculations are for $k = 15$ m/yr and $C^* = 0$.

Chapter 7 and analyzed for rate coefficients. However, incoming TSS is often reduced rather quickly, and wetland-effluent TSS results from a balance of generation and resuspension in the FWS wetland. The removal rate coefficients that pertain to the inlet region of the wetland are often quite high, ≈10 m/d (3,650 m/yr), as shown in Figure 7.8. Colloidal materials are an exception to this generality. The design recommendation suggested here is the use of a high-rate coefficient for TSS (perhaps 200 m/yr), unless the design-limiting bottleneck happens to be TSS. In that event, it is strongly suggested that settling tests be conducted on the candidate wetland influent waters.

The frequency distributions of the reference tables are generally quite broad. It is up to the designer to narrow the selection for a given application, either by choosing the preferred degree of risk (blind to modifying factors), or by delving further into the details of existing wetland data sets, and to narrow the selection to wetlands closest to the intended application in terms of operating conditions. Regrettably, there is no modern, published, all-inclusive database to which to turn. The reader is cautioned that older databases such as the NADB (Knight *et al.*, 1993) and the 1994 Danish database (in Kadlec and Knight, 1996) have been superseded. Also, old databases are uneven in the quality and quantity of the data presented. There are numerous examples of misinterpretation in such databases, and it is concluded that their use as a sole source of narrowing the information field is dangerous. Because there are now so many treatment wetlands, it is not feasible to provide detailed data. In this book, the compromise is to provide analysis, references, and the generic list of sources used herein (see Appendix A).

## Design Sizing Goals: Load Reduction versus Concentration Reduction

A key feature of treatment wetlands is the ability to design or manage the system for either concentration reduction or for mass removal, but only one at the expense of the other (Trepel and Palmeri, 2002). Wetland performance, as modeled previously, follows the rule of mass action: the removal rate of a pollutant is greater at higher pollutant concentrations in the water. The first-order model assumes a (nearly) direct proportionality: doubling the concentration doubles the removal rate. As a result of this observed behavior, removal rates decrease as water passes through the treatment wetland, and pollutant concentrations are reduced (Figure 17.3). However, the actual mass of the pollutant that is removed increases with increasing hydraulic loading. Thus, increasing hydraulic loads result in more kilograms removed, but at the expense of higher effluent concentrations. This trade-off between removal efficiency and load reduction is a key feature of wetland design for nutrient control. These may be quantified via the first-order model. A simple version for $C^* = 0$ and no water loss is

$$\text{Concentration reduction} = \left(\frac{C_i - C_o}{C_i}\right) = 1 - \left(1 + \frac{k}{Pq}\right)^{-P} \tag{17.15}$$

$$\text{Load reduction} = \text{Input load} \times \text{Concentration reduction}$$

$$= qC_i \cdot \left(1 - \left(1 + \frac{k}{Pq}\right)^{-P}\right) \tag{17.16}$$

The maximum mass of pollutant that can be removed in a given footprint area results from a hydraulic load so high that little or no concentration reduction is achieved. Under these conditions, pollutant concentration is at a maximum everywhere in the wetland, thus maximizing the mass removal rate.

If the output load of a pollutant is to be held below some regulatory limit, then that has the same general effect as specifying an outlet concentration. However, because the load of a pollutant in the wetland outflow is usually taken to

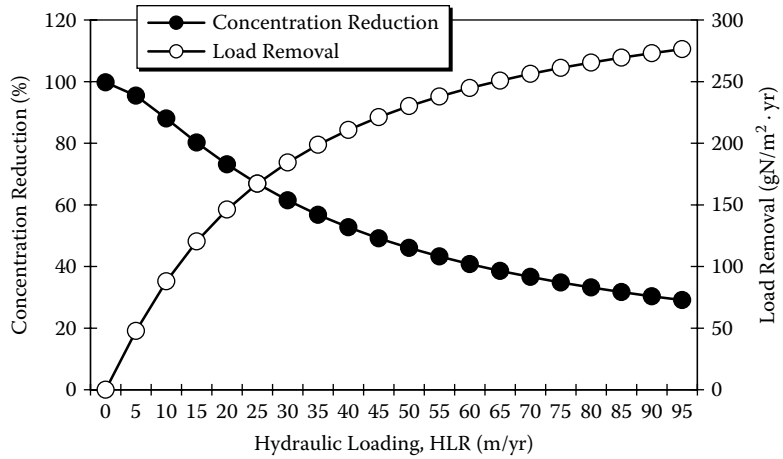



**FIGURE 17.3** Concentration reduction and load reduction as a function of hydraulic load for a hypothetical nitrate treatment wetland. Parameters: $C_i = 10$ mg/L, $k = 35$ m/yr, $C^* = 0$, $P = 4$.

be that in the surface water discharge, any reduction in flow through the wetland assists in providing lower output loads as well as higher load reductions.

All three of the potential design goals (concentration out, load out, load reduction) may be sensible, depending on the pollutant in question. These are marked as shaded cells in the example in Table 17.2. For instance, a low required outlet concentration can be of use in preventing ammonia toxicity in receiving waters. But if there is a load allocation, as might be the case for a discharge contributing to the total maximum daily load (TMDL) for an impaired water body, then the mass of the contaminant is of more direct interest. Lastly, if there is a desire to maximize the benefits of a given wetland footprint, then the goal should be to dissipate or retain the maximum amount of pollutant in the wetland.

Because it is easy to confuse these potentially conflicting goals, it is recommended that the designer clearly identify and state the purpose of the design.

## 17.2 AREA COMPUTATIONS

At this point in the development of constructed wetland technology, it would be disingenuous to provide overly simplistic design-calculation procedures. But it is also a mistake to think that adding more factors that may play a significant role in performance lends more accuracy or precision to design predictions.

### Goal Seeking: Determination of the Required Wetland Area

The fundamental and straightforward technique for determining the area is to adjust that area until the specified criterion is met. That is easy if the calculations have been set up on a spreadsheet; the area can then be sequentially and manually changed by the user until the criterion is met, or automatic searches may be invoked, such as the Solver™ routine in Microsoft Excel™. The hypothetical phosphorus example is continued to illustrate this process.

### Concentration Criterion

A common criterion for phosphorus in the United States is for monthly means to be less than 1.0 mg/L. Realistically, the project owner would not want to encounter exceedances very often. For the example, choose the compliance frequency to be 90%. From Table 10.13, the multiplier to contain exceedance at that frequency is 1.94. Therefore, the design target is adjusted downward to 1.00/1.94 = 0.52 mg/L, which becomes the value to be achieved as the wetland area is varied.

In just three manual iterations, the starting guess of 24 ha ($C_o = 0.62$ mg/L, Table 17.2) is changed to 27.6 ha ($C_o = 0.52$ mg/L). The same result may be obtained using Solver™, which provides the answer in an essentially instantaneous search from any plausible starting condition.

The exceedance containment multipliers for common constituents may be found as follows:

- $BOD_5$ Chapter 8, Table 8.6
- Organic N Chapter 9, Table 9.11
- Ammonia N Chapter 9, Table 9.21
- Oxidized N Chapter 9, Table 9.25
- Total N Chapter 9, Table 9.16
- Total P Chapter 10, Table 10.13
- Fecal coliforms Chapter 12, Table 12.4
- (TKN is not in this list because it is not normally regulated.)

### Maximum Load Criterion

The same philosophy of regulation might lead to an annual load limitation on the effluent from the system. An outlet concentration of 1.0 mg/L at a flow rate of 5,000 m$^3$/d (inflow rate) implies a maximum annual load of 1,825 kgP/yr leaving the wetland (50% load reduction). A search for the wetland area leads to a value of 10.1 ha. This is very much lower than that for the concentration goal for two reasons. First, the exceedance factor is not in play because of the annual character

of the load limit. Secondly, credit is built into the calculation for the loss of water to *ET* and for wetland infiltration.

### Minimum Load Reduction Criterion

A slightly different regulatory philosophy requires a minimum load reduction. For instance, some wetland systems in South Florida require a minimum of 75% phosphorus load reduction. A search for the wetland area leads to a value of 19.8 ha.

### Multiple Compounds of Concern

The projected outlet concentrations of all constituents of interest are calculated via the preceding Equation 17.7 (or Equations 17.12–17.14 in the case of nitrogen). All computed outgoing concentrations and loads vary with the selected wetland area. Area is adjusted until the most stringent criterion is met. Criteria often include outlet concentration specifications, each with an associated allowable frequency of exceedance. But in other cases, outlet pollutant loads may be specified (or load reduction). Loads are easily calculated using the outlet concentrations and flows from the last unit. If the contaminants are considered singly, then a last step remains: all performances are calculated on the controlling (maximum required) wetland area.

As an example of multiple contaminants, the previous phosphorus illustration, along with the additional requirements to also reduce BOD and total nitrogen, is continued. BOD enters the wetland at 30 mg/L, and total nitrogen, at 30 mg/L. The value of $C^*$ is chosen to be 2 mg/L for BOD, and $P = 1$. The value of $C^*$ is chosen to be 1.5 mg/L for TN, and $P = 3$. The rate coefficients are selected to be the medians shown for BOD and TN in FWS systems in Tables 8.2 and 9.14, i.e., $k_{BOD} = 33$ m/yr and $k_{TN} = 13$ m/yr. It is assumed that exceedances must be contained at the 90th percentile. The multipliers are 1.56 for $BOD_5$ (Table 8.6) and 1.55 for TN (Table 9.16).

The calculations of the required areas for each of the three contaminants are shown in Table 17.4. The largest area is needed for the reduction of total nitrogen—40 ha for the 5,000-$m^3$/d flow. When that area is applied to the calculation of performance for $BOD_5$ and TP, those pollutants are reduced more than necessary (Table 17.5).

**TABLE 17.4**
**Areas Needed for a Three-Parameter Wetland Sizing**

| | | $BOD_5$ | TN | TP |
|---|---|---|---|---|
| Inlet *C* | mg/L | 30 | 20 | 2.00 |
| Regulatory *C* | mg/L | 10.0 | 5.0 | 1.00 |
| Containment multiplier | mg/L | 1.56 | 1.55 | 1.94 |
| Target *C* | mg/L | 6.41 | 3.23 | 0.52 |
| Background $C^*$ | mg/L | 2.0 | 1.5 | 0.01 |
| *P*TIS | — | 1 | 3 | 3 |
| Annual *k* | m/yr | 33 | 13 | 10.0 |
| **Area** | **ha** | **30.3** | **40.0** | **27.6** |
| Hydraulic loading | cm/d | 1.65 | 1.25 | 1.76 |
| **Outlet *C*** | **mg/L** | **6.41** | **3.23** | **0.52** |

**TABLE 17.5**
**Performance for a Three-Parameter Wetland Sized for TN Reduction**

| | | $BOD_5$ | TN | TP |
|---|---|---|---|---|
| Inlet *C* | mg/L | 30 | 20 | 2.00 |
| Inlet loading | g/$m^2$·yr | 137 | 91 | 9.1 |
| Regulatory *C* | mg/L | 10.0 | 5.0 | 1.00 |
| Containment multiplier | mg/L | 1.56 | 1.55 | 2.00 |
| **Target *C*** | **mg/L** | **6.41** | **3.23** | **0.50** |
| Background $C^*$ | mg/L | 2.0 | 1.5 | 0.01 |
| *P*TIS | — | 1 | 3 | 3 |
| Annual *k* | m/yr | 33 | 13 | 10.0 |
| **Area** | **ha** | **40.0** | **40.0** | **40.0** |
| Hydraulic loading | cm/d | 1.25 | 1.25 | 1.25 |
| **Outlet *C*** | **mg/L** | **5.48** | **3.23** | **0.29** |

This concludes the preliminary calculation of the required wetland area. It may seem that there are no further steps needed, but in fact there is no guarantee that these preliminary calculations fit into reasonable, known patterns of wetland behavior. It is critical that the bounds of plausible biogeochemical cycles are not exceeded, that the required ancillary chemicals (oxygen, carbon) are present in ample supply, and that the forecasted results are reasonable compared to known performance data. Further, the seasonality of the system has yet to be investigated via forecasting.

## 17.3 CHECKING THE BIOGEOCHEMICAL CYCLES

In this phase of design, vegetation in the prospective wetland and its role in nutrient processing is brought into the picture. The rate coefficient-based analysis has provided first estimates of the quantities of materials entering and leaving the system, but there has been no allocation of the removals to the various processes that comprise the entire ecosystem function. It is not feasible to go too far in breaking down processes because the knowledge base does not support great detail. Here, the processing of carbon, nitrogen, and phosphorus are examined via the mass balances used in ecosystem analysis in Part I (see Chapter 9, Figure 9.14).

In the analysis that follows, two points are important. First, the analysis is based on estimates of what the ecosystem will be like. Such estimates cannot be precise, and, consequently, the analysis is "order-of-magnitude" only. The intent is to gain some insight into the relative importance of wetland processes that will likely be operating if the project is built. Second, the analysis is for an annual period, and hence there remains the issue of seasonality. Seasonal analysis proceeds more readily if we can first establish whether the wetland should be viewed as an agronomic system or a microbial system.

## C, N, and P Cycles

The pollutant removals simplistically described in the preceding section take no account of wetland functions; they are based on purely empirical *k*-values and outlet concentration data. It is therefore prudent to examine the empirical forecasts and to ascertain whether they comply with an estimated set of wetland processes. Of particular interest are the carbon, nitrogen, and phosphorus cycles.

The carbon cycle involves the growth, death, and decomposition of biological materials, including animals, plants, algae, and microbes. For many wetlands, the standing crop of these organic materials is relatively constant throughout the year, but the proportions of living and dead may vary considerably, as may the physical location (i.e., standing dead or litter). Further, the speed of cycling varies with the nature of the material and the time of the year (see Chapter 3). Fine detritus from microbes cycles rapidly, whereas the decomposition of some woody plant parts may take years. However, the important feature of the carbon cycle is the amount of material that does not decompose, for it is this residual that accretes in the ecosystem and forms storage for many pollutants, including phosphorus. This storage is relatively permanent under appropriate hydroperiod conditions.

An approximate assessment of the implied impacts of the carbon cycle can be made on the basis of two numerical characteristics: the speed of the cycle in grams of dry material per square meter per year, and the fraction of the cycled material that does not decompose. The speed of the cycle may also be characterized by the standing crop, in $g/m^2$, plus a turnover rate, in number of times per year. The recycled organic fraction contains carbon that is a source of support for denitrification and other heterotrophic processes. The burial fraction leads to sediment buildup and storage of nitrogen, phosphorus, and other trace contaminants that partition to organics. Typical values for the vegetative standing crop, turnover time, speed, and burial fraction are given in Table 17.6.

In general, the algal and microbial standing phytomass is small compared to the vegetative phytomass. However, turnover times are also small, so the annualized rate of uptake and burial for this component of the phytomass may be estimated to be 50–100% of the vegetative annual rates. Simply stated, the amounts of N and P processed by the big green biomass are not very much larger than those processed by the nearly invisible microbes and algae. This nondisparity has been characterized as the "buckets and teacups" analogy set forth by Richardson *et al.* (1986).

The term nutrient poor would reflect very low nutrient status, with TP < 20 µg/L and $NH_4$-N 0.2 mg/L. Nutrient-moderate wetlands would have TP < 200 µg/L, with $NH_4$-N < 1.0 mg/L. Nutrient-rich wetlands would have TP ≈ 1.0 mg/L, with $NH_4$-N ≈ 5.0 mg/L. Very-nutrient-rich systems would have TP > 5.0 mg/L, with $NH_4$-N > 10.0 mg/L. These definitions do not correspond to the equivalents for aquatic water bodies, where the dominant vegetation is plankton. In the treatment wetland context, bacterial and algal materials are of comparable importance with above- and belowground macrophytic vegetation.

The magnitude of the biogeochemical cycle increases with increasing nutrient availability—up to some presumptive limit enforced by availability of space and sunlight. This fertilizer response is not well-quantified for treatment wetlands and, therefore, cannot be used directly in design. However, rough estimates are of value in assessing the potential importance of the biogeochemical cycle—particularly to check that the empirical design calculations do not imply unreasonable ecosystem functions.

One technique for making such checks is to graphically link a presumed cycle to the mass balance calculations available from the preliminary sizing step. The biogeochemistry check, thus, is implemented via linked *ecosystem* mass balances, one each for C, N, and P. These do not replace, nor can they substitute, *the water column mass balances* used in *k*-rate removal calculations. In an annualized mass balance storage calculation, the following equations may be used:

$$G = D + B \tag{17.17}$$

$$GX_G = DX_D + BX_B \tag{17.18}$$

$$BX_B = \beta \cdot GX_G \tag{17.19}$$

where

$B$ = biomass burial, $g/m^2 \cdot d$
$D$ = biomass decomposition, $g/m^2 \cdot d$
$G$ = biomass growth, $g/m^2 \cdot d$
$X$ = fraction of the pollutant in each biomass compartment
$\beta$ = fraction of pollutant uptake that is stored

### The Carbon Cycle

The nomenclature for biomass processes in treatment wetlands is a bit confusing. The growth, death, and decomposition processes are referred to as part of the wetland carbon cycle, but more than carbon is involved. However, most vegetation and other wetland organisms are about 40% carbon; so, either dry biomass or carbon serves to track the amount of the material involved. Carbon itself is withdrawn from atmospheric sources as carbon dioxide for photosynthesis. Likewise, it is returned to the atmosphere as methane from anaerobic mechanisms, or carbon dioxide from oxidative processes (respiration included).

The ability to estimate nutrient cycling rests upon our knowledge of the biomass pools in the wetlands and their changes. Guidelines are shown in Table 17.6. According to Equations 17.17–17.19, on an annual basis, the important estimation quantities are

*Annual growth rate, $g/m^2 \cdot yr$* (standing crop phytomass = necromass + biomass times turnover per year)
*Annual burial fraction* (undecomposable residual fraction)

**TABLE 17.6**
**Estimates of Wetland Nutrient Cycling in Biomass**

| | | Nutrient Poor | Nutrient Moderate | Nutrient Rich | Very Nutrient Rich |
|---|---|---|---|---|---|
| **Organic matter** | | | | | |
| Standing crop | gDW/m$^2$ | 150 | 500 | 3,000 | 10,000 |
| Turnover time | times/year | 2 | 4 | 3 | 2 |
| Cycling rate ($G$) | gDW/m$^2$·yr | 300 | 2,000 | 9,000 | 20,000 |
| Burial fraction ($b$) | % | 2 | 10 | 20 | 25 |
| Accretion ($B$) | gDW/m$^2$·yr | 6 | 200 | 1,800 | 5,000 |
| **Phosphorus** | | | | | |
| Tissue content | mgP/kg | 1,000 | 2,000 | 3,000 | 4,000 |
| Uptake | gP/m$^2$·yr | 0.30 | 4.0 | 27.0 | 80 |
| Return | gP/m$^2$·yr | 0.29 | 3.6 | 21.6 | 60 |
| Accretion | gP/m$^2$·yr | 0.01 | 0.4 | 5.4 | 20 |
| **Nitrogen** | | | | | |
| Tissue content | %dw | 1.0 | 1.5 | 2.0 | 2.5 |
| Uptake | gN/m$^2$·yr | 3.00 | 30.0 | 180 | 500 |
| Return | gN/m$^2$·yr | 2.94 | 27.0 | 144 | 375 |
| Accretion | gN/m$^2$·yr | 0.06 | 3.0 | 36 | 125 |

*Note:* Bacterial and algal cycling are included in these values.

*Source:* For first two categories, data from Davis (1994) In *Everglades: The Ecosystem and Its Restoration*. Davis and Ogden (Eds.), St. Lucie Press, Delray Beach, Florida, pp. 357–378. For the second two, data from Kadlec (1997a) *Ecological Engineering* 8(2): 145–172.

An assumption is made that the nutrients taken up, but not buried as accretion, are returned to the water column of the FWS wetland. For nitrogen, this is the maximum estimate, as microbial processes in abovewater tissues can transfer nitrogen to the atmosphere without entering the water. These phytomass quantities, together with phytomass nutrient content (percentage or mg/kg), allow checks on the empirical removal calculations.

For purposes of design, for order-of-magnitude checks on the calculated nutrient removals, the total growth rate and burial fraction are assumed based on the strength of the water to be treated. This provides a rough annual estimate of the biomass cycle (Figure 17.4). Note that the magnitude of this cycle and the nutrient contents are a function of the degree of fertilization of the wetland (see Chapter 3).

The wetland carbon cycle is also critical to observe performance as it relates to sediment oxygen demand and to the carbon supply for denitrification. The implied supply constraints of this carbon cycle are examined in the constraint check section of this chapter.

## The Phosphorus Cycle

For phosphorus, the calculated removal is represented as a large uptake ($GX_G$)—in major part balanced by the return of soluble phosphorus from tissue decomposition ($DX_D$). For

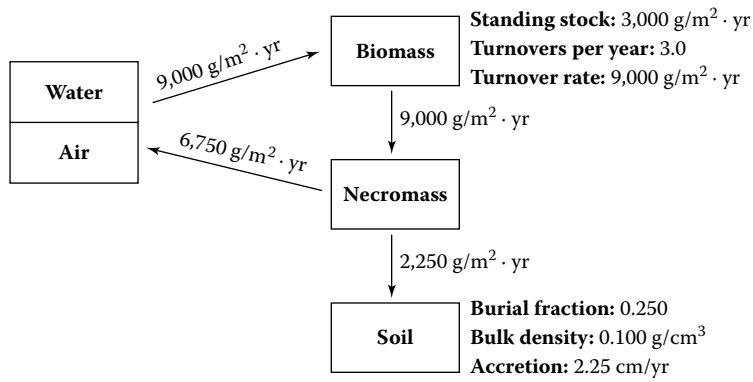



**FIGURE 17.4** Estimated annual biomass cycle in a FWS treatment wetland for a rich nutrient condition. Note that the standing stock and turnover refers to above- and belowground material, and to macrophytes, algae, invertebrates, and microbes.

a stable ecosystem past startup, the net phosphorus removal associated with the *k*-rate calculation is assigned to accretion. Sorption and the building of additional biomass are no longer sinks for phosphorus (Kadlec, 1997a). Because the *k*-rate calculation is independent of the biomass cycle calculation, there is one degree of freedom, which is taken to be the computed accretion fraction. It is known through extensive data from treatment wetlands that this percentage ranges from ≈500 to 5,000 mgP/dry kg, or from 0.05–0.5% dry weight. A somewhat narrower range is shown in Table 17.6 because extremely nutrient-poor wetlands are uncommon in treatment applications. It is unlikely that higher removals can be sustained via accretion. The corresponding accretion rates are 0.01–20 gP/m$^2$·yr. The values in Table 17.6 are provided as an order-of-magnitude check point. If a design is forecast to exceed these approximate cycling, removal, and storage values, then more detailed mass balances should be invoked. If it appears that the phosphorus removal calculated by a *k*-rate exceeds the ecosystem capacity to store, then the *k*-value should be adjusted downward.

As a graphic illustration, the design example is carried forward (Figure 17.5). Because the nutrient concentrations are high (2 mg/L P and 20 mg/L TN), it is anticipated that the nutrient-rich condition will prevail, as illustrated in Figure 17.4. The biogeochemical cycle removes 27 gP/m$^2$·yr from the water column (9,000 gdw/m$^2$·yr × 0.3% P), far more than the loading to the wetland of 9.1 gP/m$^2$·yr. The *k*-rate calculations indicate that 7.4 gP/m$^2$·yr are removed, and, hence, it is deduced that (27 − 7.4) = 19.6 gP/m$^2$·yr are returned to the water from decomposition and leaching of the biomass. The removal to accretion is thus 7.4/27 = 28% of the biomass uptake. These uptakes and returns involve aboveground plant parts (≈50%), belowground plant parts (≈15%), and microbes and algae (≈35%). The 28% burial rate for phosphorus is somewhat higher than the comparison value in Table 17.6 of 20%. If phosphorus were the controlling substance for sizing, it would be advisable to adjust the rate coefficient downward from the value of 10 m/yr used in the design forecast. But, the wetland is overdesigned for phosphorus reduction, and, hence, it is probably not necessary to revise the *k*-rate calculation.

## The Nitrogen Cycle

For nitrogen, a more complex situation occurs as a result of the multiple speciation of water column nitrogen. The details of nitrogen mass balancing have been presented in Part I, Chapter 9. The water column contains organic, ammonium, and nitrate nitrogen. Their interconversions are computed from the empirical *k*-rates. The biogeochemical cycle is linked in a manner analogous to phosphorus, but with abstraction from both the ammonium and nitrate pools in the water, and return from both the ammonium and organic pools in the water. This allocation recognizes a split of plant uptake between ammonium and nitrate (Martin and Reddy, 1997) and the fact that decomposition processes produce organic nitrogen. The nitrogen content in accreting sediments is known from extensive data from treatment wetlands to range from ≈1.0–2.5% dry weight (Table 17.6). A lower value would be associated with nutrient-poor wetlands, a higher with nutrient-rich systems. Again, because the *k*-rate calculations are independent of the cycle calculation, there is one degree of freedom, which for nitrogen is taken to be an assumed percentage of the cycled nitrogen that is buried in accreting sediments. A nitrogen deficit may be assumed to be supplied by fixation, a process known to occur in nitrogen-deficient wetland environments.

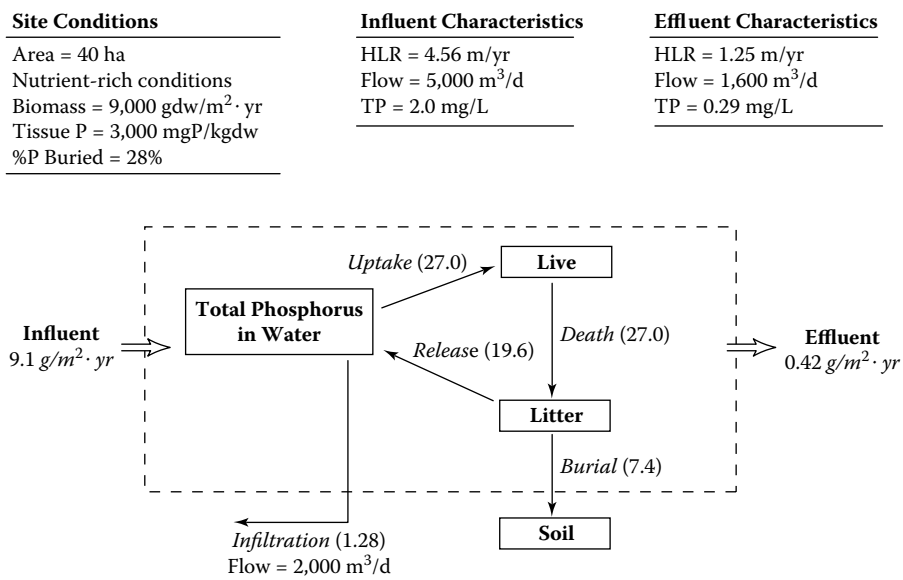



**FIGURE 17.5** Estimated annual phosphorus cycle in a FWS treatment wetland for a nutrient-rich condition. Flows, concentrations, and loadings leaving the wetland are from *k*-rate forecasts. The cycle turnover and tissue-phosphorus concentrations are presumptive. The fraction of phosphorus buried as accretion is calculated by mass balance.

| Site Conditions | Influent Characteristics | Effluent Characteristics |
|---|---|---|
| Area = 40 ha | HLR = 4.56 m/yr | HLR = 3.29 m/yr |
| Nutrient-rich conditions | Org N = 9.0 mg/L | Org N = 1.84 mg/L |
| Biomass = 9,000 gdw/$m^2 \cdot$ yr | $NH_4$–N = 10.0 mg/L | $NH_4$–N = 0.15 mg/L |
| Tissue N = 2% dw | $NO_x$–N = 1.0 mg/L | $NO_x$–N = 1.37 mg/L |
| Fraction $NH_4$–N Uptake = 0.81 | TN = 20.0 mg/L | TN = 3.23 mg/L |
| N Buried = 25% | TKN = 19.0 mg/L | TKN = 1.87 mg/L |

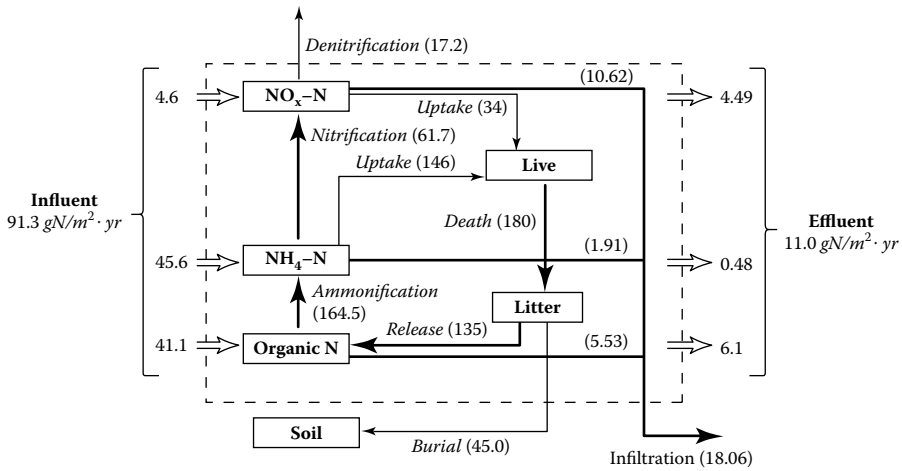



**FIGURE 17.6** Estimated annual nitrogen cycle in a FWS treatment wetland for a nutrient-rich condition. Flows, concentrations, and loadings leaving the wetland are from *k*-rate forecasts. The cycle turnover and tissue–N concentrations are presumptive. The fraction of biomass nitrogen buried as accretion is assumed to be 25%.

An illustration of the use of the nitrogen cycle and interconversions to confirm the design *k*-rate calculations is shown in Figure 17.6. The previous illustrative example is continued, but for simplicity, the details of the *k*-rate calculations for the various nitrogen species are not shown here. As indicated in Figure 17.6, the incoming nitrogen is presumed to be mostly organic (9 mg/L) and ammonia (10 mg/L), comprising nearly all of the 20 mg/L of TN. The nitrogen loading to the wetland is 91 gN/$m^2$·yr, which is a low loading that places the system in the category of an agronomic system (see Chapter 9). It is therefore expected that the biogeochemical cycling of nitrogen will play an important role in the overall reduction, but not to the exclusion of microbial processes.

Once the size of the biomass cycle has been selected, together with the phytomass nutrient content, mass balances fix all the nitrogen fluxes for the specified inflows and outflows. For this proposed design example, the cycling of biomass nitrogen is very important. The required nitrogen to build the annually cycled biomass is 180 gN/$m^2$·yr, which is just about double the amount of nitrogen supplied in the wastewater (Figure 17.6). However, there is no concern that the plants will starve, because there is a correspondingly large return flux of nitrogen from leaching and decomposition of necromass. It is again necessary to keep in mind that these uptakes and returns involve aboveground and belowground plant parts as well as microbes and algae.

The ultimate fate of removed nitrogen in the example is apportioned to accretion (56%), denitrification (21%), and seepage (23%). Interestingly, this does not mean that there are only small amounts of nitrification occurring, because about two thirds of the incoming nitrogen load winds up being nitrified (61.7 out of 91.3 gN/$m^2$ yr, Figure 17.6) because of leakage and a small amount of biomass uptake. The question then arises as to whether there is sufficient oxygen supply to support the nitrification implied by this mass balance. This implied supply is evaluated in the next section.

## 17.4 CHEMICAL SUPPLY CONSTRAINTS

Traditional chemistry assumptions indicate a requirement for carbon to support heterotrophic denitrification and oxygen to support nitrification.

Table 17.7 lists some of the more important chemical and biological constraints on wetland processes. The top block of information shows that the carbon balance, and, hence, the implied biochemical oxygen demand (BOD), is due to removal of incoming BOD and to the carbon formed by the biogeochemical cycle. Denitrification requires 4.0 g of chemical oxygen demand (COD) to reduce 1 g of nitrate nitrogen (U.S. EPA, 1993b; Crites and Tchobanoglous, 1998). Decomposing biomass is far more important than added BOD in the water in many instances. However, some moderate fraction of the biomass decomposition takes place in air, thus reducing the available carbon for denitrification.

The second block concerns nitrification, for which there are associated oxygen and alkalinity requirements. An oxygen equivalent is retrieved as the result of denitrification (third block, Table 17.7), as well as some of the required

**TABLE 17.7**
**Supply and Demand Constraints**

| Process | Units | Basis |
|---|---|---|
| BOD removed | $gO/m^2 \cdot d$ | *k*-rate load reduction |
| Nitrate-N equivalent of BOD removed | $gN/m^2 \cdot d$ | 0.25 Mass ratio to COD (BOD) |
| SOD equivalent of decomposing biomass | $gO/m^2 \cdot d$ | 0.1–0.5 Mass ratio |
| Max oxygen required for BOD and SOD | $gO/m^2 \cdot d$ | Sum of BOD and SOD |
| Reaeration | $gO/m^2 \cdot d$ | 0–4 |
| Oxygen required for traditional nitrification | $gO/m^2 \cdot d$ | 4.57 Mass ratio to N |
| Oxygen required for anammox | $gO/m^2 \cdot d$ | 1.72 Mass ratio to N |
| Alkalinity required for nitrification | $g/m^2 \cdot d$ | 7 Mass ratio to N |
| Oxygen from denitrification | $gO/m^2 \cdot d$ | 2.86 Mass ratio to N |
| Carbon required for denitrification | $gC/m^2 \cdot d$ | 1.1 Mass ratio to N |
| Alkalinity produced by denitrification | $g/m^2 \cdot d$ | 3 Mass ratio to N |
| Biomass produced | $g/m^2 \cdot d$ | 2–50 |
| Biomass accreted | $g/m^2 \cdot d$ | 2–25% of production |
| Usable carbon generated | $gC/m^2 \cdot d$ | 10% of biomass |
| Carbon accreted | $gC/m^2 \cdot d$ | 2–25% of production |
| Carbon burned | $gC/m^2 \cdot d$ | 75–98% of production |
| Sediment generated | cm/yr | 0.05–0.2 bulk density |
| Sulfur for denitrification | $gS/m^2 \cdot d$ | 1.69 Mass ratio to N |

alkalinity for nitrification. The carbon produced in the wetland supplies some or all that is necessary for denitrification (fourth block, Table 17.7). The accretion process implies a rate of sediment buildup determined from the sediment bulk density. Finally, the possibility of sulfur-driven denitrification carries a demand for sulfide-sulfur.

These constraints are critical to design because wetland chemical processes cannot proceed without the necessary ancillary compounds. Nowhere is this more apparent than in the need for oxygen to drive ammonia removal routes. Therefore, if inadequate supplies of ancillary chemicals are forecast, rate constants and loadings must be reduced until the constraints are met. Alternatively, additional supplies may be introduced into the wetland. This may require costly supplements, such as the addition of methanol to fuel denitrification (Gersberg *et al*., 1984). However, sometimes rearrangements of flows can resolve the supply problem, such as the feedforward of high BOD unpretreated influent for a carbon supplement for denitrification (Burgoon, 2001).

## Oxygen Supply

The various oxygen requirements for the example design are calculated (estimated) in Table 17.8. Removal of BOD (0.35 $gO/m^2 \cdot d$) and nitrification (0.73 $gO/m^2 \cdot d$) appears to exert only a small demand, which should easily be supplied by atmospheric reaeration. The decomposition of the generated necromass would consume something like 10 $gO/m^2 \cdot d$, but that decomposition is in part due to anaerobic processes and, in part, occurs above the water as standing dead materials oxidize. Another part of oxidation may involve root respiration, which comprises plant oxygen transfer. Therefore, it is difficult to attach great significance to the apparent need for large amounts of oxygen for necromass oxidation.

## Carbon Supply

The carbon produced by biomass in the example wetland (4.93 $gC/m^2 \cdot d$) far exceeds that necessary for denitrification (0.05 $gC/m^2 \cdot d$), and in fact could be supplied easily by the reduction of BOD (0.13 $gC/m^2 \cdot d$). Thus, despite the fact that only a fraction of the biomass-produced carbon is available in the water, there is no apparent shortage of carbon for denitrification in this example.

Because there is a large phytomass cycle, the accretion of solid residuals is an important factor. The implied rate of sediment buildup is 2.3 cm/yr, which is comparable to the rates observed in operating wetlands (Kadlec, 1997a).

## Intersystem Performance Checks

The question addressed next is, how does the proposed design compare with results from existing wetlands for which there is operating data? The annual performance period is retained for this phase of investigation.

The intersystem loading chart for phosphorus (Figure 17.7) shows that the overall scatter follows an increasing trend of outlet concentrations with increasing loading. However, in design, the loading variable is usually restricted to a fixed inlet concentration, whereas the hydraulic loading is variable in response to different choices of the wetland area. Therefore, in any particular design, only a subset of the larger loading group is relevant. For phosphorus and most common pollutants, a particular inlet concentration subset shows a more modest increasing trend than that of the whole set.

A generic loading chart looks similar to the conceptual "cloud" indicated in Figure 17.8. The larger data group encompasses all systems for all inlet concentrations, whereas each subgroup at a specific range of inlet concentrations occupies a smaller group. The left side of either data cloud represents less efficient systems, and hence these are prototypes of low risk for the design. The right side represents more efficient systems, a riskier assumption. Any performance calculation is represented by a single point on this plot and, therefore, may be compared to the intersystem data scatter on this chart.

The loading graphs that allow intersystem comparisons for common constituents may be found as follows:

- TSS — Chapter 7, Figure 7.19
- $BOD_5$ — Chapter 8, Figure 8.9
- Organic N — Chapter 9, Figure 9.17
- Ammonia N — Chapter 9, Figure 9.37
- TKN — Chapter 9, Figure 9.23
- Oxidized N — Chapter 9, Figure 9.51
- Total N — Chapter 9, Figure 9.29
- Total P — Chapter 10, Figure 10.17
- Fecal coliforms — Chapter 12, Figure 12.11

**TABLE 17.8**
**Supply and Demand Constraints for the Example**

| | Annual Flux | | Daily Flux | | Basis | |
|---|---|---|---|---|---|---|
| BOD removed | 129 | gO/m²·yr | 0.35 | gO/m²·d | — | |
| SOD equivalent of decomposed biomass | 3,600 | gO/m²·yr | 9.9 | gO/m²·d | 32/12 | Mass ratio |
| Max oxygen required for BOD and SOD | 3,729 | gO/m²·yr | 10.2 | gO/m²·d | — | |
| Oxygen required for nitrification | 265 | gO/m²·yr | 0.73 | gO/m²·d | — | |
| Alkalinity required for nitrification | 120 | g/m²·yr | 0.33 | g/m²·d | 7 | Mass ratio |
| Oxygen from denitrification | 59 | gO/m²·yr | 0.16 | gO/m²·d | 48/14 | Mass ratio |
| Carbon required for denitrification | 19 | gO/m²·yr | 0.05 | gC/m²·d | 1.1 | Mass ratio |
| Carbon equivalent of BOD removed | 48 | gO/m²·yr | 0.13 | gC/m²·d | 12/32 | Mass ratio |
| Alkalinity produced by denitrification | 51 | g/m²·yr | 0.14 | g/m²·d | 3 | Mass ratio |
| Biomass produced | 9,000 | g/m²·yr | 24.7 | g/m²·d | — | |
| Biomass accreted | 2,250 | g/m²·yr | 6.16 | g/m²·d | 25% | of Production |
| Carbon generated | 1,800 | gC/m²·yr | 4.93 | gC/m²·d | 20% | Content |
| Carbon accreted | 450 | gC/m²·yr | 1.23 | gC/m²·d | 25% | of Production |
| Carbon burned | 1,350 | gC/m²·yr | 3.70 | gC/m²·d | 75% | of Production |
| Sediment generated | 2.3 | cm/yr | — | | 0.1 | g/cc Bulk density |

After the necessary wetland area has been calculated based on the limiting pollutant and the performance for the other contaminants of concern recalculated for that area, the position of the design may be plotted on each loading chart of concern. For the example design that has been carried forward, these are shown in Figures 17.9 (BOD), 17.10 (TN), and 17.11 (TP). It is seen that in all cases, the design point is close to the central tendency of the intersystem data. That is not an accident, because the rate coefficients used in the example were selected to be at the 50th percentile of the intersystem values. In general, there is no design requirement to choose that percentile, and the loading charts can assist in interpretation of the consequences of picking higher- or lower-rate coefficients.

As should be expected, the wetland area is sensitive to the choice of rate coefficients. The two modes of design calculations are the area search and performance forecast that follows once the limiting (largest) area has been found. Figure 17.12 shows the effect of changing rate coefficients for phosphorus during the area-seeking phase of design. Recall that the goal was to achieve an outlet concentration of 0.52 mg/L. For a rate coefficient of 10 m/yr, this required 27.5 ha. Picking a lower rate coefficient (2.8 m/yr) results in a higher area (67 ha) and, thus, in a lower phosphorus loading (5.4 gP/m²·yr) (Figure 17.12). Picking a higher rate coefficient (25 m/yr) results in a lower area (12 ha) and, thus, in a higher phosphorus loading (30.4 gP/m²·yr). It is seen that the effect of changing the estimate of the *k*-value moves the design point from the left to the right on the loading chart. Note that at low *k*-values for phosphorus, the area exceeds that needed for nitrogen reduction, and, thus, the limiting constituent would change from nitrogen to phosphorus.

In the forecast mode, the effect of changing the estimate of the *k*-value moves the design point up and down on

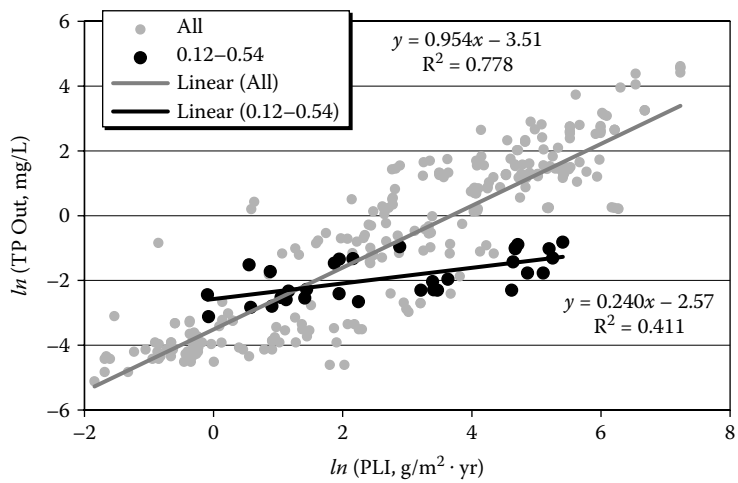



**FIGURE 17.7** The phosphorus loading chart for FWS wetlands replotted from Figure 10.17. Log-linear regressions are shown for the entire data set and for the inlet concentration range 0.12–0.54 mg/L.

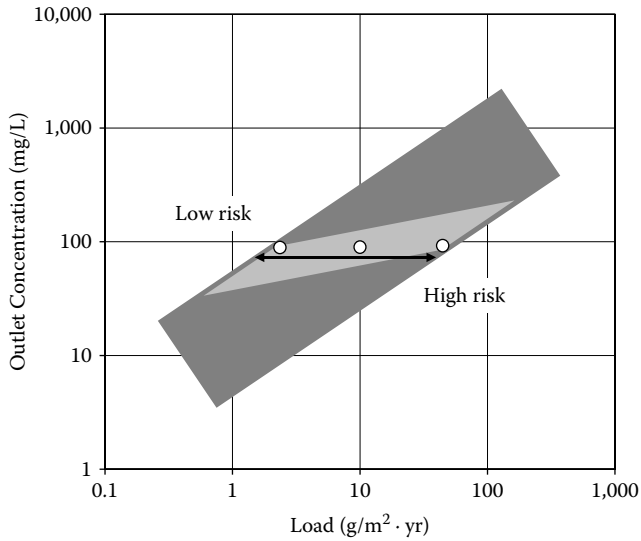



**FIGURE 17.8** Concepts of risk on the pollutant-loading chart. The overall data set of comparison systems occupies a band of increasing concentrations with increasing load. However, the data set for a particular inlet concentration occupies a smaller band, oriented at a lesser slope.

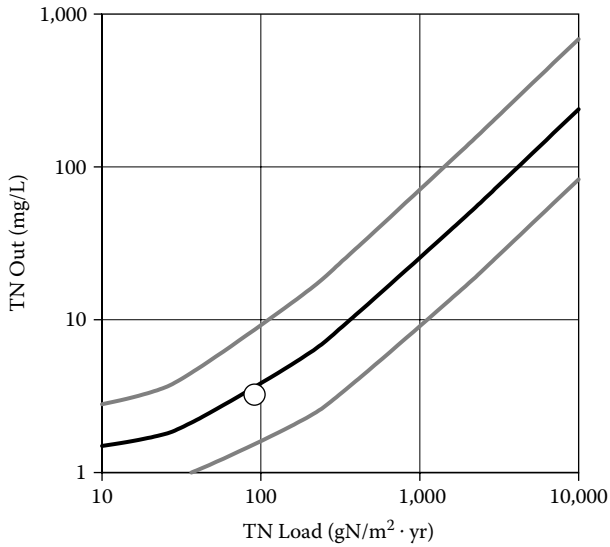



**FIGURE 17.10** Location of the example design on the TN-loading graph. The upper and lower lines are the approximate bounds of data shown in Figure 9.29. The central line represents the central tendency of annual data.

the loading chart. The system area is fixed at that needed for nitrogen reduction, 40 ha in the example. As seen in Figure 17.13, if the phosphorus *k*-value is too low, the outlet phosphorus concentration exceeds the maximum allowable design value of 0.52 mg/L, and, thus, the limiting constituent would change from nitrogen to phosphorus.

Clearly, there are a number of choices open to the designer for selection of wetland performance. The procedure set forth here advocates the use of rate coefficients, complemented by the use of loading charts as a cross-check on intersystem performance.

## 17.5 ADJUSTMENTS FOR SEASONALITY

The methods of analysis, up to this point, have utilized annual average performance measures. However, some wastewater constituents display seasonal removal behavior in FWS treatment wetlands, as has been discussed on a case-by-case basis in Part I. Seasonal patterns are particularly evident for nitrogen species, driven either by temperature effects on microbial processes or seasonal plant growth patterns, or both. Consequently, the magnitudes of seasonal swings in pollutant reduction are

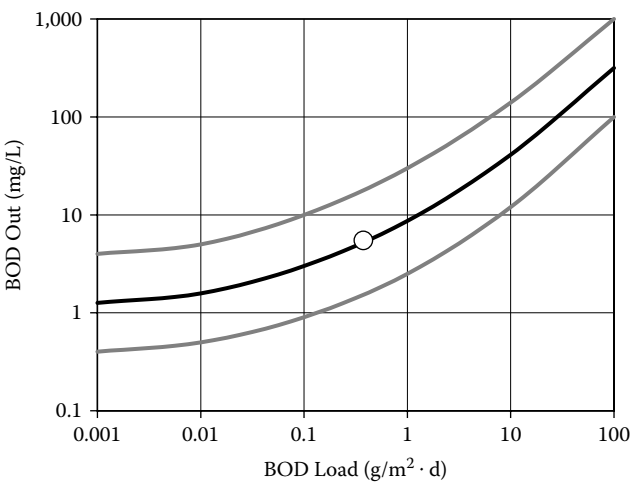



**FIGURE 17.9** Location of the example design on the BOD-loading graph. The upper and lower lines are the approximate bounds of data shown in Figure 8.9. The central line represents the central tendency of annual data.

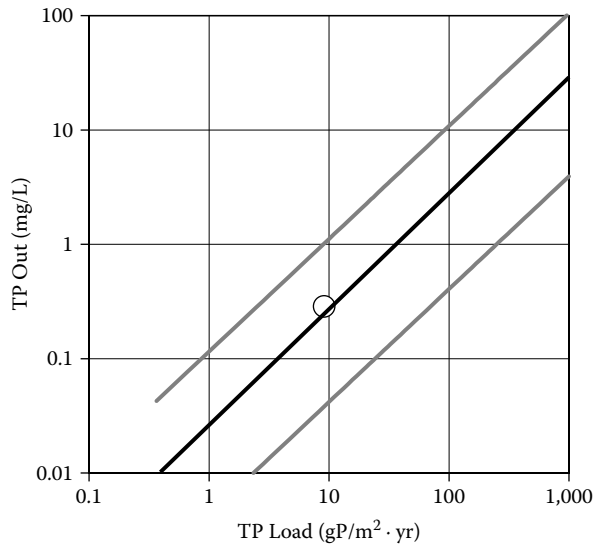



**FIGURE 17.11** Location of the example design on the TP-loading graph. The upper and lower lines are the approximate bounds of data shown in Figure 10.17. The central line represents the central tendency of annual data.

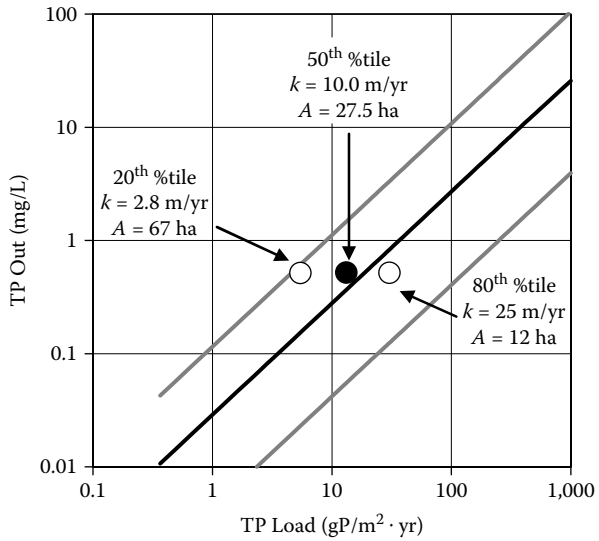



**FIGURE 17.12** Location of the example design on the TP-loading graph. The upper and lower lines are the approximate bounds of data shown in Figure 10.17. The central line represents the central tendency of annual data.

sensitive to the magnitude of the annual temperature cycle or the growth-senescence patterns of a particular climatic region. There are three fundamentally different ways of dealing with these effects: temperature coefficients (theta factors), monthly rate coefficients, and outlet trend amplitudes. If theta factors are appropriate, then the water temperature must be forecast.

### Wetland Water Temperature

A first approximation for long detention wetlands has been shown to be the mean daily air temperature during the unfrozen season (see Part I, Chapter 4). The next level of approximation involves estimating longitudinal profiles of wetland water temperature via the use of accommodation coefficients. The water temperature may depend on wetland size due to lack of time to reach the balance condition. In that case, design iteration may become necessary. The prediction of wetland water temperatures is briefly summarized here.

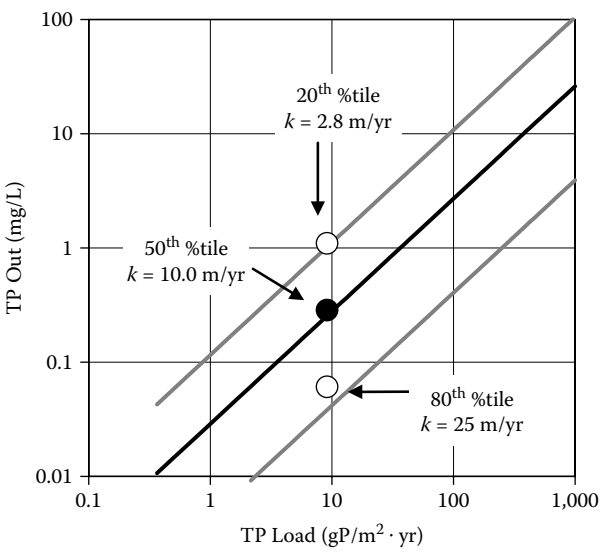



**FIGURE 17.13** Location of the example design on the TP-loading graph. The upper and lower lines are the approximate bounds of data shown in Figure 10.17. The central line represents the central tendency of annual data.

### Balance-Point Temperature

A treatment wetland may contain one or two thermal regions, depending on water loading (detention time). For long detention times, there is an inlet region in which water temperatures adjust to the prevailing meteorological conditions and an outlet region in which that adjustment is complete. The value reached is determined by the balance of energy flows and is termed the *balance temperature*. For short detention times, near the wetland inlet, the adjustment may not be completed. To a rough approximation, wetland water balance temperatures are equal to air temperatures during the unfrozen season. More accurate forecasts are possible utilizing the energy balance, as described in Chapter 4.

### Wetland Water Temperature Cycles

The annual cycle of wetland water temperatures in mild to warm climates follows a sinusoidal pattern, with a summer maximum and a winter minimum. In northern climates, the onset of frozen conditions typically is accompanied by under-ice water temperatures of 1–2°C. The sinusoidal model, truncated for frozen conditions, is as follows:

For the unfrozen season ($t_1 < t < t_2$),

$$T_w = T_{avg}\left(1 + A \cdot \cos\left[\omega(t - t_{max})\right]\right) \tag{4.27}$$

For the frozen season ($t_2 < t < t_1$),

$$T_w = T_o \tag{4.28}$$

where

$A$ = fractional amplitude of the sinusoid, dimensionless
$\omega$ = yearly cycle frequency = $2\pi/365 = 0.0172\ d^{-1}$
$t$ = time, Julian day
$t_1$ = ice-out time, Julian day
$t_2$ = freeze-up time, Julian day
$t_{max}$ = time of annual maximum temperature, Julian day
$T_w$ = water temperature, °C
$T_{avg}$ = annual average water temperature, °C
$T_o$ = under-ice water temperature, °C

Table 4.6 lists the parameters for this sinusoidal model for a variety of FWS wetlands at different latitudes. The deterministic trend represents the central tendency of water temperatures, but there are also stochastic variations. For instance, the standard deviation of the monthly means for Columbia, Missouri, is 1.6°C.

*The Accommodation Zone*

The inlet zone of a treatment wetland exhibits temperature changes as the water approaches the balance temperature. For short detention times (typically less than three days for FWS), the adjustment may not be completed, and the balance temperature is not reached (Kadlec, 2006c).

An empirical exponential model may be easily calibrated and used to describe the approach to the balance temperature:

$$T_w = T_b + (T_{wi} - T_b)\exp\left(-\frac{\eta t}{\rho c_p h}\right) = T_b + (T_{wi} - T_b)\exp\left(-\frac{t}{\tau_A}\right) \tag{4.40}$$

where

$c_p$ = heat capacity of water, $4.182 \times 10^6$ J/kg·°C
$h$ = water depth, m
$T_w$ = wetland water temperature, °C
$T_b$ = wetland balance temperature, °C
$T_{wi}$ = inlet water temperature, °C
$\eta$ = accommodation coefficient, MJ/m²·d·°C
$\rho c_p$ = volumetric heat capacity of water, MJ/m³·°C
$t$ = nominal detention time to an internal point, d

The quantity $\tau_A = \rho c_p h/\eta$ represents characteristic accommodation time for the wetland water on its travel through the system, during which 63.2% of the change from inlet to balance temperature is achieved. At $3\tau_A$, 95.0% of the change is accomplished. Data for free water surface wetlands indicate that adaptation takes a detention time on the order of one to three days (see Table 4.9).

## Temperature Coefficients

Temperature effects on $k$ are usually summarized using the modified Arrhenius equation

$$k_T = k_{20}\theta^{(T-20)} \tag{17.20}$$

where $k_T$ is the rate constant at temperature $T = T$°C and $k_{20}$ is the rate constant at 20°C. Values of the temperature correction factor ($\theta$) have been estimated for data sets with adequate operational temperature data.

Frequency distributions for theta factors and corresponding $k_{20}$ values for common constituents may be found as follows:

- $BOD_5$ Chapter 8, Table 8.4
- Ammonia N Chapter 9, Table 9.29
- TKN Chapter 9, Table 9.16
- Oxidized N Chapter 9, Table 9.40
- Total N Chapter 9, Table 9.20
- Total P Chapter 10, Table 10.12
- Fecal coliforms Chapter 12, Table 12.5
- *Note:* Data are presently not adequate to determine a temperature coefficient for either TSS reduction or ammonification.

## Rate-Constant Adjustment

The goal of design calculation is the selection of the wetland area that will provide the necessary treatment in all seasons of the year. The annualized calculations that have gone before are presumed to represent conditions roughly corresponding to the average annual water temperature. However, it is now necessary to adjust these rate coefficients downward to winter conditions, which range downward to about 2°C.

The controlling pollutant in the example is total nitrogen. It is also shown that this is an agronomic wetland, with considerable influence of the biomass cycle. It is probably not appropriate to use theta-factor corrections under these conditions, but, nonetheless, it shall be done here for illustration purposes. From Table 9.20, the median $k_{20}$ = 21.5 m/yr and the median $\theta$ = 1.056. Suppose it is forecast that the low winter water temperature is going to be 4°C. (This would apply to the midsection of the eastern United States, e.g., Kentucky/Tennessee). The corresponding $k_4$ = 9.0 m/yr. The annual calculation used is $k$ = 13 m/yr, so a larger wetland area is needed, which is found to be 52 ha—up from 40 ha.

Note that if the temperature coefficient is large and the winter temperature is low, e.g., 1–2°C, then the area may become quite large, and winter storage may be warranted.

The median theta factors for some pollutants are very close to, or even less than, 1.000, implying very little temperature dependence. BOD removal in FWS wetlands has a median of $\theta$ = 0.985 (Table 8.4), and fecal coliform reduction has a median of $\theta$ = 0.963 (Table 12.5). Of course, sustainable, postsorption phosphorus removal mechanisms rely heavily on the balance of vegetative uptake over return, and, therefore, agronomic models should apply. That is reflected in median $\theta$ = 1.005 (Table 10.12). It is various nitrogen species that exhibit theta factors, which implies a need for temperature correction in design.

## Monthly Rate Coefficients

As discussed in Chapter 9, temperature coefficients apply to microbial controlled systems but not necessarily to agronomic systems. It would be desirable to have monthly rate coefficients for the various sequential nitrogen conversions under agronomic conditions, but these are generally not available at this time. What is clear is that agronomic wetlands display a peak of ammonia removal during the spring growth period and that this does not correspond to the maximum of water temperature (see Chapter 9, Figure 9.28).

The mean ammonia behavior of 11 cold-climate wetlands under partial or total agronomic control is shown in Table 17.9. Note that the rate coefficient is highest in May, which is typically the maximum growth month. Summer is, in general, a time of greater ammonia reduction, perhaps reflecting microbial activity as well as growth in some instances. The average total nitrogen loading to these wetlands was 130 gN/m²·yr, just over the putative limit of 120 for agronomic wetlands; the ammonia loading was a bit lower at 90 gN/m²·yr. It is noteworthy that January remains the "bottleneck" month, with a rate coefficient that is about

**TABLE 17.9**
**Apparent First-Order Rate Constants for Ammonia Removal in FWS Treatment Systems in Cold Climates**

| Month | Rate Coefficient (m/yr) |
|---|---|
| January | 2.4 |
| February | 4.1 |
| March | 7.1 |
| April | 14.7 |
| May | 20.3 |
| June | 13.1 |
| July | 11.5 |
| August | 12.4 |
| September | 11.7 |
| October | 8.0 |
| November | 5.8 |
| December | 3.0 |
| **Annual Average** | **9.5** |

*Note:* Model parameters are $P = 3$ and $C^* = 0$; data from 11 wetlands; the mean nitrogen loading was 130 gN/m$^2$·yr (N = 11).

four times lower than the annual average and about ten times lower than the growing season maximum.

### Trend Amplitudes

When contaminant concentrations are well above wetland background levels, the seasonal variations in wetland effluent concentrations may be described equally well by a *P-k-C** model or an empirical sinusoidal model. Both employ the three constants with which the data is fit. Under such conditions, either may form the trend basis from which the exceedance frequencies are determined. The absence of a significant temperature coefficient for the rate coefficient (different from unity) does not necessarily imply the absence of an annual cycle in wetland effluent concentrations. If the wetland is large enough to reach approximately background concentrations, then there may be a seasonal pattern to the outlet concentration resulting from a seasonal pattern in *C**.

However, when the wetland effluent concentration is close to background, the value of *C** prevails and the rate constant, whether agronomic or microbial, does not control behavior. This situation is known to occur with fair frequency for TSS and BOD. Performance forecasting then depends on understanding the seasonality of *C**.

It is possible to use an Arrhenius factor for *C** (Kadlec and Knight, 1996; Stein *et al*., 2006b). Values ranging from $0.935 < \theta < 1.029$ were reported for COD, depending upon vegetation type (Stein *et al*., 2006b). In general, *C** for COD was found to decrease with increasing temperature ($\theta < 1.000$). A central tendency for a TSS of $\theta = 1.065$ was reported by Kadlec and Knight (1996), which reflects an increase in *C** with increasing temperature.

The parameters for seasonal trends in TSS for several FWS wetlands are shown in Table 7.7, with an example cycle shown in Figure 7.16. On an average, the magnitude of the cycle is 39% of the mean, with a maximum in midsummer on yearday 166. The parameters for seasonal trends in BOD for several FWS wetlands are shown in Table 8.5, with example cycles shown in Figure 8.15. On average, the magnitude of the cycle is 27% of the mean, with a maximum in midsummer on yearday 129.

A hypothetical annual TSS *C** cycle is illustrated in Figure 17.14. If the winter to summer temperature swing is 15°C, the corresponding swing $\theta = 1.056$ for *C**. But note that the TSS is highest in summer, and, therefore, it is the summer peak and the associated scatter that are of concern in design. There is no design control over the TSS background.

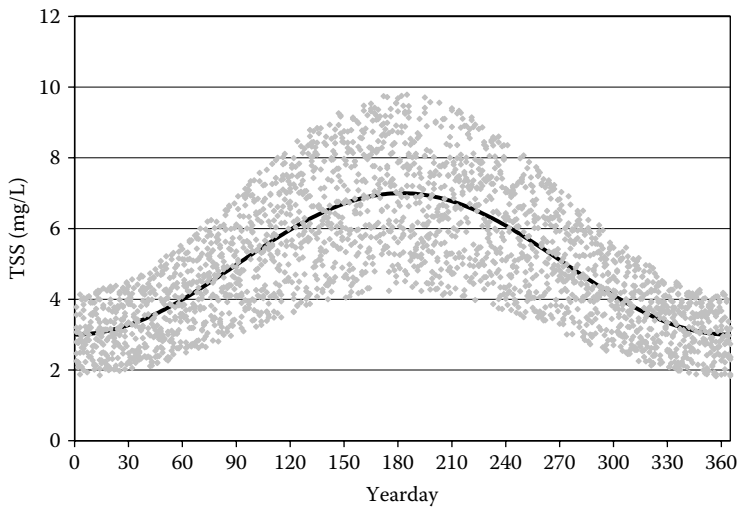



**FIGURE 17.14** Hypothetical scatter of TSS around a seasonal trendline. The annual mean is set at 5 mg/L, and a ± 40% scatter is indicated. The peak TSS occurs in midsummer.

The annual values of BOD and TSS may not reach as low as $C^*$, in which case the annual cycle parameters are used to adjust the design target. The amplitude of the sinusoidal trend can serve to show what "extra" concentrations may occur seasonally. A procedure for accounting the seasonality is to add the seasonal effect to the scatter effect. The trend multipliers tabulated for use with the *P-k-C** model can be extended to include the seasonality. For instance, the scatter in Figure 17.14 is 40%, and can be contained in a design using a trend multiplier of 1.40. However, the summer peak in the trend, 7 mg/L, is also 40% higher than the annual mean of 5 mg/L. The overall multiplier is thereby increased to 1.96, which is used to contain both the random scatter about the mean trend and the seasonal trend. This is the procedure recommended in Kadlec and Knight (1996), who reported a 100th percentile total containment (trend plus scatter) multiplier of 1.9 based on data for 49 wetlands.

### Winter Storage versus Winter Operation

TSS and BOD reductions in treatment wetlands are approximately independent of season, and the decision for winter storage involves only hydraulic concerns with freezing conditions. But, winter rate constants are about one third and one tenth the summer values for phosphorus and ammonia, respectively (Kadlec and Reddy, 2001). Thus, for nutrient removal, the winter removal-process slow-down adds to the hydraulic concerns. There are several options for dealing with winter slow-down, including the following:

1. Provide no winter storage, and increase the wetland size to meet desired winter performance.
2. Provide winter storage by adding storage lagoon volume, but operate the wetland year-round and at a reduced (bleed) flow in winter.
3. Provide winter storage by adding storage lagoon volume, do not attempt to operate the wetland in winter, and control the warm-season releases according to water-temperature patterns.
4. Provide winter storage by adding storage lagoon volume, do not attempt to operate the wetland in winter, and release water from the pond and wetland at a uniform rate throughout the warmest season.

Options 3 and 4 are the choices for climates so severe that the hydraulic operation is questionable because of freezing. In more moderate climates, where ice formation is absent, or hydraulic operation of the wetland can continue with minor ice formation, all options may be considered.

Winter operation is hydraulically possible for lagoons, even for fairly severe cold, where ice thicknesses may range up to a meter (Heaven and Banks, 2005). Indeed, most lagoon systems receive water year-round. Winter operation is also possible for FWS wetlands, even in moderately cold conditions, because water can be managed to flow under ice. Complete freeze-up leads to over-ice water flows and ice buildup. Cold water temperatures lead to large reductions in microbial processes, and the vegetation cycle is dormant in frozen conditions. Oxygen transfer to under-ice water is blocked by ice and snow. These individual process effects combine in complex ways to produce the overall observed slow-down of treatment.

Performance modeling may be used to resolve the requirements of the four options for pond volumes and wetland areas. Typically, the storage options will require the least land (smallest footprint) and the most earthmoving, whereas Option 1 (no winter storage) will require the most land and the least earthmoving. Controlled discharge (Option 3) is operationally the least convenient, because it requires operator attention to set the flows on a periodic schedule throughout most of the year. For North American conditions, although Option 3 often leads to capital investment economics that favor controlled discharge, operational considerations almost invariably lead to a uniform warm season release (Option 2).

### An Illustration

A hypothetical example of some of the design and performance factors for a large-scale system is given in Table 17.10. The flow to the system is 4,000 $m^3$/d in all seasons, and the effects of seepage, rain, and *ET* are assumed negligible for simplicity. The design target for this example is to meet a monthly ammonia effluent concentration of 3 mg/L on an average during all the months of the year, and this value includes the allowance for random variability. The degree of storage pond treatment is as predicted by procedures in the lagoon literature (U.S. EPA, 1983c; Crites *et al*., 2006), and the treatment in the wetland is computed using the *P-k-C** model with $P = 3$, $C^* = 0$, and a temperature dependent $k$ value, and with $k_{20} = 0.05$ m/d and $\theta = 1.09$. The pretreatment facilities, upstream of wetlands and storage lagoons, are presumed to produce 20 mg/L. Under these conditions, the design bottleneck is winter for the no-storage operation. Storage moves the bottleneck to early spring and late autumn, which are the times of lowest temperature during the unfrozen discharge season. Option 2, employing a winter bleed flow through the wetland, has a dual requirement: neither winter nor summer effluent concentrations must exceed the target. Therefore, the wetland area is determined by the spring and fall bottlenecks, and the winter bleed flow is set to meet the target in winter. Option 1, year-round wetland operation, uses a wetland size that meets winter requirements. The flow and storage time series for the various options are shown in Figures 17.15 and 17.16. Associated effluent ammonia concentrations are summarized in Figure 17.17.

The ranking of alternatives indicates an increasing footprint (total area) as storage alternatives move from Options 2 to 3 to 4 (Table 17.10). Option 2, use of a winter bleed flow, has the smallest footprint, 58 ha, just a little more than half the year-round operation footprint of 100 ha. This system includes 87 days of lagoon storage. If the choice is to avoid winter discharges altogether, then the best option is to utilize a controlled release strategy. Small releases are used in early

**TABLE 17.10**
**Summary of Comparison of Winter Flow and Storage Options for a FWS Ammonia Reduction Example**

| | | No Storage | Winter Bleed | | Controlled Release | | Uniform Release | |
|---|---|---|---|---|---|---|---|---|
| **Parameter** | **Unit** | **Wetland Only** | **Pond** | **Wetland** | **Pond** | **Wetland** | **Pond** | **Wetland** |
| Flow | $m^3/d$ | 4,000 | 4,000 | 2,000–7,800 | 4,000 | 2,600–9,400 | 4,000 | 8,000 |
| Inlet *C* | mg/L | 20.0 | 20.0 | 11.3 | 20.0 | 9.7 | 20.0 | 11.3 |
| Max temperature | °C | 17.0 | 17.0 | 17.0 | 17.0 | 17.0 | 17.0 | 17.0 |
| Min temperature | °C | 2.0 | 2.0 | 2.0 | 2.0 | 2.0 | 2.0 | 2.0 |
| Max *k* | m/yr | 14.9 | 9.6 | 14.9 | 9.6 | 14.9 | 9.6 | 14.9 |
| Min *k* | m/yr | 3.9 | 3.7 | 3.9 | 3.7 | 3.9 | 3.7 | 3.9 |
| Storage | days of flow | 0 | 87 | — | 138 | — | 184 | — |
| Working depth | m | 0.25 | 2.0 | 0.25 | 2.0 | 0.25 | 2.0 | 0.25 |
| Max month outlet *C* | mg/L | 3.0 | 13.8 | 3.0 | 11.7 | 3.0 | 11.6 | 3.0 |
| Mean outlet *C* | mg/L | 1.7 | 11.9 | 2.3 | 9.8 | 2.2 | 9.7 | 1.7 |
| Area | ha | 100 | 18 | 40 | 28 | 39 | 38 | 51 |
| **Total area** | **ha** | **100** | — | **58** | — | **67** | — | **89** |
| Volume | 1,000 $m^3$ | 250 | 450 | 100 | 700 | 98 | 950 | 128 |
| **Total volume** | **1,000 $m^3$** | **250** | — | **550** | — | **798** | — | **1,078** |

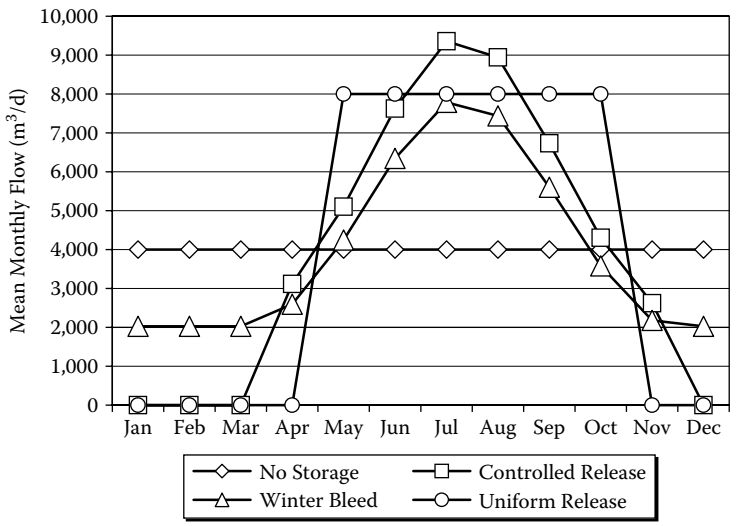



**FIGURE 17.15** Wetland flows for various strategies for the seasonal storage and release example.

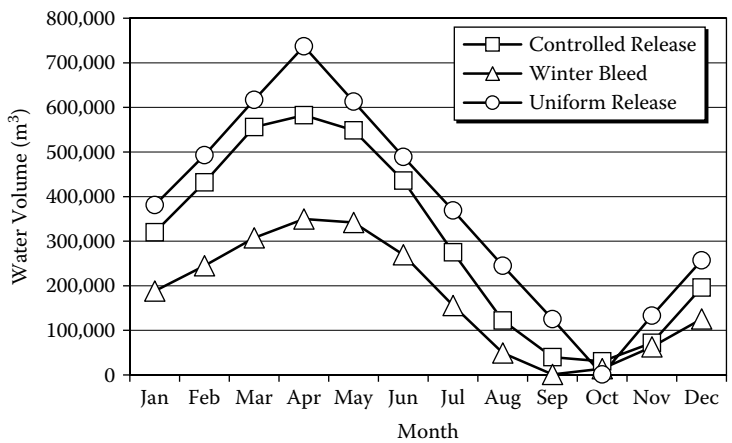



**FIGURE 17.16** Water volumes in storage for a seasonal storage and release example.

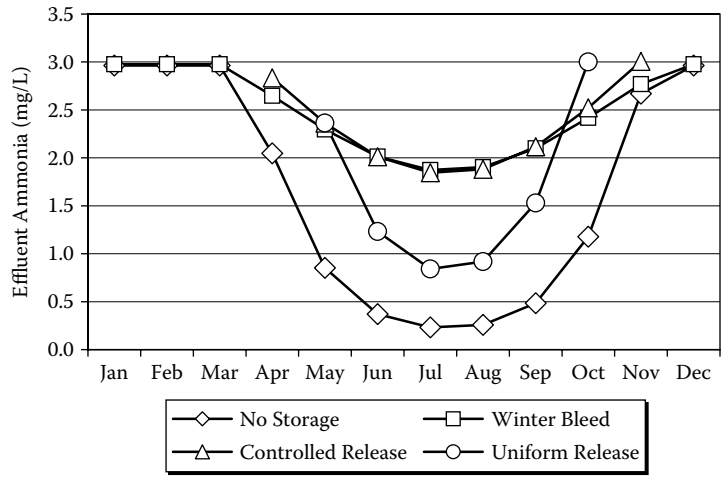



**FIGURE 17.17** Calculated wetland effluent ammonia concentrations for different storage strategies for a seasonal storage and release example.

spring and late autumn, and maximum release takes place in midsummer, the time of warmest water temperature. An area of 67 ha, containing 138 days of storage, is forecast to achieve the required ammonia standard. However, the convenience of a steady summer discharge might lead to the choice of the 89 ha system, which has 184 days of storage. The extra two months of storage are needed to avoid the (high) uniform flows during early spring and late autumn, or else the wetland will become nearly as large as that for no storage. Land acquisition, and perhaps liners, are area-specific costs, which form a large proportion of the total system capital cost and, hence, Options 2 and 3 are favored for lowering those costs.

Earth moving is another major cost in natural systems, and there are large differences among the three alternatives in the illustration. Year-round wetlands require the least earth moving, and a complete winter shutdown requires the most. Table 17.10 shows that total basin volumes (lagoon plus wetland) of the alternatives vary widely for the example, with the winter operation of the wetland needing only 25% of the earth moving that a uniform summer discharge system would require. Thus, the presence or absence of liner requirements, or the type and cost thereof, can shift the economic advantage from one option to another. This example suggests that wetlands should be used in winter if hydraulically practical, because land and liner costs typically will not outweigh earth-moving costs.

### Wetland Systems with Storage

There are considerable numbers of treatment wetlands that operate seasonally, with off-season storage (Table 17.11). Many of these are preexisting lagoon systems with wetlands added, so storage was already available. A common mode of operation of lagoons and facultative ponds is pulse discharge during spring or autumn, thus taking advantage of the maximum dilution potential while receiving surface waters. Some or all of the needed storage capacity may, in fact, be present prior to wetland add-ons.

## 17.6 STORMWATER WETLANDS

Stormwater wetlands are nearly always FWS systems because of the need to hydraulically accommodate large-event flows and because there is no perceived health hazard associated with the water. Design follows one of the four following tracks:

1. Percentage of the contributing watershed
2. Design storm detention
3. Annual average performance
4. Detailed dynamic modeling

The first two of these are technology-based because they provide prescriptions that imply a general level of treatment rather than specific numerical concentration or loading goals. Technology-based volume or area specifications are attractively simple because they do not require any information about the incoming water chemistry. They suffer from an inability to be adjusted for removal targets other than the original presumptions. The second two are performance-based methods, which depend on forecasts of wetland reductions of target pollutants. Performance-based sizing calculations may be either on an average basis, usually annual, or on a short time scale to capture the full dynamics, usually daily. For example, annualized performance can be related to hydraulic loading and inlet TSS by using Equation 14.8 as a design prediction tool. Exploration of this formulation shows that it is consistent with the technology-based procedures in the 75–80% removal range.

All of the methods require knowledge of the runoff volume characteristics of the contributing watershed; the performance-based methods also require the incoming contaminant concentrations.

Associated with stormwater wetlands is the concept of bypassing. Bypassing is necessitated when site constraints prevent the treatment of all imaginable flows. The constraint may occur because of limited land availability, which creates

**TABLE 17.11**
**Characteristics of Pond Wetland Systems by Latitude**

| System Name (Years of Operation) | Type | Area (ha) | Latitude | Winter Operation | | HLR (cm/d) | TSS (mg/L) | BOD (mg/L) | $NH_4$-N (mg/L) | TP (mg/L) | FC (#/100 mL) |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Houghton lake, Michigan (29) | P–W | 200 | 44 | N | in | 0.3 | 30 | 17 | 6.1 | 3.15 | 89 |
| | Nat | | | (warm) | out | — | 10 | 4 | 0.08 | 0.04 | 20 |
| Minot, North Dakota (12) | P–W | 51 | 48 | N | in | 3 | 32 | 16 | 4.5 | — | 381 |
| | Con | | | (warm) | out | — | 14 | 10 | 2.1 | — | 219 |
| Cannon beach, Oregon (17) | P–W | 7 | 45 | N | in | 25 | 39 | 31 | — | 12.2 | 19 |
| | Nat | | | (wet) | out | — | 9 | 8 | — | 4.6 | 53 |
| Lake Nebagamon, Wisconsin (1) | P–W–I | 0.8 | 46 | N | in | 2.7 | 30 | 10 | 4 | — | — |
| | Con | | | (warm) | out | — | 10 | 4 | 0.05 | — | — |
| Bellaire, Michigan (12) | P–W | 7 | 44 | N | in | 1.0 | — | — | 8.54 | 2.78 | — |
| | Nat | | | (warm) | out | — | — | — | 0.73 | 0.30 | — |
| Drummond, Wisconsin (24) | P–W | 6 | 46 | N | in | 0.14 | — | — | 0.13 | 2.36 | — |
| | Nat | | | (warm) | out | — | — | — | 0.17 | 0.94 | — |
| Musselwhite, Ontario (6) | P–W | 2.5 | 54 | N | in | 52 | — | 6 | 12.8 | 0.018 | — |
| | Nat | | | (warm) | out | — | — | 5 | 3.9 | 0.012 | — |
| Estevan, Saskatchewan (11) | P–W | 11 | 49 | N | in | 3.0 | 33 | 9 | 2.82 | 2.49 | — |
| | Con | | | (warm) | out | — | 24 | 5 | 0.35 | 2.18 | — |
| Saginaw, Michigan (5) | M–W | 8 | 43 | N | in | 0.30 | — | — | 457 | — | — |
| | Con | | | (warm) | out | — | — | — | 17 | — | — |
| Isanti-Chisago, Minnesota (6) | M–W | 0.6 | 46 | N | in | 3.5 | 45 | — | 8.9 | — | — |
| | Con | | | (warm) | out | — | 4 | — | 0.76 | — | — |
| Vermontville, Michigan (35) | P–W | 4.6 | 43 | N | in | 1.00 | — | — | 6.5 | 1.8 | — |
| | Con | | | (warm) | out | — | — | — | 5 | 0.64 | — |
| Mt. Angel, Oregon (11) | P–W | 3.6 | 45 | N | in | 13.3 | 11 | 25 | 2.6 | 1.5 | — |
| | Con | | | (wet) | out | — | 8 | 14 | 2.2 | 1.4 | — |

*Note:* Years = operational period; operation is yearlong (Y), warm, wet, or dry season; P = pond, AP = aerated pond, W = wetland, I = infiltration.

*Source:* Adapted from Kadlec (2003d) *Water Science and Technology* 48(5): 1–8.

a hydraulic conveyance constraint on how much water can be passed through the system. For instance, the intense rains of subtropical regions can occasionally create runoff flows that simply cannot be routed through a reasonably sized wetland. The second constraint arises owing to the availability of maintenance water. If the wetland is sized large enough to accommodate major events, then the demand for maintenance water during interevent periods may be too large.

Agricultural stormwater systems might differ somewhat from urban stormwater systems because the target may be sediment or nutrient control and load or concentration reduction.

Chapter 14 has provided the experience base for the design of stormwater wetlands. Here, the design sizing procedures are outlined. These rely on source characterization, just as for the continuous flow wetlands discussed in Chapters 14 and 16.

### Inflow Estimation

The amounts of water to be expected to reach the wetland depend on the size and character of the watershed. Rainfall records provide an accessible and useful means to determine the amount of water received by the watershed, and runoff fractions provide the resultant flows to the wetland. Runoff coefficients vary with land use; they are typically high (80–90%) for impervious surfaces but lower (20–30%) for more absorbent conditions in the watershed (Table 14.1). An example of the determination of the runoff coefficient for particular applications is the New York calculation (Center for Watershed Protection, 2001):

$$\text{Runoff coefficient} = R_V = 0.05 + 0.009I \qquad R_V \geq 0.20 \tag{17.21}$$

where

$I$ = percent impervious

However, regional considerations can easily preempt generic information such as the impervious cover estimates summarized in Table 17.12. For instance, runoff coefficients for the Everglades agricultural area are about 50%, not because of imperviousness but because of extremely high water tables

**TABLE 17.12**
**Imperviousness of Various Land Uses**

| Land Use Category | Mean Impervious Cover |
|---|---|
| Agriculture | 2% |
| Open urban land (parks, golf courses, cemeteries) | 9% |
| One-acre lot residential | 14% |
| Quarter-acre lot residential | 28% |
| Multifamily residential | 44% |
| Institutional (schools, churches) | 28–41% |
| Light industrial | 48–59% |
| Commercial | 68–76% |

*Source:* Data from Center for Watershed Protection (2001) *New York State Stormwater Management Design Manual*. Report by the Center for Watershed Protection (CWP) for the New York State Department of Environmental Conservation, Albany, New York.

and high rainfall. Accordingly, runoff should be considered highly site specific.

A design storm is specified according to regulatory or negotiated considerations. For instance, the 90th percentile of the event number distribution is used in New York (90% of all rain events are less than this event). The rainfall amount varies with geographical location but is approximately 3 cm in the northeastern United States. Given the area of the watershed and the runoff coefficient, the volume of water that the design event delivers to the wetland, $V_{design}$, may be determined. This amount may then be converted to the annual volume reaching the wetland, via the ratio of the annual total rainfall to the amount of the 90th percentile. More detailed estimates may be made on the basis of the variation of $R_V$ with the rainfall amount, whereby very small rains are assumed to produce no runoff. Further, by implication, larger events are bypassed, either in total or partially.

## Hydraulic Considerations

Interpretation of performance for event-driven wetlands depends on the amount of a given event that is retained in the system and by the length of the interevent period, rather than by detention time or the average hydraulic loading. Detention and loading may, nonetheless, be computed if desired. As an example, consider a wetland of volume $V$ m$^3$ that quickly receives the same volume of water as an inflow. If the wetland were in plug flow, then the new water would just displace the antecedent wetland water. That effluent water would have been subjected to batch treatment over the period since the preceding event. Thus, the effluent concentration would be determined by the concentration that entered with the preceding event and by the detention during the interevent period, and would have nothing to do with the concentration of the water that displaced it.

Stormwater wetlands receive pulses of widely varying volume and frequency and are not usually anywhere close to plug flow. The TIS model may be used to explore how much of the water of events of different sizes will stay within the wetland (see Chapter 6 for details of the TIS model). In general, the fraction of the event that remains in the wetland increases as the number of tanks increases, to the limit of plug flow (see Figures 14.5 and 14.6). Some existing design methods specify the particular return frequency event for which the runoff is to be nominally contained in the wetland (Schueler, 1992). This design event is characterized by the value 1.0 in Figure 14.5. At this event size, the actual fraction of the event that is retained is strongly dependent on $N$, the number of TIS. It is therefore desirable to maximize this number in the design. Walker (1998) suggested that computational fluid mechanics could be used to investigate the geometrical design factors that contribute to the retention that is to be expected. This procedure utilizes a dispersion coefficient to explore deviations from plug flow, and it leads to Figure 14.6.

## Percentage of the Contributing Watershed

The amount of water and pollutants that reach stormwater treatment wetlands from contributing watersheds are not typically known in advance. The number and duration of the events that produce the input, together with the interevent spacing, are thought to affect the efficiency of the stormwater wetland. Because of this variability of actual flow rates, detention time and hydraulic loading are difficult to define, and other single-number sizing rules have evolved. One such method of presumptive sizing is a specification of the wetland-to-watershed area ratio (WWAR). This rule of thumb states that the wetland size should be a specified fraction of the contributing watershed, usually in the range of 1.0–5.0%. A little arithmetic shows that this is equivalent to the range of hydraulic loading rates cited earlier for continuous flow treatment wetlands. In a moderate climate,

Annual rainfall = 60 cm
Average annual rain rate = 60/365 = 0.164 cm/d
Watershed runoff coefficient = 0.75
Runoff = 0.75 × 0.164 = 0.123 cm/d
Watershed/wetland area ratio:
WWAR = 1/.025 = 40 (for 2.5% of the watershed)
Average annual wetland HLR = 40 × 0.123 = 4.9 cm/d

Because this average annual HLR is close to the median of the distribution of HLRs for FWS wetlands (e.g., 4.0 cm/d for TSS systems and 4.8 cm/d for TP systems), it is reasonable to expect that stormwater wetlands designed in this way would perform somewhere near the average for the emergent marsh database set. For instance, the mean reduction for phosphorus in 284 FWS wetlands was 38% at a median HLR = 4.8 cm/d; the mean reduction for several constructed stormwater marshes with an average WWAR = 4.3% was also 57% (Strecker *et al*., 1992).

Ratios of 2% for marshes alone and 1% for pond–marsh combinations have been advanced (Schueler, 1992), but some authorities require more. Ontario and Alberta target 5%. However, such generalizations are dangerous as they do not extend to other event-driven wetland applications. For example, Braskerud (2001a) found 50–60% TSS removal in four wetlands over a six-year period, with WWAR between 0.03 and 0.07%.

## Design Storm Detention

A stormwater wetland may also be sized to contain a specific volume of water, usually the volume associated with a rain event of a specified return frequency or probability of occurrence. For instance, Schueler (1992) suggests that wetlands have sufficient volume to fully contain any rain event up to the 90th percentile of the rainstorm quantity distribution. Again, this may be shown to match the loading and detention design ranges used for point-source wetland systems.

In the vicinity of Washington, D.C., there is 104 cm annual rainfall, and the 90th percentile storm is 3.18 cm:

Watershed area = 40 ha = 400,000 $m^2$
Watershed runoff coefficient = 0.75
Design storm runoff volume = 0.75 × 0.0318·400,000 = 9,540 $m^3$
Wetland area @ 0.3 m depth = 9,540/.3 = 31,800 $m^2$ = 3.18 ha
WWAR = 100 × (3.18/40) = 8%
Annual flow = 0.75 × 1.04 × 400,000 = 312,000 $m^3$
Average annual detention time = 9,540/312,000 = 0.03 years = 11 days
Average annual wetland HLR = 312,000/31,800 = 9.8 m/yr = 2.7 cm/d

This single-number design technique has the advantage of allowing a variable percentage of watershed, depending on the annual rainfall pattern and annual rainfall total. As in the case of WWAR design, the loading and detention times correspond to the median values for point-source treatment wetlands. It is, therefore, not surprising that Schueler (1992) lists pollutant reductions that are in the mid-range for other treatment wetlands. For instance, the total phosphorus removal is projected to be 45%, compared to the 38% mean for FWS systems in general.

The selection of volume or area also corresponds to a projected level of TSS removal. For example, Schueler (1992) suggests that 75% TSS reduction can be reliably achieved by stormwater wetlands that capture the 90th percentile of rain events. Although confirmation of that assumption remains elusive, the criterion has been adopted in New York (Center for Watershed Protection, 2001). Other states, such as Wisconsin, have adopted similar criteria, which recommends a design rain of 3.8 cm and suggests that this corresponds to an 80% reduction (Prey *et al.*, 1995).

## Annual Average Performance

Another level of detail may be added via annual average performance calculations. The average hydraulic loading is computed from individual events of the year, utilizing precipitation records and the definition of the maximum event to be treated. Incoming concentrations are determined from the watershed data or from estimates. The flow-weighted concentrations are used. Both regression equations and rate coefficient formulations have been proposed.

### Regression Equations

Duncan (1998), as referenced and reported by Wong *et al.* (1999), suggested the use of regression equations for purposes of sizing. He presents relations based on the analysis of 76 stormwater wetland systems, presumably including pumped systems. The proposed relations are

$$\text{TSS percent remaining} = 78q^{0.33}C_i^{-0.49} \quad R^2 = 0.80 \tag{14.8}$$

$$\text{TP percent remaining} = 12q^{0.44} \quad R^2 = 0.71 \tag{14.9}$$

$$\text{TN percent remaining} = 14q^{0.43} \quad R^2 = 0.78 \tag{14.10}$$

where
$C_i$ = inlet concentration, $g/m^3$
$q$ = hydraulic loading rate, m/yr

Carleton *et al.* (2001) also suggest that the WWAR may be used as a predictor of performance. Their regression equations are developed for small data sets, including natural wetland systems, and have low correlation coefficients ($R^2 \approx$ 0.15–0.45). Within the range 0 < WWAR < 0.1, the data scatter was particularly large. Regressions of the data for TSS, TP, and nitrogen compounds in Tables 14.5–14.10 are generally unsatisfactory. For TSS, for example, the remaining fractions increase slightly with WWAR.

### Annualized Rate Coefficients

It has been suggested that the design techniques from continuous flow wetland performance may be transferred to event-driven systems by using rate coefficients to forecast average performance (Wong and Geiger, 1997; Carleton et al., 2001). However, the rate coefficients are not necessarily the same as those found for the continuous flow systems. Stormwater data sets analyzed by Strecker *et al.* (1992) and extended by Carleton *et al.* (2001) include about half of the natural wetland systems. Strecker *et al.*, in fact, found that natural wetlands performed somewhat better than constructed wetlands, but the use of natural wetlands is not encouraged owing to real and perceived negative impacts (U.S. EPA, 1993e).

The median annual areal rate constants reported by Carleton *et al.* (2001) for FWS-constructed gravity systems (N = 9) are as follows:

| | |
|---|---|
| Total phosphorus: | 8.3 m/yr |
| Ammonia: | 5.0 m/yr |
| Nitrate: | 6.7 m/yr |

These are very low compared to the *k*-values reported for continuous flow systems in Part I. Analysis of the few detailed data sets that exist indicates that higher rate coefficients than these apply to individual events. For instance, Kadlec (2001a) reported $k = 100$ m/yr for phosphorus, and Werner and Kadlec (2000b) measured 125 m/yr for nitrate, both based on pumped event dynamics.

The designs of the 16,000 ha of constructed stormwater treatment areas (STA wetlands) of South Florida, which target phosphorus removal, were essentially based on annualized rate coefficients. The prototype was an event-driven natural wetland that produced $k = 10$ m/yr for $C^* = 2$ μg/L and $P = \infty$ (Walker, 1995). Subsequent analyses have shown that the constructed systems have annualized *k*-values of the same general magnitude. However, the dynamic *k*-values are higher (Walker and Kadlec, 2005). Further, the *k*-values are sensitive to the characteristics of the event sequences. It is likely that more accuracy can be gained only by dynamic simulations.

## Detailed Dynamic Simulations

The most detailed design calculations involve dynamic simulations of wetland performance, accounting for nonideal flow patterns and the time series of flows and concentrations that enter the wetland. This procedure has been calibrated for some few wetlands (Wong *et al*., 2006), but the input parameters are of unknown transferability to other wetlands and sediment types. For instance, data from stormwater wetlands at Elbow Valley, Alberta (Amell, 2004), indicate a first-order removal rate coefficient for TSS on the order of 0.6–0.9 m/d, whereas the rate coefficient found by Wong *et al*. (2006) were 16–21 m/d. The magnitude of this difference is not surprising, given the extreme variability in the TSS settling rates, shown in Figure 7.8. There are some suggested procedures that account for the size distribution of the incoming TSS (Lawrence and Breen, 1998), but such measurements require significant data acquisition for the source water. In terms of model nomenclature, the parameters *k*, *C**, and *N* are not known in advance. Wong *et al*. (2006) found it necessary to calibrate the dynamic TIS model for the number of TISs. Treatment wetland technology is not yet capable of determining that number *a priori* from design and layout details. Computational fluid mechanics would offer a way to do this (Walker, 1998), but there is not yet a verification of the accuracy of this procedure using wetland field data.

At present, the only generally available dynamic simulation tool developed solely for the design of event-driven wetlands is for phosphorus removal in warm climates. The dynamic model for stormwater treatment areas (DMSTA) computer routine is given at Web site http://www.wwwalker.net/dmsta/. This model has seen extensive use in the design of constructed wetlands in South Florida, including eight large systems of the order of 1,000 ha each. There are numerous other computer codes that have varying capabilities to deal with time-variable stormwater flows and concentrations, together with some form of constituent tracking. Some of these have been listed in Chapters 9 and 10. However, calibrations to wetland data are typically lacking, except for DMSTA, which has been calibrated to more than 100 wetland data sets, ranging from mesocosms to large full-scale systems.

A key feature of the dynamic simulation of event-driven wetlands is the inclusion of the storage of contaminants. Nutrients are stored in vegetation and sediments and cycled on a preset time scale. Movements of pollutants into and out of storage have important effects on the long-term reductions in these pollutants.

Detailed dynamic simulations are very data hungry. Rather extensive time series of inflows, concentrations, and meteorology are needed. The probable rainfall sequence may be constructed from historical records, but the inflows and concentrations will typically have to be estimated. As a consequence, full dynamic simulations are likely to be restricted to situations involving large and expensive wetlands.

## Flow Equalization versus Pulse Operation

Event flows create large instantaneous loads on a treatment wetlands that, if not contained, will be the cause of low degrees of treatment. The peaking factor (PF) may be defined in a manner similar to that used to describe the fluctuations in continuous flows, namely, as the ratio of the peak flow to the mean flow. To illustrate the effect of flow equalization, consider a highly idealized example in which an average of 2,000 $m^3$/d at 100 mg/L is to be treated in 8 ha. The *P-k-C** values are selected to be $P = 3$, $k = 10$ m/yr, and $C^* = 0$. For continuous, uniform flow, the calculated exit concentration is 39 mg/L. Suppose the peaking factor is 10, i.e., the flow all occurs at 20,000 $m^3$/d, but only for 10% of the time. The calculated exit concentration is then 90 mg/L. The reduction has dropped from 61% to 10%. This example is unrealistic for several reasons; for example, there are changes in storage in the water body, which have not been accounted.

Next, we broaden this illustration to include storage and (batch) treatment during the interevent periods. The event sequences will be limited to a time series of ten-day events, separated by periods that increase as the peaking factor increases. The average flow is maintained at 1,000 $m^3$/d. Treatment continues during the interevent period, but there is no flow. Water from the next flow event mixes with the antecedent water and displaces it.

It is supposed that the design should achieve 50% reduction; two alternatives are to be compared: a wetland only, and a storage pond plus wetland. The pond is sized to fully equalize the flow sequence and thus generate a uniform continuous flow. The pond performs no treatment.

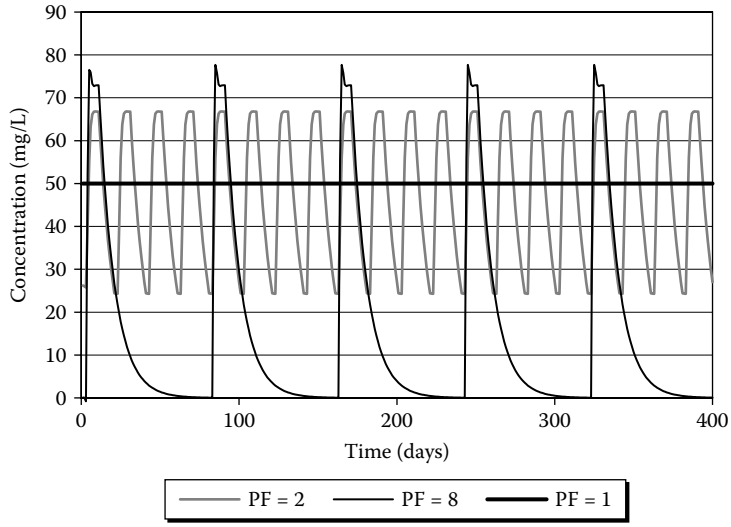



**FIGURE 17.18** Outlet concentration time series of equalized and non-equalized flow scenarios for event train treatment in a hypothetical example. See text for assumptions and conditions.

First, consider the alternative of allowing the pulses to pass through the wetland with no equalization. The concentration time series for the three peaking factors for non-equalized flow are shown in Figure 17.18. The case of PF = 1 corresponds to the continuous uniform flow and produces a uniform outlet concentration of 50 mg/L. When the PF = 2, the flow occurs only 50% of the time, and the wetland acts as a batch reactor the other 50% of the time. Concentrations swing from 25 mg/L at the end of the batch time to 65 mg/L at the end of the flow time, but the flow-weighted outlet concentration remains at 50 mg/L. When the PF = 8, the flow occurs only 12.5% of the time, and the wetland acts as a batch reactor the other 87.5% of the time. Concentrations swing from 0 mg/L at the end of the batch time to 75 mg/L at the end of the flow time, but the flow-weighted outlet concentration remains at 50 mg/L. Clearly, in the hypothetical example, event flows lead to large swings in outlet concentrations; and, indeed, this is expected and observed for real systems as well (see Figure 14.9 for example).

The wetland sizes to accomplish a flow-weighted outlet concentration in the hypothetical example range from 2.85 ha for continuous flow to 24.4 ha for a peaking factor of 16. High peaking factors are in fact observed; for instance, they range up to 13 for STA6 in South Florida (see Figure 14.9). The question thus arises as to the desirability of adding flow equalization to the treatment train. In the hypothetical example, it is supposed that a storage pond would operate in a working depth range of 0.0–2.5 m, exclusive of freeboard and minimum depth requirements. It is then possible to compute the pond volume and area necessary for providing full equalization. Under the equalized scenario, the wetland continues to be sized at 2.85 ha. Figure 17.19 shows, for the

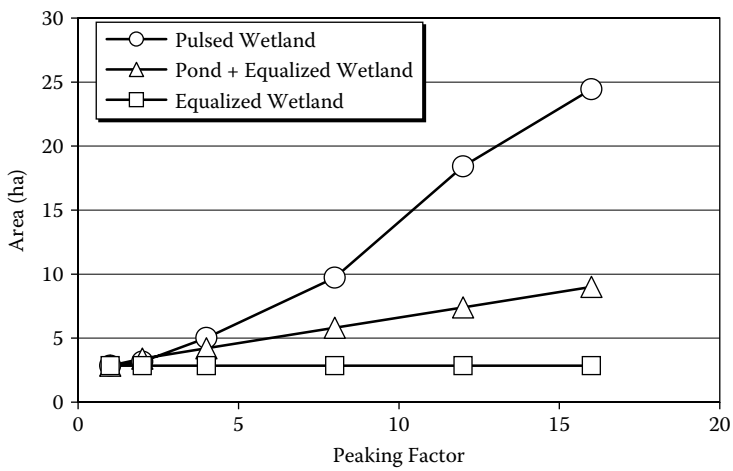



**FIGURE 17.19** Total working footprint area of equalized and non-equalized flow scenarios for event train treatment in a hypothetical example. See text for assumptions and conditions.

conditions of the hypothetical example, that the area requirement for the equalized flow alternative is always less than that for a pulsed wetland with no equalization and, at PF = 16, the combined storage plus wetland is only 37% as large as the pulsed wetland, to achieve the design goal of 50 mg/L. However, rough estimates suggest that earth moving would not vary much between alternatives (not shown). As a result, the decision to equalize would normally be made on the basis of land cost and availability.

Of course, the results of this hypothetical example should not be used in an actual design decision, because it is highly unlikely that all the conditions would be met. Nevertheless, the example provides an illustration of the concepts that would be exercised in a real design situation. As will be seen, most urban stormwater systems would contain a small forebay for solids collection, and thus provide some small measure of equalization, in any case.

## SUMMARY

This chapter has integrated the analytical elements of preceding chapters into a sequential approach to designing and evaluating a FWS treatment wetland system. Sizing is the first step in design, for which complementary techniques have been integrated. First-order removal equations are used first. The rate coefficients are selected from the percentile tables of Part I. Hydraulic efficiency is chosen via the selection of the number of TIS. Calculations include the possibility of a nonzero background concentration, corresponding to a return flux from the ecosystem. Rate constants and background concentrations vary with season or temperature in some instances. Goals may include concentration reduction and/or load reduction, which are conflicting objectives. Excursions may need to be contained, for which the exceedance frequency data of Part I is used. The model equations are solved on spreadsheets, with a tank-to-tank calculation procedure. Nitrogen compounds further require linked species computations. Wetland area is adjusted to meet the project objectives.

There is nothing about the rate-constant approach that acknowledges the limitations of the wetland biology or ancillary chemistry. Therefore, the cycling of carbon, nitrogen, and phosphorus is checked to ensure that there are no unreasonable requirements for the ecology, oxygen, and carbon supplies. Rate coefficients may need adjustment if the presumed capacities of the ecosystem are exceeded. Loading charts are a method of understanding the behavior of existing systems. Designs that fall outside the range of the existing data scatter would require further justification.

For some pollutants, adjustments to the design may be required to account for seasonal or temperature-driven behavior. Nitrogen reduction is the primary situation in which either seasonality can occur for agronomic systems or temperature-driven rates can occur for microbially-driven wetlands. If the winter season creates too much of a constraint, leading to very large wetlands, then the possibility of winter storage requires consideration.

Stormwater wetland design may follow a similar path, but technology-based design strategies are also available. The time-variable nature of event-driven systems can lead to a requirement for dynamic simulation in design.

# 18 Implementation of FWS Wetlands

The purpose of this chapter is to provide information that will increase chances for successful wetland establishment and to help with understanding and overcoming difficulties that may arise during construction and start-up of wetland treatment systems.

Wetland treatment system establishment can be subdivided into three topics: (1) physical design, (2) construction, and (3) vegetation establishment. Physical design encompasses all the choices of how the wetland basin is built, and how the water is conveyed to and through the system. With the exception of urban stormwater wetlands (Schueler, 1992), there is no single comprehensive literature source that explains options for layout and configuration of the wetland cells that typically comprise a treatment wetland.

Site civil construction is a broad discipline of engineering and has been refined through centuries of experience. This chapter does not attempt to discuss all aspects of this engineering practice but rather describes the highlights of site civil construction that are most relevant to wetland treatment system construction. A brief general discussion of wetland site civil construction is provided by Tomljanovich and Perez (1989). All aspects of wetland construction are discussed in detail in the *Wetlands Engineering Handbook* (U.S. Army Corps of Engineers, 2000). This handbook discusses engineering procedures for establishing necessary hydrologic conditions, geotechnical design and soils handling for site modification, selecting appropriate vegetation and planting schemes, and establishing substrate conditions conducive to the desired functions. It also discusses baseline assessments of existing site conditions, monitoring strategies to determine long-term success, and contracting considerations. However, the objective of this handbook is the construction of mitigation/habitat wetlands, not treatment wetlands.

Establishment of wetland vegetation seems deceptively simple, but is not, and is frequently outside of the experience of many site civil contractors. Therefore, this important aspect of treatment wetland establishment should generally be subcontracted to specialists, nursery owner/operators, or wetland builders.

## 18.1 PHYSICAL DESIGN

The size of the wetland has been determined, at least initially, by the procedures of the preceding Chapter 17. The general characteristics of the site have been investigated in Chapter 16. There remain a number of design decisions on arrangement, vertical and horizontal placement, and methods of moving and controlling water.

### Siting

General conditions of the potential site will have been considered during the establishment of the basis of design. In general, there will remain a number of possibilities for the location of the system within the overall site confines. Conditions that should be evaluated during physical design of a wetland treatment system include geography, soils and geology, runoff water volume and chemistry, ecological and socioeconomic factors, and ancillary regulations. The importance of each of these conditions may vary, but all should be investigated to some extent. Detailed studies may be needed to determine the importance of those site conditions that affect technical feasibility.

### Floodplains and Floodways

A hypothetical siting situation is shown in Figure 18.1, which indicates that the regional waterways, a river and its tributary, are geographic features to be reckoned with. Constructed wetlands located in floodplains with extreme flood conditions, such as along major rivers, must have dikes that are designed to allow the passage of floods and/or that are sized to exclude flood waters. Constructed treatment wetlands have typically been allowed in floodplains under constrained conditions. Designs should avoid damage due to high-frequency recurrence events, and therefore would usually be built to withstand the 25-year flood event without impairment of function. Similarly, wetland facilities would usually be designed to withstand the 100-year flood event without severe damage. However, floodplains may be jurisdictional wetlands, thus creating regulatory questions. Nevertheless, many FWS wetland systems have been constructed in floodplains such as those at Columbia, Missouri; Brawley and Imperial, California; and Tres Rios, Arizona.

If a wetland is built in a floodplain, it is necessary to demonstrate that its presence will not back up floodwaters upstream of the project. In other words, the project should not block the floodway of the river. For most major rivers of the United States, there exist published maps showing the boundaries of the 100-year floodplain and the floodway.

### Other Regulatory Concerns

Wastewater treatment and disposal are regulated by an ever increasing number of federal, state, and local laws, rules, ordinances, and standards. In some cases, the most challenging part of implementing a wetland treatment project is complying with regulations through the permitting process. In fact, regulations may hinder innovative project design

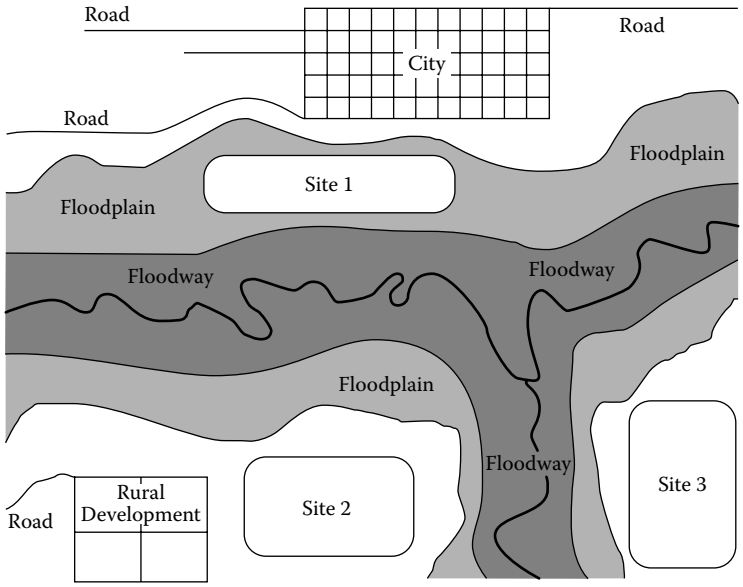



**FIGURE 18.1** Siting and flood concerns for a hypothetical treatment wetlands project. Communities are adjacent to a river and its tributary. The floodway occupies the central valley corridors and is bounded by the floodplain. Three potential constructed wetland sites are indicated, one in the floodplain.

for wetland treatment systems, even when the new design improves upon existing technology. Permitting an innovative project may delay implementation and increase cost with no incremental benefit for environmental protection. As a result of regulatory constraints, designers may opt for conventional technology with well-known deficiencies instead of technology that is perceived to have a greater risk—but with potentially high environmental benefits.

A detailed knowledge of pertinent regulations is essential to evaluate the feasibility of a wetland treatment project. This section provides an overview of the regulations that affect the use of wetlands for wastewater treatment in the United States as well as examples of some specific state situations. An updated and more detailed survey of federal, state, and local ordinances should be conducted to determine those that might be relevant to specific projects.

## Endangered Species

Under the Endangered Species Act, the U.S. Fish and Wildlife Service is mandated to identify and list plant and animal species that are threatened or endangered by extinction or that are considered likely to be threatened in the future. Threatened and endangered species cannot be harmed, killed, or otherwise negatively affected by human activities. The U.S. Fish and Wildlife Service reviews potential impacts to federally listed species and can veto Clean Water Act permits (such as a Section 404 wetland fill permit or an NPDES permit) by issuing a "jeopardy" opinion.

The potential for occurrence of threatened or endangered species inhabiting a project site should be considered during project planning. Typically, a list of threatened and endangered species that could occur at a project location is compiled from federal and state natural resource agency records. Potential project impacts on any of these species are assessed. If negative impacts on a species are anticipated during this review, it may be prudent to conduct a field survey to ascertain whether that species actually occurs at the proposed site. This confirmation step is important in eliminating the chance of finding a "fatal flaw" after considerable planning, design, or construction monies have been spent.

Experience indicates that the presence of federally protected species does not always necessitate substantial changes to wetland treatment project siting or design. In a number of cases, the discharge of treated wastewater effluents to constructed wetlands has enhanced the population of listed wetland-dependent species by creating additional habitat and food resources. For example, populations of bald eagles, wood storks, and snail kites, all federally protected bird species, have been increased in and around the 485-ha Orlando Easterly constructed wetland treatment system in Florida. Thus, the potential impacts of a project on protected species must be evaluated on a species-by-species basis. As a further illustration, the operation of the 2,400-ha STA1E treatment wetland in south Florida was prevented for a period of weeks because of the nesting activities of burrowing owls (*Athene cunicularin floridana*) and stilts (*Himantopus mexicanus*) (Figure 18.2).

## Cultural Resources

The U.S. National Historic Preservation Act requires that sites of significance for the early peoples of America be preserved. That normally involves avoidance of construction activities that might disrupt ancient village sites and burial grounds. For example, a historic archaeological site was excluded from the Coyote Hills, California, urban stormwater wetland (Figure 18.3). The Musselwhite, Ontario, mine-water treatment



**FIGURE 18.2** Nesting black-necked stilts (*Himantopus mexicanus*) caused brief stoppage of the STA1E project in Florida.

wetland was sited with due recognition of the line-of-sight from the regional First Nations burial ground.

### Layout and Configuration

The area determined to achieve the project treatment goals may typically be arranged in a number of ways. The principal decisions are

1. How many independent flow paths?
2. How many cells in each?
3. What aspect ratio for the cells or flow path?

There is a need to set the compartmentalization of the system. The number of flow paths in the design of surface flow wetlands is based on considerations of redundancy, maintenance, and topography. All constructed wetland treatment systems should have at least two cells that can operate in parallel to allow for operational flexibility (cell resting, rotation of flows, or maintenance). The ability to take cells out of service is usually viewed as very desirable. Maintenance activities do not normally require frequent shutdowns, but it is usually a good idea to allow at least two flow paths so that complete system bypass is not needed. Having at least two parallel cells is especially important because of unexpected events such as vegetation die-off, pretreatment failures and subsequent wetland contamination, and berm and other structural failures. Multiple flow paths allow the loading rate to be manipulated to meet varying inflow water quality. Also, parallel flow paths allow cells to be drained for replanting, rodent control, harvesting, burning, leak patching, or other possible operational controls. In the extreme long term, replacement of structures and piping become necessary. Some of the older FWS treatment wetlands are now reaching this point in their service life. Large systems may profitably incorporate more than two flow paths for purposes of internal flow control and to accommodate the site boundaries, site




**FIGURE 18.3** This archaeological site was excluded from the Coyote Hills urban stormwater wetland.

topography, and desired aspect ratios. However, multiplicity of inlet and outlet control structures can add significant cost to the overall project.

## Compartmentalization

Compartmentalization interacts with the design-area calculation because more cells and higher aspect ratios lead to higher tanks-in-series (*N*TIS) numbers, and hence to higher *P*TIS numbers. As seen in the previous chapters, higher *P*TIS does not help much for low desired-removal percentages but is a necessity if very high percentages are required. Aspect ratio is also a determinant of *N*TIS, with fewer for low length-to-width ratios. A single FWS wetland cell of modest aspect ratio ($1 < L/W < 5$) is likely to behave as three or four *N*TIS in tracer testing. When compounded with the weathering factor that influences most wastewater parameter reductions, it is likely that a *P*-value for the *P-k-C** model will be two or three. That may be perfectly adequate if the treatment does not attempt to achieve outlet concentrations close to *C**, but as the effluent target approaches *C**, the value of *P* becomes more and more important. If the computation is sensitive to *P*, then it becomes desirable to use cells in series. Use of three cells in series will lead to *P*-values between six and nine, beyond which is a region of diminishing returns.

Site constraints of steeply sloped ground may mandate terraced, multicell design. Because it is not possible to maintain uniform level and shallow depths on steep slopes, the change in elevation is accomplished via drop structures between cells in series. Inman *et al*. (2001; 2003) reported that site slopes of up to 25% called for 22 cells arranged on three separate flow paths at the Clayton County, Georgia, wastewater-polishing wetland facility. That project required the movement of 420,000 $m^3$ of earth over the 22 ha of wetlands or close to 2 m of cut and fill over the site area.

Costs increase as the number of cells increases. The number of cells required in each flow path must be determined by balancing the cost of more cells with the need for high-hydraulic efficiency. The ratio of berm area to wetland surface area increases with more cells. Figure 18.4 shows the design variables associated with increasing cells in series for a hypothetical fixed rectangular project footprint. Compartmentalization increases the *N*TIS for the footprint. The number of transfer structures increases with the number of cells, but the vegetated area becomes smaller due to more berm area and more water distribution zone areas. If the total area were not fixed, the total footprint would increase with compartmentalization. For an extreme example of dual-flow paths and multiple cells, see Figure B.24 in Appendix B for the Orlando Easterly project, which contains 34 structures for 17 cells on two flow paths.

## Internal Cell Arrangement

The arrangements of internal features of a single wetland cell have very large implications for the efficiency of the cell. Both the volumetric efficiency, $e_V$, and the detention time distribution efficiency, $e_{DTD}$, are very sensitive to the vegetation patterns within the cell. It is probably these factors that give rise to the majority of the intersystem scatter in performance data.

Each individual wetland cell may be contoured in a number of different ways. The goals of internal physical design are to provide efficiency of treatment, adequate conveyance of water, and perhaps habitat values. Wetlands have a tendency to channelize from points of inlet to points of outlet. If permitted, this operational feature reduces the gross areal efficiency of the wetland. Control of the bottom elevation and

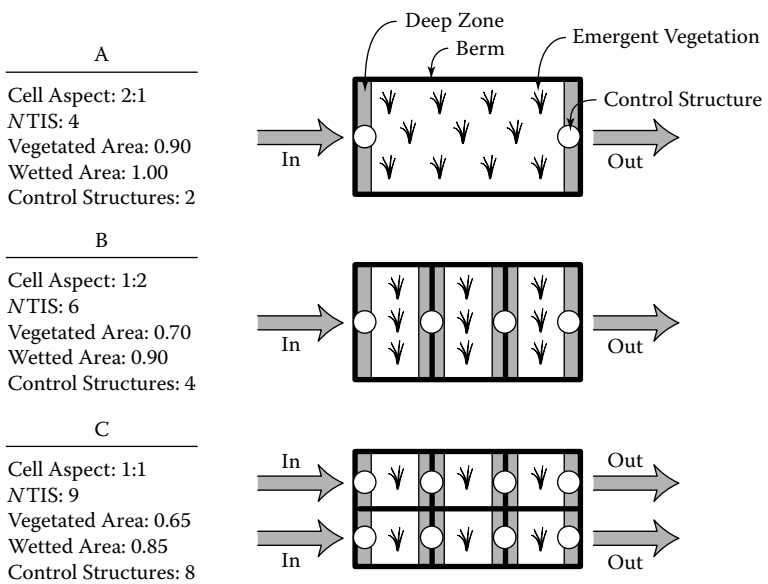



**FIGURE 18.4** Options for cells in series and parallel. *N*TIS are rough estimates. Areas are examples only; actual numbers are size-dependent.

vegetation density can, in principle, prevent poor flow distribution. But in practice, the bottom of the wetland can neither be constructed nor maintained at tolerances that promote full areal contacting. Nevertheless, care must be taken to degrade any preexisting ditches, roads, or berms on the site, because these will exert possibly undesirable flow control in the FWS wetland.

There are no hard and fast rules for design for high-areal efficiency for an individual wetland cell. Some general recommendations are

1. Avoid very small length-to-width ratios.
2. Avoid blind spots in corners.
3. Avoid unvegetated short-circuit paths.
4. Reestablish flow distribution at intermediate points in a flow path.
5. Maintain good bottom uniformity during construction and start-up: minimize the formation of topographic channels parallel to flow.

Figure 18.5 illustrates some of these ideas.

## Aspect Ratio

The length-to-width (aspect) ratio is important in basin design because of its effect on flow distribution and hydraulic short circuiting. Theoretically, a constructed wetland with a high-aspect ratio is not better for treatment than one with a lower aspect ratio, as long as flow is distributed effectively. It has been speculated that long, narrow flow paths are closer to plug flow than short, wide flow paths. But the interior microchannels of a uniform depth, uniformly vegetated FWS wetland have small dimensions; recirculation eddies are limited by depth (on the order of 0.3 m for a FWS wetland) or by the lateral spacing of plant stems or clumps of stems (also a fraction of a meter). Thus, the effective length-to-width ratio is predetermined and widening the wetland only adds more parallel channels.

There is a large literature dealing with the effects of aspect ratio (length-to-width, L:W) on the behavior of inert tracers and on the performance of ponds and wetlands for pollutant reduction as a function of aspect ratio. Thackson *et al*. (1987) presented data that illustrate the importance of L:W on the effective volume ($e_V$) of unvegetated ponds, showing an increase with L:W. Since that time, computational capabilities have permitted theoretical exploration of the values of $e_V$ and $e_{DTD}$ for basins of different shapes and inlet configurations (see Chapter 2 for a discussion of these efficiencies). For instance, Persson *et al*. (1999) investigated the effect of L:W using the two-dimensional, depth-integrated code of MIKE-21 (Warren and Bach, 1992). Jenkins and Greenway (2005) also investigated empty ponds using the two-dimensional, depth-integrated code of TDFLOW (Jenkins and Keller, 1990). Both numerical studies utilized point inlets and point outlets, and found a theoretical increase in $e_{DTD}$ (increase in *N*TIS) with increasing L:W, up to 10–30 TIS for L:W = 15. However, both studies neglect wind mixing, which may be the dominant driving force in open water systems.

Just as importantly, increasing aspect ratio will decrease the inefficiency of utilizing a point inlet or outlet, but there are other steps that may be taken to minimize poor inlet or outlet distribution of flows. A large aspect ratio tends to improve volumetric efficiency, quantified by the ratio of actual to theoretical detention time. In other words, it is not possible to get closer to the plug-flow model, but it is possible to utilize the entire wetland area. For empty ponds, a L:W of over 10 may be needed to minimize the effect of jet flow short-circuiting and corner dead zones (Figure 18.6).

FWS treatment wetland data show that "longer and narrower" does not necessarily mean less effect of mixing, and hence a closer approximation to plug flow. For example, the tracer studies at Richmond, Australia, showed *N*TIS = 4.6 for an open water unit of L:W = 25 and *N*TIS = 5.0 for a *Myriophyllum* FWS wetland of the same aspect (Bavor *et al*., 1988). A number of tracer tests of the wetland cells

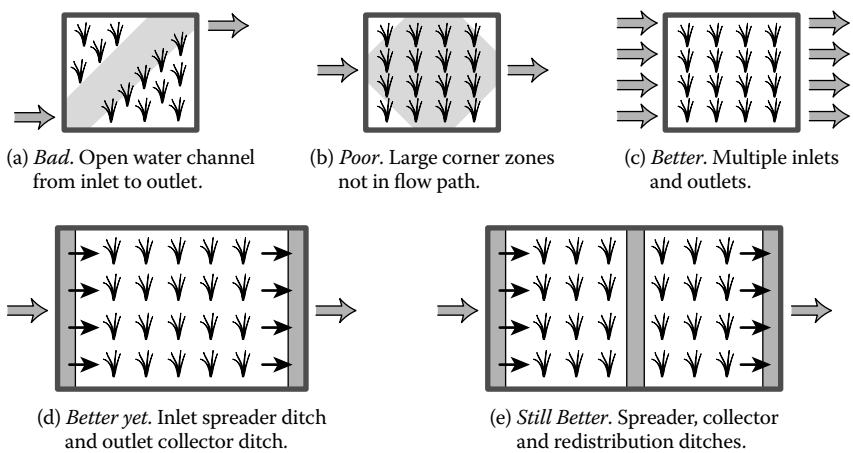



**FIGURE 18.5** Concepts of cell internals: inlet distribution, outlet collection, and internal redistribution. (From Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

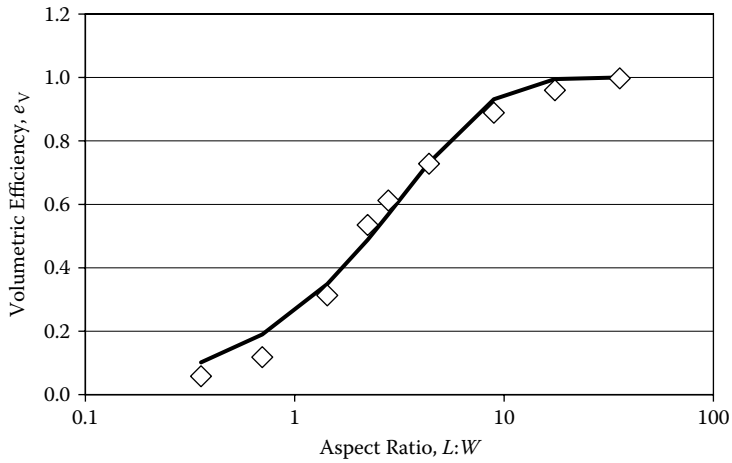



**FIGURE 18.6** The effect of aspect ratio on the volumetric efficiency of unvegetated ponds with point inlets and point outlets. (Simulation data from Jenkins and Greenway (2005) *Ecological Engineering* 25(1): 61–72.)

7B and 9B, both with L:W = 12.5, at the Sacramento, California, project found *N*TIS = 4.7 ± 0.8 and 5.0 ± 1.4, respectively (Nolte and Associates, 1998b). On the other hand, the data from the Champion, Florida, project show *N*TIS = 11 for a FWS wetland with L:W = 10 (Knight *et al*., 1994). Likewise, the South Florida Water Management District (SFWMD) periphyton treatment wetland project found *N*TIS = 9 for L:W = 5 and *N*TIS = 25 for L:W = 45 (CH2M Hill, 2003a). The effect of aspect ratio in these studies is confused by issues of wind, vegetation patterns, and deep zones. Thus, it is dangerous to assume that aspect ratio alone can provide the desired improvements in efficiency.

The literature contains a few "recommendations" for L:W ratios to be used in design. For instance, Crites *et al*. (2006) recommend 2:1 < L:W < 4:1. U.S. EPA (2000a) comments that, in general, FWS wetlands are built with L:W > 1:1. These appear to be reasonable suggestions that go with the recognition that lower *system* aspect ratios may be used provided cells in parallel are used. However, there is not much quantifiable treatment performance motivation to prefer one aspect ratio over another, as long as the design stays in a reasonable range, such as 2 < L:W < 10. However, there are sometimes limitations on aspect ratio due to the hydraulic profile (see Chapter 2 and subsequent sections of this chapter).

If one looks beyond the volumetric improvements, data from FWS wetland studies indicate somewhat better performance for higher length-to-width ratios (Herskowitz, 1986; Knight *et al*., 1994; CH2M Hill, 2003a), but the margin is not large when the pollutant reductions are low-to-moderate (0–50%).

Higher aspect ratios increase the area of external berms that must be constructed to enclose a given wetland area. Therefore, economics may argue for low L:W ratios (Knight, 1987). Other methods for maintaining effective flow distribution, such as deep zones, should be considered as alternatives to reduce the need for higher length-to-width ratios.

## Sinuousity

One of those alternatives is to use smaller internal divider berms to create a sinuous flow path of high-aspect ratio (Figure 18.7). An extreme example of this design is the Tucush, Peru, wetland with 14 cross-passes in each of two flow paths (see Figure 16.1). The interior berms create longer flow paths at the expense of cost and area. Not only do the berms subtract from the wetted area but the frequent flow reversals invite dead zones in the corners of the switchbacks. These corner zones may be minimized by orienting the flow in the long dimension of the cell.

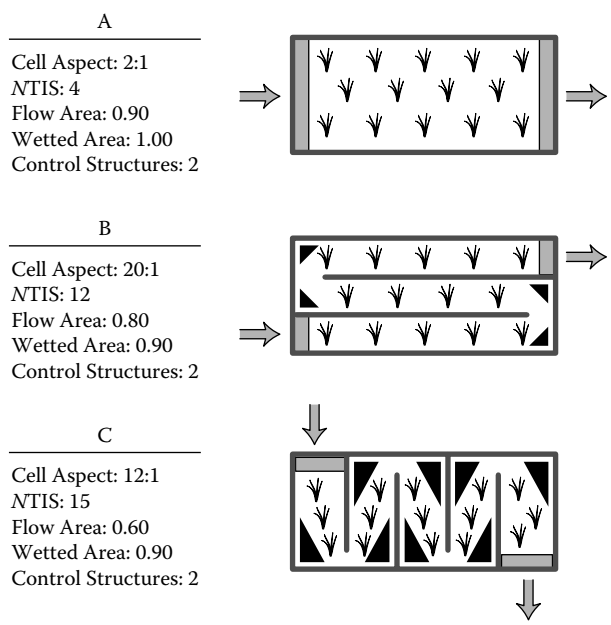



**FIGURE 18.7** Options for interior berms creating sinuosity. *N*TIS are rough estimates. Areas are examples only; actual numbers are size-dependent.

Computational fluid mechanics, in conjunction with first-order reaction rate kinetics, was used by Abbas *et al*. (2006) to simulate the effect of baffles creating sinuosity in ponds of different aspect ratios. These calculations were vertically averaged, and thus might be considered representative of unvegetated wetlands as well. Higher length-to-width was effective in promoting biochemical oxygen demand (BOD) reduction, but more internal baffles were yet more effective. Ratios 1:1 < L:W < 100:1 were investigated for point inlets and outlets. The swing in BOD reductions was from 10 to 95%.

Field testing of pathogen reduction was done in a side-by-side test at Lidsey, United Kingdom, utilizing baffled and unbaffled ponds, 1 m deep, with point inlets and outlets (Bracho *et al*., 2006). The unbaffled pond had L:W = 9:1; the adjacent pond of the same shape and size had three longitudinal baffles, creating L:W = 79:1. Tracer testing indicated $N$TIS = 2.1 and $e_V = 0.33$ for the unbaffled pond, and $N$TIS = 7.3 and $e_V = 0.38$ for the baffled pond. The low efficiencies were attributed to "jet flow short-circuiting." The reduction of fecal coliforms was 82.1% in the unbaffled pond and 99.4% in the baffled pond.

Interestingly, the use of divider berms to create back-and-forth sinuous flow paths has been found to provide low-volumetric efficiency while showing moderate-to-high $N$TIS values in some systems. Tracer tests at the sinuous Hillsdale, Michigan, site showed $N$TIS = 4.3 and 7.4, and $e_V = 0.33$ and 0.51, and for the east cell of L:W ≈ 16 (four cross-passes in a 1:1 footprint); and $N$TIS = 37, and $e_V = 0.34$, and for the west cell of L:W ≈ 18 (five cross-passes in a 1.6:1 footprint). The reason was found to be the fact that the flow path became channelized (deeper) in the center portion of the sinuous path, leaving excluded sides and corners.

A side-by-side comparison of sinuous and nonsinuous cells for phosphorus removal was conducted by the SFWMD (Figure 18.8) (CH2M Hill, 2003a). The hydraulics of the sinuous cell were superior with $N$TIS = 25 versus $N$TIS = 9 for the nonsinuous cell. However, the volumetric efficiency was only 58% versus 88% for the nonsinuous cell. The phosphorus performance of the sinuous cell was superior to that of the nonsinuous cell with higher $k$-values.

Divider berms are very prone to cross-berm leakage, which may create bad short-circuits. This was the case at the SFWMD project. Also, the divider berms occupy some measurable fraction of the total footprint, which detracts from overall areal effectiveness. Results to date indicate that internal sinuosity may have little net effect.

## Flow Distribution

Both field results and computational studies have shown that point inlets and point outlets are not a good idea. Pond studies are not a good model in this regard, because inlets to ponds are very frequently point inlets. The corner zones of the pond are often not easily accessed by the incoming water. However, the presence of vegetation in FWS wetlands can lead to even more difficulties for water to reach corner zones. Computational fluid mechanics suggests that if the water is distributed uniformly from side-to-side of the wetland inlet, rather than at a point, there will be a significant improvement of the hydraulics (Persson *et al*., 1999). The changed distribution is forecast to increase $N$TIS from 1.5 to 6.7, and $e_V$ from 79 to 89% for a basin of L:W = 2:1. Wörman and Kronnäs (2005) computed that as the width of the water inlet increases relative to the wetland width, both $e_V$ and $N$TIS should increase.

Stairs (1993) experimentally determined the effect of inlet cross-spreading by performing tracer tests both before and after the removal of a perforated pipe spreader. With the spreader, $N$TIS = 4.47 and $e_V = 83.6\%$. Without the spreader, for a point inlet, $N$TIS = 3.27 and $e_V = 66.5\%$. Thus, the theoretical results of simulations have been verified in the field.

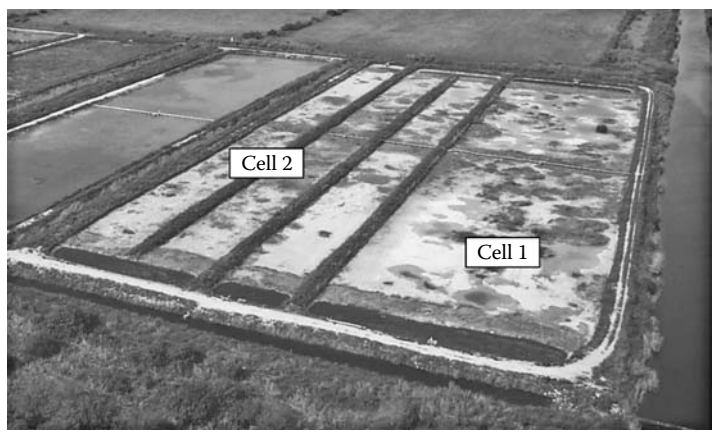



**FIGURE 18.8** Cells 1 and 2 of the South Florida Water Management District periphyton treatment wetland project. Cell 1 had an aspect ratio of 5:1, whereas Cell 2 had L:W = 45:1, because of a three-pass sinuous design. Both were two hectares in size and are shown empty in this photo. Cell 1 tracer tested at $N$TIS = 9, whereas Cell 2 tested at $N$TIS = 25. (Photo courtesy CH2M Hill.)

The collection of the water at the wetland outlet poses the same problems of corner dead zones. Therefore, collection across the wetland width is advantageous.

### *Bathymetry: Deep Zones and Speed Bumps*

Wetlands have a tendency to channelize from points of inlet to points of outlet. If permitted, this operational feature reduces the gross areal efficiency of the wetland (see Figure 4.21, for instance). Control of the bottom elevation and vegetation density can, in principle, prevent poor flow distribution. But in practice, the bottom of the wetland can neither be constructed nor maintained at tolerances that promote full areal contacting. Bottoms do not start completely level, and a number of processes contribute to the formation of local highs and lows such as soil heaving and sediment deposition. Nevertheless, care must be taken to degrade any preexisting ditches, roads, or berms on the site because these will exert possibly undesirable flow control in the FWS wetland. Dierberg *et al*. (2005) demonstrated the poorer treatment that occurs along such channelized short circuits.

Compartmentalization involves emergent berms with transfer structures, but such cross-berms may also be constructed so as to be always or occasionally under water. For example, the treatment wetland at Bulwer Island, Australia, was constructed with a repeated pattern of $0.4 \rightarrow 0.2 \rightarrow 1.0$ m depths along the flow path (Simi and Mitchell, 1999). The intent of this design was to break up longitudinal channeling and to promote different microenvironments. Underwater cross berms have also been placed in some of the stormwater treatment areas in South Florida. It is not known if these have positive effects on treatment. The use of cross-benches was forecasted to promote good hydraulics, *N*TIS = 23 and $e_V$ = 0.80, by Persson *et al*. (1999).

Deep zones in surface flow constructed wetlands are supposed to serve several purposes (Knight and Iverson, 1990). These deeper areas extend below the bottom of the vegetated basin areas by at least 1 m to exclude the development of rooted macrophytes (Figures 18.9 and 18.10). Such unvegetated cross ditches provide a low-resistance path for water to move laterally and provide a nearly constant head across the wetland. They also provide for extra detention time, but in a deep water zone. Such ditches often become covered with duckweed (*Lemna* spp.) and can be used by wetland birds and fish as reliable habitat. These redistribution ditches change the potential for short-circuiting within the wetland, because high-speed rivulets are intercepted and mixed with quiescent water in the deep zone. However, the redistribution ditch adds a potential for wind mixing that compensates the reduced short circuiting. But water is more effectively distributed over the wetland, improving the gross areal efficiency to some extent. Lightbody *et al*. (2007) provided evidence from laboratory models and computations that indicated improvement due to deep zones at high degrees of removal. However, that improvement was contingent on the assumption that serious short-circuiting occurred in the vegetated sections of the system.

Knight *et al*. (1994) operated a set of six treatment wetlands receiving treated paper-pulp mill effluent for two years. Two pairs of these received the same influent water, one of each pair containing two internal deep zones comprising 25% and 45% of the area (Table 18.1). There was variability in flow (different HLRs) and aspect ratios (2.5 and 10) between the pairs. Tracer data from that project were here reanalyzed and characterized by a tanks-in-series (TIS) detention time distribution. The effect of deep zones was to decrease the number of TIS (Table 18.1). And, for the larger cell pair (A and B), the volumetric efficiency was improved by deep-zone addition. The load reductions for all contaminants were increased by the inclusion of deep zones but by only a very small margin for the higher aspect ratio pair (E and F). One

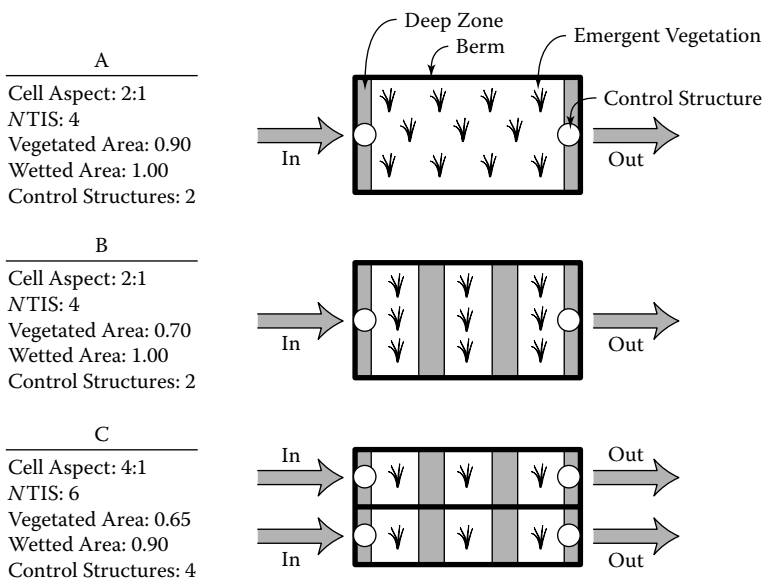



**FIGURE 18.9** Options for deep zones in series and parallel. *N*TIS are rough estimates. Areas are examples only; actual numbers are size-dependent.



**FIGURE 18.10** The Saginaw, Michigan, treatment wetland system contains two flow paths, each bifurcated by one cross-deep zone.

**TABLE 18.1**
**Performance of the Champion Wetlands, July 1991–June 1993**

| Wetland | A | B | E | F |
|---|---|---|---|---|
| Area ($m^2$) | 4,048 | 4,048 | 1,012 | 1,012 |
| Aspect | 2.47 | 2.47 | 9.88 | 9.88 |
| DZ% | 5 | 25 | 5 | 45 |
| Mean depth (m) | 0.27 | 0.38 | 0.27 | 0.58 |
| HLR (cm/d) | 3.16 | 3.38 | 7.82 | 8.13 |
| Nominal HRT (d) | 8.54 | 11.23 | 3.45 | 7.13 |
| *N*TIS | 4.6 | 3.6 | 10.7 | 2.0 |
| Volume efficiency | 0.63 | 0.81 | 0.91 | 0.74 |
| Inlet TN (mg/L) | 9.1 | 9.1 | 9.1 | 9.1 |
| Outlet TN (mg/L) | 3.5 | 2.5 | 4.7 | 4.7 |
| TN LR ($g/m^2 \cdot yr$) | 65 | 81 | 126 | 131 |
| Inlet $NH_4$-N (mg/L) | 3.4 | 3.4 | 3.4 | 3.4 |
| Outlet $NH_4$-N (mg/L) | 0.7 | 0.3 | 2.0 | 2.0 |
| $NH_4$-N LR ($g/m^2 \cdot yr$) | 31 | 38 | 40 | 42 |
| Inlet $NO_x$-N (mg/L) | 0.90 | 0.90 | 0.90 | 0.9 |
| Outlet $NO_x$-N (mg/L) | 0.13 | 0.11 | 0.13 | 0.12 |
| $NO_x$-N LR ($g/m^2 \cdot yr$) | 9 | 10 | 22 | 23 |
| Inlet BOD (mg/L) | 21 | 21 | 21 | 21 |
| Outlet BOD (mg/L) | 10 | 7 | 13 | 13 |
| BOD LR ($g/m^2 \cdot yr$) | 127 | 173 | 228 | 237 |
| Inlet TSS (mg/L) | 58.6 | 58.6 | 58.6 | 58.6 |
| Outlet TSS (mg/L) | 10.1 | 6.9 | 12.9 | 13.6 |
| TSS LR ($g/m^2 \cdot yr$) | 559 | 638 | 1304 | 1335 |
| Inlet TP (mg/L) | 0.97 | 0.97 | 0.97 | 0.97 |
| Outlet TP (mg/L) | 0.37 | 0.28 | 0.58 | 0.59 |
| TP LR ($g/m^2 \cdot yr$) | 6.9 | 8.5 | 11.1 | 11.3 |

*Source:* Hydraulic and concentration data are from Knight *et al.* (1994) *TAPPI Journal* 77(5): 240–245; the DTD analyses are from this work.

interpretation of these results is that the net effect of better efficiency and worse *N*TIS results from interception of short circuits in the wide cell but that these are less important for the high-aspect cells. Thus, wide cells profit from deep zones but narrow cells do not. Another important feature of these results was the fact that the added volume provided by deep zones resulted in smaller volumetric rate constants, an observation in agreement with similar results detailed in Part I.

The artifact of L:W was not present in a study of 12 research wetlands operated under varying conditions at a site west of the city of Phoenix, Arizona (Kadlec, 2006a). These were constructed as a triplicated design with zero, one, two, and three internal deep zones, all containing an inlet distribution and an outlet collection deep zone, together comprising 12.5–35% of the wetland areas. There were no differences in the internal hydraulics with deep-zone numbers. Deep-zone numbers in the wetlands did not affect nitrogen treatment performance. No differences with deep-zone numbers were found for temperature, dissolved oxygen, pH, or nitrogen removals or rate constants. In this study, there was no large treatment benefit or detriment of incorporating internal deep zones in free water surface wetlands.

Moore and Niswander (1997) operated a set of six treatment wetlands receiving diluted dairy wastewater for two years. Two wetlands had a central internal deep zone comprising 45% of the area, whereas the other four had no internal deep zone. The authors concluded that deep-center sections did not show any significant impact on treatment efficiency. Data for the second year, past the startup period, are shown in Table 18.2, as reported in NADB v.2 (NADB database, 1998). None of the outlet concentrations are different at the 5% level. The wetland areas were all identical and, hence, the hydraulic loading to all six wetlands was uniform at 3.95 cm/d. For a presumed 4 TIS hydraulic pattern, the areal *k*-values do not differ at the 5% level. For instance, for total Kjeldahl nitrogen (TKN), $k = 10.1 \pm 1.1$ m/yr compared to

**TABLE 18.2**
**Oregon State University Pilot Wetland Performance, Means, and Standard Deviations among Wetlands**

| Parameter | Inlet | Outlet (N = 4) No IDZ | Outlet (N = 2) One IDZ |
|---|---|---|---|
| pH | 7.50 | 7.11 | 7.25 |
| | | 0.05 | 0.03 |
| DO | 2.35 | 0.15 | 0.19 |
| | | 0.12 | 0.06 |
| $NH_4$-N | 148 | 84 | 78 |
| | | 4 | 6 |
| TKN | 204 | 107 | 102 |
| | | 7 | 6 |
| TSS | 653 | 195 | 202 |
| | | 22 | 6 |
| BOD | 888 | 307 | 267 |
| | | 39 | 7 |
| TP | 36.5 | 19.6 | 18.4 |
| | | 1.1 | 1.5 |

*Note:* IDZ = inlet deep zone; data are averages of biweekly data, January 1994–February 1995.

*Source:* Data from NADB database (1998) *North American Treatment Wetland Database (NADB), Version 2.0*. Compiled by CH2M Hill. Gainesville, Florida.

$k$ = 11.0 ± 1.1 m/yr for the two cells with deep zones. However, the inclusion of the deep zone added to the water volume, and increased the average depth from 30 to 75 cm, and consequently increasing the nominal detention time from 7.6 to 19 days. As a consequence, the $k_V$ values for TKN were different at the 5% level, 0.093 ± 0.01 $d^{-1}$ for no deep zones versus 0.040 ± 0.004 $d^{-1}$ for deep zones. As for the Champion study, these Oregon State University results suggest that the extra depth added by deep zones carries a penalty in volumetric performance.

Importantly, all three studies demonstrate a penalty for deep zones in terms of lower *volumetric* rate constants for systems with deep zones. The presumption of constant $k_V$, which is reiterated in much of the treatment wetland literature (Crites and Tchobanoglous, 1998; U.S. EPA, 1999; U.S. EPA, 2000a; Water Environment Federation, 2001), is, therefore, inappropriate for design of FWS wetlands with deep zones. This further substantiates the growing body of knowledge that wetland areal models are to be preferred over volumetric models for FWS wetlands, in general. For deep zones, the added water volume and its attendant-added detention time do not contribute proportionately to pollutant removal.

Taken together, these studies do not demonstrate a clear advantage or a clear disadvantage for deep zones. Other wetland attributes, such as aspect ratio, vegetation community type, and compartmentalization, may be equally or more important for water quality improvement. Nevertheless, deep zones may be important for habitat value, may provide for increased reaeration, and may provide opportunity for pathogen reduction via UV radiation. The amount of open water that may be included in a cell remains a controversial issue. Based upon anecdotal evidence, U.S. EPA (2000a) contends that the central part of a flow path must be open water/SAV to provide oxygenation for ammonia reduction. However, Kadlec (2005e) found no particular support for that view in an examination of data from many systems, including ponds as the open water extreme. Open water sections in a cell provide better habitat value than does a uniform dense stand of emergent plants (Weller, 1978). However, habitat values must be balanced by the potential for wildlife to interfere with treatment. Excessive waterfowl use can add pathogens to the water. Muskrats, nutria, and beavers want the deep water for refuge and can seriously impair the vegetation, hydraulics, and possibly treatment (Kadlec *et al*., 2007).

An important function of an inlet deep zone is the settling of particulate matter that might otherwise create difficulties in the inlets of vegetated sections. This has been discussed in Chapter 7 but deserves consideration here as part of the process of developing a layout for the treatment wetland. It is useful to remove large sediment loads before they reach the vegetation, so that the eventual cleanout is from a pond rather than from vegetation. The Imperial, California, system employs a 3.9-ha, two-cell sedimentation basin before wetlands (Figure 18.11). Over a four-year operating period, the basins trapped 3,820 tons of sediments, amounting to approximately 10 cm of accretion. Those solids would have been detrimental to the inlet zone of the 4.7-ha follow-on wetlands. Table 7.8 indicates that it is easily possible to settle river-borne solids in just a few days detention in a presettling basin with removals of over 90%. The sizing of this inlet element of the treatment train is dependent upon the settling characteristics of the specific solids in question, as well as the accumulation rate of the solids. An inlet deep zone built for flow distribution may fill up quickly and require emptying at frequent intervals (see Figure 7.13). A simple mass balance suffices for the sizing of sedimentation basin storage for the desired loading and emptying frequency, whereas a settling calculation determines the area needed for a given basin-operating depth.

### *Fringing, Banded, and Clumped Vegetation*

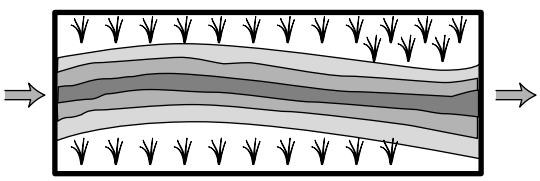
Many natural wetlands are riverine, meaning that the wetland contains a central stream channel, which may meander through a zone of emergent vegetation. Indeed, this wetland form is so common that it is mistakenly presumed to be a good prototype for a treatment wetland. Nothing could be further from the truth. A central channel will convey most of the water, and the fringing vegetation will effectively receive flow only on rising flood conditions and discharge it on the ebbing stage. Therefore, configurations similar to Figure 18.12A are usually to be avoided. Simply stated, if the water needs to go through the plants, do not plant them off to the side of the flow path. Jenkins and Greenway (2005)



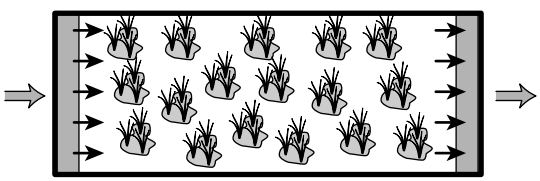
**FIGURE 18.11** Aerial view of the Imperial, California, treatment wetland. Two sedimentation basins, in parallel, precede the four wetland cells. The first cell is built sinuous with fringing vegetation. The second is built with cross-banded vegetation. The final two cells are out of the view to the upper right.

provide numerical calculation results for several aspect ratios and cover percentages, which show that hydraulic efficiencies are low for fringing vegetation.

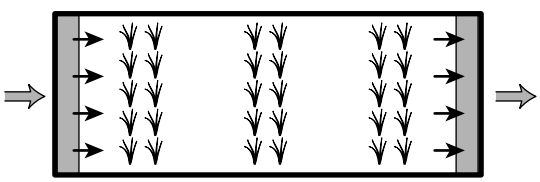
The Jenkins and Greenway (2005) computational research also demonstrated that cross-banded vegetation is a better option for hydraulic efficiency than fringing vegetation. Therefore, any open water that may be included in the cell should be oriented across the flow direction, whether or not it is deeper than the average. However, as will be discussed in the following text, adequate conveyance may be difficult to maintain if large aspect ratios are combined with significant banded or uniform vegetation.



(a) *Bad*. Bench and channel design provides fringing vegetation, dead zones and a short circuit due to topography.



(b) *Better*. Inlet distribution, outlet collection, and hummocks.



(c) *Still Better*. Inlet distribution, outlet collection, and banded vegetation.

**FIGURE 18.12** Concepts of cell internal vegetation patterns. Fringing, patterned, and banded vegetation may be used.

These concepts were utilized in the Brawley, California, treatment wetland (Figure 18.13). The first cell was built with a sinuous path and fringing vegetation; the second cell was built with banded vegetation. Only overall system data were taken, thus no information on relative efficiencies is available.

A third concept for the vegetation pattern is the use of small islands or hummocks, either emergent or submerged (Thullen *et al*., 2005). At the present time, there is no information on the hydraulic efficiency of this alternative or any pollutant removal comparison.

## Fitting the Wetlands to the Site

Given the total required wetland area and the concepts of system configuration, there still remains the placement of the wetland on the site. The principal considerations are adaptation to the boundaries and contours of the site, minimization of intercell conveyance, and minimization of earthmoving.

Site boundaries often determine the external shape of the overall system, because there is often no extra land or the ability to choose the shape of the available land. In that event, the various pieces of the overall system must conform to the space available. The topology of the conceptual layout is retained, but shapes and perhaps areas are sacrificed. A good example is the initial layout of the Columbia, Missouri, treatment wetlands (Brunner *et al*., 1992; Brunner and Kadlec, 1993). The conceptual configuration was selected to be three banks of cells in parallel. The available lands were bounded by streams, roads, towns, railroads, and the floodway of the Missouri River. As a consequence, the actual



**FIGURE 18.13** Aerial view of the Brawley, California, treatment wetland. The first cell of two in series is built in a hairpin, sinuous with fringing vegetation. The second is also a hairpin but with cross-banded vegetation.

layout was not completely rectangular nor were the cells all in close proximity (Figure 18.14). Despite the long-transfer distances, this complex of cells is gravity driven. A transfer pump is provided at the system outlet, to supply the treated water to the Eagle Bluffs wildlife wetlands, where it is used to foster habitat. A fourth bank of wetland cells was added to the system in 2003.

## 18.2 HYDRAULICS

Linear velocity can become a consideration in design for very large wetlands. For instance, total suspended solids (TSS) removal depends on sedimentation and trapping within the wetland. Excessive linear velocities lead to large values of shear stress on deposited solids, and therefore can lead to

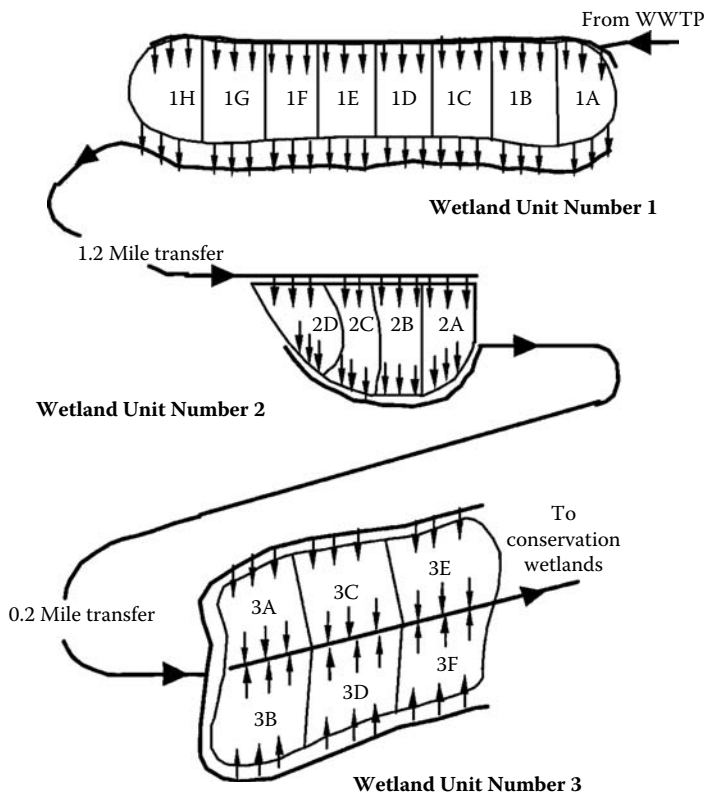



**FIGURE 18.14** Arrangement of cells within the initial configuration of the Columbia, Missouri, wetland treatment units. All cells receive water via submerged, gated distribution pipes along the inlet edge; each cell has three outlet structures. (From Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

the potential for resuspension of those solids. The critical shear stress for resuspension has not been determined for any treatment wetland, as discussed in Chapter 7. As conservative interim design criteria, it is recommended that linear velocity be kept below a value which would resuspend fine particles. The minimum velocity for resuspension has been suggested to be about 20 cm/s (U.S. Army Corps of Engineers, 2000) (Figure 18.15). However, it is well to bear in mind that existing FWS wetlands operate at velocities lower than this, mostly below 100 m/d, and, hence, there is no field test of the criterion. The large Florida phosphorus control wetlands have all been subjected to a maximum velocity criterion of 3.0 cm/s (2,600 m/d).

Theory indicates that higher water linear velocities may favor improved pollutant removal because of improved mass transfer (Kadlec and Walker, 2004). In stagnant water, pollutant transport to removal sites is by molecular diffusion, whereas in turbulent flow, eddies carry materials to solid surfaces.

As the length-to-width ratio is increased, both the linear velocity and the head loss increase. At some point, the head loss will produce an inlet water depth that is unacceptably large, as detailed in Chapter 2. A hydraulic analysis would usually be used to check whether an aspect ratio, set from site considerations and constraints, produces an acceptable hydraulic profile.

## Hydraulic Profiles

The primary purpose of calculating the expected hydraulic profile is to determine if the proposed wetland can pass the design flow without an excessive "pile-up" of water at the inlet. The water depths for any particular cell configuration and flow may be calculated using known procedures of varying complexity. Complicated, two-dimensional calculations may be required for complex wetland geometries. Examples of readily obtainable computer codes include the suite of Hydraulic Engineering Center (HEC) models (Feldman, 1981). However, simpler one-dimensional spreadsheet calculations often suffice to determine the approximate shape of the water surface profile through a FWS wetland cell along the flow direction. The water balance and friction equations combine to give the equation for depth (see Chapter 2; also Kadlec, 2003b):

$$aWh^b\left(-\frac{d(h+B)}{dx}\right)^c = Q \tag{2.23}$$

$$H_{\text{at } x=L} = \mathrm{H}_L \tag{2.24}$$

where

$a, b, c$ = constants
$B$ = elevation of the bed bottom above datum, m
$h$ = water depth, m
$H$ = water depth, m
$L$ = wetland length, m
$Q$ = volumetric flow rate, $m^3/d$
$x$ = distance from wetland inlet, m
$W$ = wetland width, m

Here it is assumed that the bottom is uniform across the flow direction, but possibly sloped, and a power law describes vegetative resistance to flow. The coefficients $a$, $b$, and $c$ are selected to be representative of the type and density of the vegetation (see Chapter 2). The usual specification for the outlet depth is determined as a weir setting plus the height over that weir, but other devices may also result in an outlet depth specification. Because the depth is known (set) at the outlet, the integration of Equation 2.24 proceeds from the outlet upstream to the inlet, thus resulting in the designation as a backwater calculation.

The wetland-sizing procedure will have determined a hydraulic loading for the design, $q = Q/LW$, where $Q$ is the design flow. The wetland width is set from the selected aspect

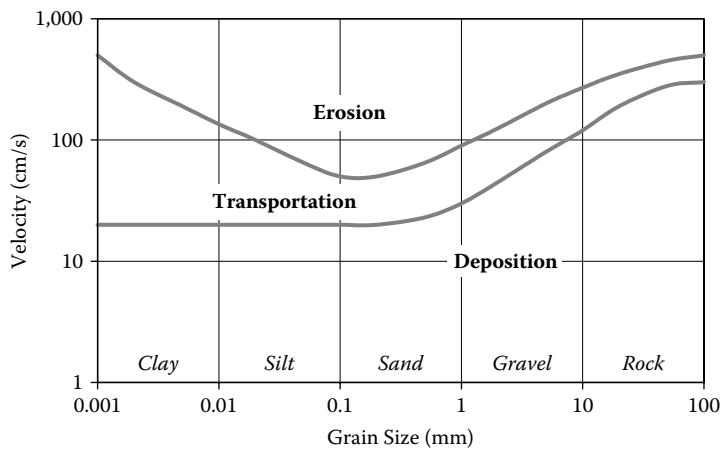



**FIGURE 18.15** Relationship between grain size, velocity, and sediment movement. At low velocities, deposition is dominant, whereas at higher velocities, resuspension and transport may occur. (From Hayes *et al*. (2000) *Wetlands Engineering Handbook*. ERDC/EL TP-WRP-RE21, U.S. Army Corps of Engineers: Washington, D.C.)

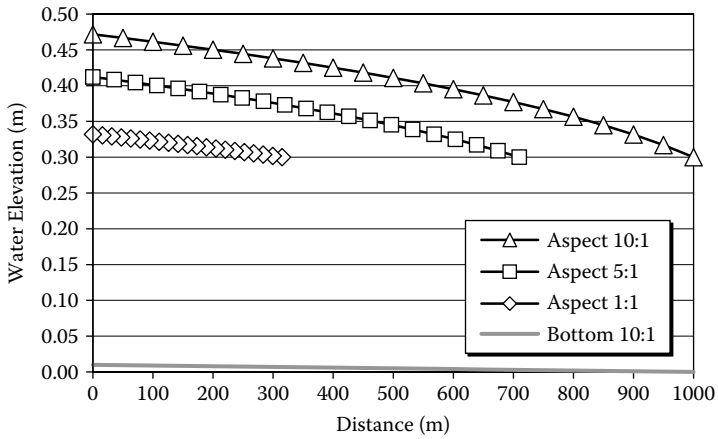



**FIGURE 18.16** Wetland hydraulic profiles for aspect ratios of L/W = 1:1, 5:1, and 10:1. The height over the weir is set at 0.3 m (left-hand side). Bottom has been set as nearly level. The design hydraulic loading is 10 cm/d. Note that water surface is not a level pool nor is it parallel to the bottom.

ratio and the design area. The backwater calculation then determines the hydraulic profile, $h(x)$.

As an illustration, consider a wetland designed for a hydraulic loading of 10 cm/d (10,000 $m^3$/d on 10 ha). Different aspect ratios may be chosen in recognition of site constraints and the desire for high-hydraulic efficiency provided by high-aspect ratio. For this illustration, three ratios are selected: 1:1 (316 m × 316 m), 5:1 (710 m × 142 m), and 10:1 (1,000 m × 100 m). The exit weir is set for a depth of 30 cm. For the three cases, the inlet depths for dense vegetation are 33, 41, and 46 cm, respectively (Figure 18.16). These are all probably acceptable hydraulic profiles.

A field example of an unacceptable head loss situation occurred for the Imperial, California, wetland (see Figure 18.11). This system is comprised of four cells with a combined aspect ratio of 30:1 (TTI and WMS, 2006). In 2003, the hydraulic-loading rate to the Imperial wetland was increased to determine the impact it would have on wetland performance. However, during this period, the water overtopped the inlet levees, and thus loading had to be reduced. However, the reduced hydraulic-loading rates used after the overtopping event were still well above the original design flow rate at which the wetland was operated throughout 2001 and most of 2002. The levees overtopped because physical constraints made it impossible for water to pass through the wetland at the rate it was being applied. This demonstrates that when considering different wetland designs, it is important to take into account the fact that the physical characteristics of the wetland may constrain the hydraulic-loading rate.

## 18.3 EARTHMOVING: DIKES, BERMS, AND LEVEES

### Berm Design

Berms are designed based on hydraulic and geotechnical considerations. The purpose of berms is to regulate and contain water within specific flow paths. Figures 18.17 and 18.18 show typical design features of constructed wetland berms for lined and unlined systems. Exterior wetland berms are kept as small as possible while still providing adequate containment and freeboard to prevent unauthorized flow releases. Interior berms may be used to augment flow distribution but do not have to be designed to control offsite water releases.

Exterior berm freeboard should be adequate to prevent overtopping during sudden storm events (based on a storm event frequency of 10, 25, or more years) and allow overflow of less frequent storm events through controlled and protected emergency overflow points. Berm freeboard should also consider that the wetland will gradually fill with vegetation and with mineral and organic sediments that increase flow resistance and decrease freeboard during system life. Berm height should equal the sum of the maximum desired normal water level (for example, 45 cm), the return storm rainfall amount (for example, 20 cm for a 25-year storm event), and the lifetime loss of freeboard due to sediment and plant accumulation (approximately 1 cm/yr in some wetlands receiving municipal wastewaters). For a 20-year life, this hypothetical wetland should have an emergency spillway height of at least 85 cm with an additional 20 cm or so of berm above that level. Any additional berm height provides additional system life and insures against unauthorized discharges. In the event that an open water system is contemplated, the freeboard should also be adequate to contain wind setup and wave action. The foregoing example provides 1 m of freeboard, which would be sufficient to control wind set and waves for an unobstructed fetch of about 1 km (U.S. Army Corps of Engineers, 2000).

Berms should be constructed on the basis of standard geotechnical considerations. The materials that are available dictate how berms will be designed and constructed. Surface liners or internal clay plugs may be required to minimize berm seepage if sandy or other permeable materials are used for berm construction. External seepage collection channels may be necessary if soils are unconsolidated. An exterior slurry

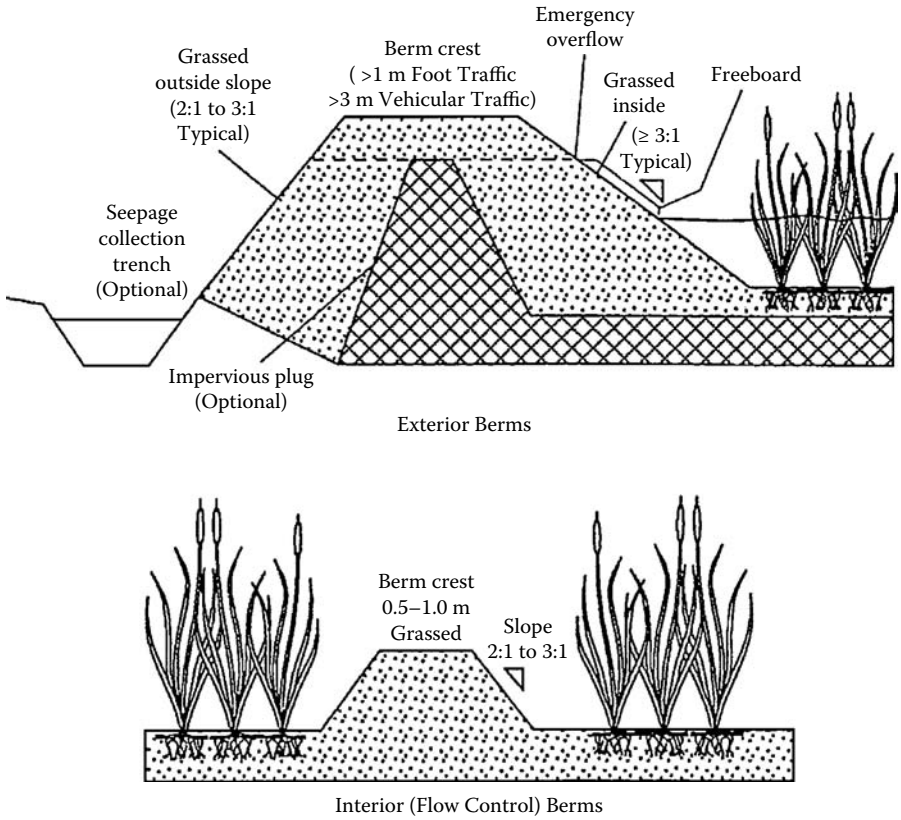



**FIGURE 18.17** Design considerations for constructed wetland berms. (From Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

wall tied into deeper, low-permeability sediments can be used to limit offsite infiltration.

Berm slope is dictated by geotechnical considerations and a slope-stability analysis. Minimum berm slopes typically used in constructed wetlands are 2:1 (horizontal:vertical), and slopes up to 10:1 or 20:1 are used when a shallow littoral shelf is desired to create habitat diversity. Berm width should be adequate for the intended use. For example, if the berm will be used for vehicular traffic, it should be more than 3 m in width.

Dikes are used for access by walking or driving. A vehicle access dike needs to be more than 3 m wide at the top; interior divider berms designed for pedestrian access may be as narrow as 1 m. Dikes greater than about 5 m in width are less likely to be fully penetrated by muskrats. The side slopes of these are typically at a three-to-one slope and may be rip-rapped with stone to prevent erosion or rodent burrowing (Figure 18.19). The interior of a containment dike usually is designed to contain a compacted core and a layer of sealant, which extends above water level. Water containment dikes are subject to local dam safety regulations.

## Basin Bottom Contouring: Cut and Fill

The constructed wetland complex of cells and dikes requires earthmoving. The cost for this activity consists of two distinct

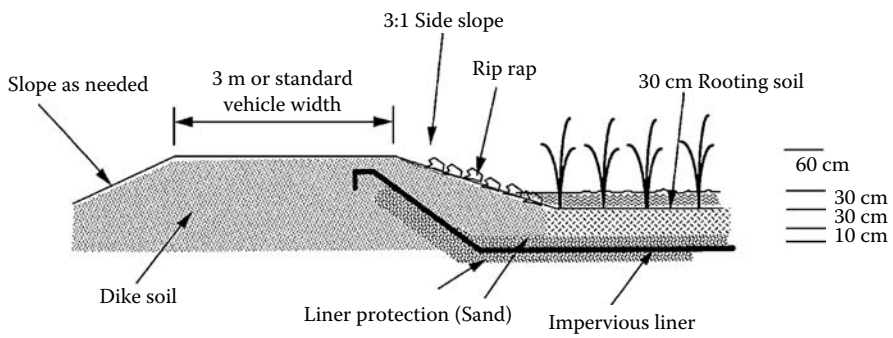



**FIGURE 18.18** Typical constructed wetland exterior berm with a liner.



**FIGURE 18.19** Wetland berm protection using stone riprap at the Lake Nebagamon, Wisconsin, treatment wetland. This divider berm is not drivable. Photo shows wetland just after planting.

components: the cost for moving soils within the site boundaries (cut and fill) and the costs for importing or exporting material from the site, from external borrow sources or to spoil disposal sites. Equal cut and fill is the preferred option because it avoids import/export costs. Selection of the bottom elevation of the cells, together with proper positioning on the site with respect to its topography, generally allows balancing of cut and fill.

Rough estimates of the internal volume of a wetland cell may be made based upon the intended geometry. For the common rectangular geometry, the excavated volume may be calculated from:

$$V_B = \frac{h_E}{6}\left[LW + (L - 2sh_E)(W - 2sh_E) + 4(L - sh_E)(W - sh_E)\right] \quad (18.1)$$

where

$h_E$ = excavated depth, m
$L$ = wetland length at grade, m
$s$ = side slope, m/m
$V_B$ = excavated volume, $m^3$
$W$ = wetland width at grade, m

The volume of earth needed for the levees for one cell, neglecting the corners, is

$$V_L \approx \left[2(L + W_T + 2sh_L) + 2(W + W_T + 2sh_L)\right](W_T h_L + sh_L^2) \quad (18.2)$$

where

$h_L$ = height of levee above grade, m
$V_L$ = levee volume, $m^3$
$W_T$ = levee top width, m

The overall depth of the created basin is

$$h = h_L + h_E \quad (18.3)$$

This overall depth of basin is required to meet the freeboard and water depth requirements of the design. As an example, suppose that the length and width at grade are 100 and 50 m, respectively, and the total depth is to be 1.5 m. The levee is set at a top width of 3 m and side slopes of 3:1. Equations 18.2 and 18.3 balance at a cut depth of 0.45 m with 1.05 m of levee above grade. For many projects, there is enough importance of earthmoving to warrant the use of one of the several available software packages that allow very accurate balancing of cut and fill over variable topography.

### Liners and Rooting Media

The bottom of the wetland, as well as the core of containment dikes, may be formed of compacted clays or bentonite. Locally available clays are preferred from the standpoint of cost reduction. Plastic liners may be feasible for smaller size wetlands. This clay layer or other sealant should not be penetrated by plant roots, so that it retains its integrity. The topsoil from the site should be stockpiled and replaced within the wetland to form a rooting medium. The roots and rhizomes of emergent macrophytes now in use—cattails, bulrushes, and *Phragmites*—usually occupy the top 30–40 cm of the soil column. Therefore, a layer of that thickness should be used. The original topsoil from the site may be useable; and if so, it should be stockpiled separately from the other soils during construction. This topsoil will contain seeds of the wetland plants of the region, which may assist in vegetating the wetland. If acceptable topsoil is not available at the site, it may need to be imported or soil amendments used to optimize plant survival and growth. Final grading to tolerances of about +3 cm is necessary to maintain sheet flow conditions in shallow marsh areas.

For some treatment wetland applications involving hazardous substances, double liners may be required. For example, landfill leachate wetlands may need to be double lined. Liner costs are the key factor in design decisions and are addressed in Chapter 23.

### Synthetic Liners

Wetland cells may need to be lined with clay or plastic if regulatory requirements prohibit mixing with groundwater or if natural infiltration rates will make it difficult to maintain surface water wetland conditions. The decision to install a liner in a wetland system and which type of liner to use depends on the project goals and regulatory requirements. Some wetlands are unlined, either because the *in situ* native soils are deemed to have sufficient sealing properties or because groundwater recharge is a function of the system. If the wetland must be lined, there are a variety of liner materials available. The two most common liner materials used are 30-mil (0.76 mm) polyvinyl chloride (PVC) and 40-mil (1.0 mm) high-density polyethylene (HDPE). PVC liners are generally factory-seamed, one-piece liners used on small projects less than 0.1 ha in size. HDPE liners generally come in rolls and are field seamed for larger projects.

Liner materials include: polyvinyl chloride (PVC), polyethylene (high-density HDPE, and linear low-density LLDPE), polypropylene (PPE), and compacted clay. Some liners can include a scrim, that is, a woven net of nylon or polypropylene embedded in the plastic material or encloses the clay (Interstate Technology and Regulatory Council, 2003). Scrims provide extra strength and resistance to tears in the material, and liners with scrims will cost more.

For a given thickness, PVC is generally the least expensive material and the easiest to work with in the field. PVC has the best puncture resistance of all the materials (except those with scrims). It has the most flexibility but also has the least resistance to ultraviolet degradation, and therefore must be covered to protect it from degradation. Because the wetland construction process generally covers the liner with soil or gravel, PVC is a good first choice. A minimum thickness is generally considered to be 0.76 mm (30 mil).

Polyethylene (PE) comes in two forms for liners: linear low-density polyethylene (LLDPE) and high-density PE (HDPE). HDPE is harder to work with in the field than LLDPE, especially in cool weather. The thickness should be 1.00 mm (40 mil) to be equivalent to 0.76 mm PVC for puncture resistance. UV resistance is good for both PE materials. Field repairs are easy. Seaming with tape should provide good waterproof characteristics and is easily performed in the field, at though welds are usually required to affect a more permanent seal at the seams. It is usually 10% more expensive than PVC. On large projects, this cost differential can be offset by the ease of field seaming. Polypropylene (PPE) has good installation properties because field seaming and repairs are easy. It has the best puncture resistance of all three plastics. A major disadvantage is that it is the most expensive of the three materials.

### Reinforced Plastics

Some plastics can include a scrim, that is, a woven net of nylon or PPE embedded in the plastic material. If the project can afford them, liners with scrims such as 1.5 mm (60 mil) Hypalon, 1.5 mm (60 mil) XR-5, or 1.1 mm (45 mil) PPE are the toughest, most resistant to puncture and tears, and UV radiation (U.S. EPA, 2000a; Interstate Technology and Regulatory Council, 2003). Vehicles such as front-end loaders and trucks can drive on them. This presents certain advantages in placing the gravel media (subsurface flow wetlands) or soil (surface flow wetlands). The contractor's savings in placing the gravel media or soil may offset the difference in cost of liner materials. Hypalon is a chloro-sulfonated PE with a nylon scrim. It has excellent UV properties and puncture resistance. It is expensive, and repair is difficult after ageing. Hypalon may be less palatable to nutria, muskrats, and beavers than other synthetic liners based on anecdotal evidence.

XR-5 is a PVC material with a scrim, which has very high strength and excellent puncture resistance, and is easy to repair. XR-5 is the most expensive of all synthetic liners and requires specialized equipment for installation. Reinforced 1.1 mm (45 mil) PPE has all of the advantages of PPE but with higher strength and puncture resistance.

### Clay Liners

Natural materials, such as clays or bentonite, may also be used as liners. Some are composed of a layer of clay between two scrims of finely woven PPE or PE. These have a distinct advantage for irregular-shaped cells because of difficulties of installation of plastic materials in nonrectangular geometries. Bentonite may be considered, but costs depend on the *in situ* soil and the distance from the bentonite supply, which in North America is in the western United States (Wyoming). However, source location is also a key issue for native clays because a nearby source may not be available. In most cases, clay with permeability less than $10^{-6}$–$10^{-7}$ cm/s would be required. The installed thickness would usually be on the order of 30 cm with compaction. These factors combine to make the use of clay a costly alternative, requiring economic comparison with synthetic materials.

## Erosion and Flood Protection

A treatment wetland site may be unavoidably in the floodplain of a river. The question of protection of the wetland from flooding then arises. There are two aspects to potential flooding: maintaining the physical integrity of the system and the effect on treatment during flood events.

Physical damage to dikes and structures is not likely to be serious if the system is not placed in the floodway of the river. The main flood currents should not impinge upon the wetlands. Several treatment wetlands have easily survived gentle inundation without any significant damage, for example, Des Plaines, Illinois; Jackson Bottoms, Oregon; Tarrant County, Texas; and Columbia, Missouri. Care must be taken to allow uniform flooding to prevent uplift of a sealed bottom by hydrostatic forces. In some cases, it may be necessary to install underdrains beneath a wetland liner to relieve hydrostatic pressure in the event of rising regional groundwater. Outlet structures should be designed to allow backflooding (not through-flooding) of wetland cells.



**FIGURE 18.20** Wetland dike erosion occurring due to wave action at Stormwater Treatment Area 1W in Florida. Winds at the site have a long fetch across this large wetland cell. Hurricane-force winds caused this damage.

Damage to the wetland vegetation is not likely for inundation up to a week or two. However, prolonged submergence, such as the midwestern U.S. floods of 1993, which lasted for many weeks, will drown the wetland plants.

The treatment efficiency of the wetland will obviously be reduced under flood conditions. Generally, the dilution effect of the flood flow is so great that reduced efficiency can easily be tolerated for the brief duration of the flood. And, some of the floodwater will receive beneficial detention in the wetland after the flood recedes.

The sides of the dikes must be protected from runoff and wave action, and the attendant erosion during a period of wetland vegetation establishment. Emergent vegetation will prevent the wind fetch necessary to cause wave erosion, but those plants are absent during start-up and in wetlands with intentional patches of open water that abut the dike. Rainfall runoff can erode newly constructed dikes as well. In such circumstances, it is important to stabilize the bank with an appropriate cover crop (Figure 18.20).

## 18.4 WATER CONTROL STRUCTURES

Structures are needed to bring water into the wetland, maintain a desired water level, and to collect the discharge from the wetland. A very large number of options are available, and the choice will depend upon the magnitude of the flow to be managed. Wastewater may be conveyed to a wetland treatment system site by use of a pressurized forcemain, by a gravity pipeline, or by an open channel. Construction of pump stations, pipelines, and open channels are not described in this chapter, and the reader should consult general references on hydraulic design and construction for more information. Once the wastewater is brought to the wetland site, it is typically distributed into the wetland via some distribution system consisting of pipes, channels, or coarse rock beds. Inlet control devices may be used to split the flow between parallel FWS treatment trains. Adjustable weirs are preferred for splitter structures, as this allows the flow split to be adjusted between the treatment trains. Proportional flow splits can be achieved by using weirs of different widths.

### Controlling Inflow

Water flows entering the wetland complex may require flow balancing among flow paths, or complete shutoff of a cell or flow path for maintenance. The wetland inflow may be subjected to flow measurement and water quality monitoring. Therefore, inflow structures may be needed. The complexity of inlet structures depends upon their intended use. A simple upturned pipe may suffice for small flows (Figure 18.21), or complex remote-controlled gates might be needed for large flows (Figure 18.22).

### Spreading the Water

The inlet device initiates the flow of wastewater into the constructed wetland. Because water quality treatment is the primary objective, the inlet device initiates a sheet flow of wastewater in the wetland by broadly and evenly distributing the influent. Multiple inlet devices may be used to distribute flows to large wetlands. The inlet device must be able to shut off during maintenance or resting of the constructed wetland. In some cases, the device is used to monitor flow and water quality. Given the strong impetus to spread the entering water across the wetland width, there remains the subject of how to accomplish that objective. The alternatives include:

- Point inlet
- Sprinklers and dribblers (Figure 18.23)
- Spreader canal
- Gated pipe (Figure 18.24)



**FIGURE 18.21** Inlet structures include point discharges, perhaps with pre-aeration at the Olentangy Wetlands in Ohio.

- Multiple tees
- Perforated pipe
- Large rock zone
- Spreader swale
- Castellated weir

Figure 18.25 shows several approaches that have been used to distribute inflows and create sheet flow conditions in constructed wetlands. These often involve gated distribution pipes or spreader canals, but multiple inlet points have also been successfully used. A single inlet point is usually not an appropriate design based on hydraulic efficiency considerations. However, in very small systems, point inlets may be acceptable. For instance, they may be configured to provide aeration of the incoming water (Figure 16.21 and Figure 18.21). Selection of one of these alternative inlet configurations is based on wastewater characteristics, cost, and engineer preference. In cold climates, inlet water distribution and its associated plumbing must be kept below the ice layer.

The key to successful water spreading is to provide both side-to-side introduction and head equalization from side to side. The latter is accomplished by the use of an inlet spreader canal or deep zone. This provides a low resistance flow path across the flow direction, and thus equalizes the water stage across the inlet. This spreader canal may be filled with coarse rock if desired for animal control. Only if the cross width is very great will it be necessary to consider the side-to-side head loss.

Inflow can be distributed by gravity flow or by pressurized flow. Gravity flow is desirable because it conserves energy and reduces operation and maintenance costs. Gravity flow distribution systems may require larger pipe sizes to reduce head loss to acceptable levels. Pressurized inflows may result in high orifice velocities that can cause sediment erosion or physical damage to emergent wetland plants. By providing a greater number of individual outlets and greater cross-sectional area, erosion is less likely to result.

Gated distribution pipes have been used at a number of constructed surface flow wetland treatment systems. The header pipe material is selected on the basis of required system life and cost. Inlet pipes have been made from a variety of materials, including irrigation gauge aluminum pipe, PVC water pipe, and ductile iron. Because the header pipe is exposed to temperature extremes and UV radiation, PVC will have a limited life expectancy and may break. Aluminum pipe may dent. Ductile iron is strong and has a long life, but it corrodes and generally costs more. The pipe must be sized to control head loss so that flow is distributed evenly over the distribution pipelines. A clean-out should be provided at the end of distribution pipes so that accumulated solids can be removed periodically. Pressure relief valves should be installed in pipes where air blocks might occur.



**FIGURE 18.22** Wetland inlet structure at Stormwater Treatment Area 1 East in Florida. This automated structure, remotely controlled by telemetry, is one of 13 that bring water into the 2,080-ha constructed wetland. Its capacity is 730,000 $m^3/d$.



**FIGURE 18.23** Sprinkler and rock bed system used to distribute the inlet flows to the FWS wetland at Mandeville, Louisiana.

Gated distribution pipes use a variety of outlet configurations, depending on specific goals. Simple, plastic, sliding outlet gates may be used on aluminum irrigation pipe to regulate flow volume (from off to wide open). PVC fittings or simple orifices are commonly used in PVC distribution systems. If flow distribution is a concern because of the length of the distribution pipe (potentially a problem when the distribution pipe is more than about 20 m long), swivel outlet pipes can be used to more precisely control the elevation of individual outlets. Swivel outlets also can be used with ductile iron pipe, although they may be more difficult to regulate after they begin to corrode. Perforated pipes can be used successfully on short, gravity-fed, distribution pipes. Orifices should be at least 5 cm in diameter to minimize plugging by wastewater solids and to reduce the need for frequent maintenance.

Level spreader swales and castellated weirs also provide, in theory, a gravity-fed distribution system. Outlets are hard structures that are precisely leveled during construction. These are difficult to install, because a very high degree of vertical control on positioning is required. Because these typically provide no adjustment, they are not recommended for treatment wetlands.

Inlet selection is partly based on the need for fail-safe operation under the prevailing climatic conditions. The need for uniform transverse distribution was discussed in the preceding text. Problems to be avoided include freeze-up in



**FIGURE 18.24** The inlet to the New Hanover County treatment wetland near Wilmington, North Carolina. This warm climate system utilizes gated pipe distribution onto a concrete mat slope into an inlet deep zone. The liner underneath may be seen exposed in the background, as the divider between the two flow paths.

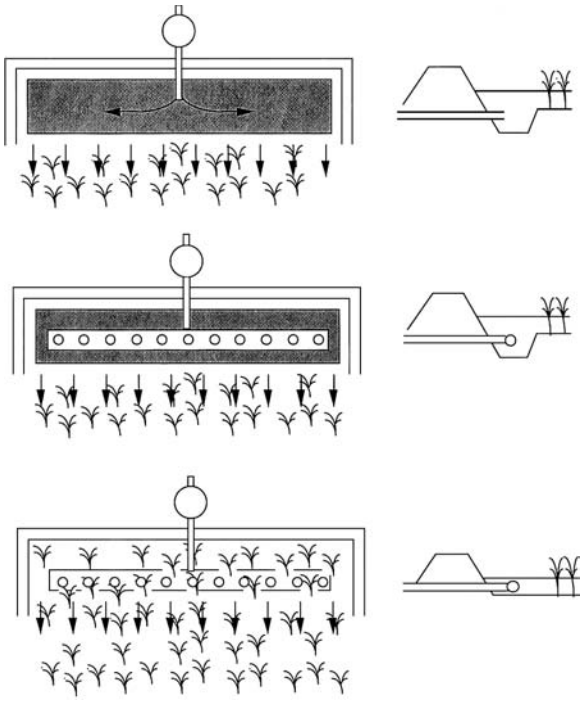


**FIGURE 18.25** Inlet distribution options include point discharges, spreader deep zones, and gated or perforated pipes, or combinations.

northern climates and clogging by debris or animals. Freezing can be prevented by use of buried feeder mains followed by point discharge structures or submerged distribution headers. Allowance must be made for ice formation on the surface of the wetland waters. Incoming waters that originate in lagoons can often contain leaves, sticks, and other debris. These can block small orifices in a distribution system. The collection well in the lagoon system is frequently a favorite habitat for turtles, which can block flow. Screens or other appropriate prevention measures should be considered.

If the inlet distribution system should fail, the route of the incoming water should be acceptable for continued treatment and unimpaired system integrity. This is accomplished by the incorporation of an emergency overflow at the inlet.

### Outlets

Wetland outlet design is important for maintaining sheet flow distribution, for controlling water level, and for monitoring flow and water quality. Many outlet designs have been used in successfully constructed wetlands, ranging in complexity from a corrugated metal pipe embedded in a berm to remote-controlled motor-actuated gates.

### Water Collection

As shown in Figure 18.5B, flow distribution in a rectangular wetland basin with a single outlet point is compromised and effective hydraulic retention time is decreased. This situation can be improved by adding outlet weirs along the downstream berm of the wetland or by using a perpendicular deep zone to collect and route flows to a single outlet weir. If a terminal deep zone is used to accomplish this flow control goal, it should be kept as small as possible to discourage a long residence time and subsequent algal growth.

### Level Control

Outlet structures are designed to be the primary control on water level within the wetland. Selection of a particular device should include ease of operation as well as ability to set the desired water level. The water level within the FWS wetland cell is controlled at the downstream end of the wetland through the use of a hydraulic control structure. Having a separate control structure in each FWS cell allows the water level in the cells to be adjusted independently. This operational flexibility is very important for managing water levels during plant establishment, allowing access for maintenance, and planning for winter operations.

The hydraulic control structure should have enough vertical adjustment to accommodate the full range of water levels that will be expected to occur over the life of the facility. The outlet structure should have enough weir capacity to pass the peak storm events within the designed freeboard range. A conceptual example of a good outlet control structure for a FWS wetland is shown in Figure 18.26.

For small wetlands, common outlet control structures utilize stacked stoplogs (flashboards) or adjustable weir gates (Figure 18.27). Stoplog designs made from channel iron and dimensioned lumber are not suitable for small-scale projects because their high leakage rates can interfere with maintaining a stable water level. Manufactured stoplog systems have much lower leakage rates and are better suited to wetland projects. Water level adjustment is performed through the addition or removal of stoplogs. With stoplog systems, adjustment increments are limited to the height of the stoplog. Adjustable weir gates offer the significant advantage of being continuously adjustable. A deep-water zone at the outlet provides a sump that allows for greater height adjustment, which is useful in the event that the cell must be drained.

An outlet control device with a fixed elevation should not be used for a FWS wetland, as these do not provide the degree of operator control necessary for successful system operation.

Outlet weirs or pipes do not need to be sized to handle peak wastewater inflow rates because of the amplitude dampening effect of volumetric storage in the wetland cell. However, weirs should be designed to pass maximum average daily or weekly flows plus the volume of rainwater that might fall on the wetland surface from a storm event, while not compromising the design depth range. Weirs can be designed to pass these storm flows quickly (broad-crested horizontal weir) or slowly (v-notch or vertical notch weirs). Weir design is also important in quantifying wetland outflow rates. Integrating flow meters can be designed to work with almost any weir configuration.

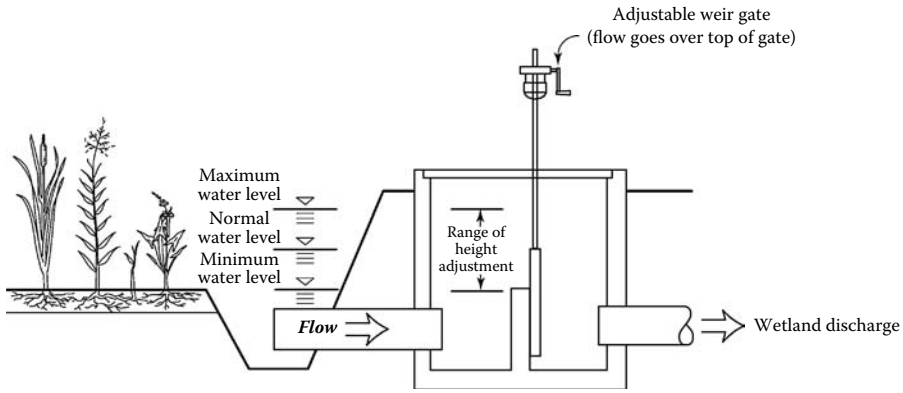



**FIGURE 18.26** Typical outlet level control structure for a FWS wetland. (From Wallace and Knight (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements*. Final Report, Project 01-CTS-5, Water Environment Research Foundation (WERF): Alexandria, Virginia. Reprinted with permission.)

One of the most important functions of outlet devices is to provide convenient control of wetland water levels. The required water level is based on requirements for plant growth, hydraulic residence time, and cell maintenance. In most constructed surface-flow wetland treatment systems, water levels should be controllable between 0 (the cell bottom) and 60 cm. If water level adjustments will be frequent, making the use of stoplogs impractical, then variable control weir gates can be designed. For large applications, these weir designs can become quite sophisticated and may rely on

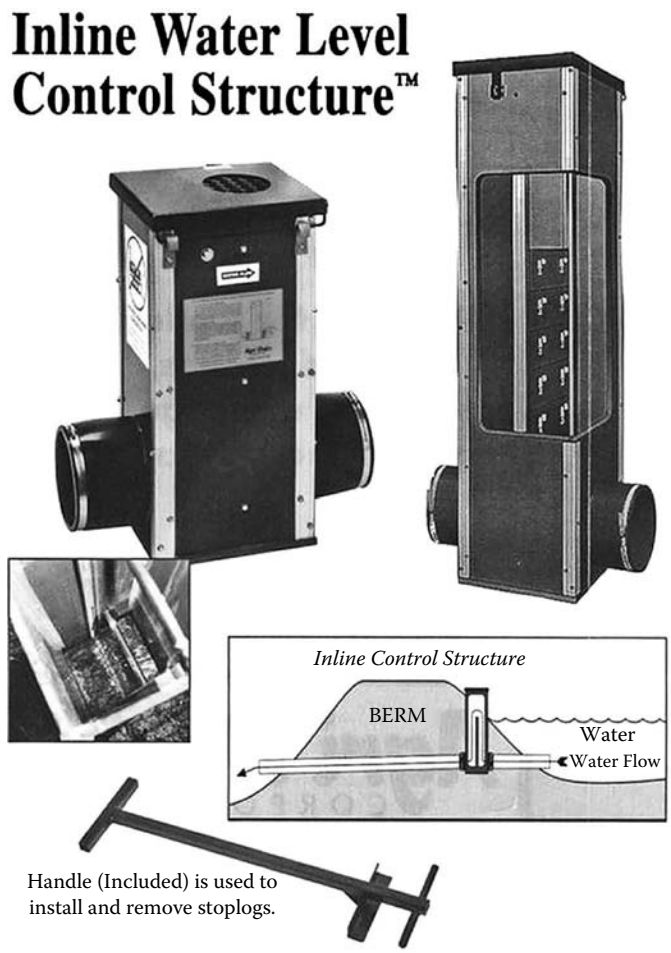



**FIGURE 18.27** Prefabricated stoplog structures are available in pipe sizes to 60 cm with capacities up to 3,500 $m^3$/d. The one pictured is manufactured by the Agri Drain Corporation, Adair, Iowa. (Photo from company literature. Reprinted with permission.)

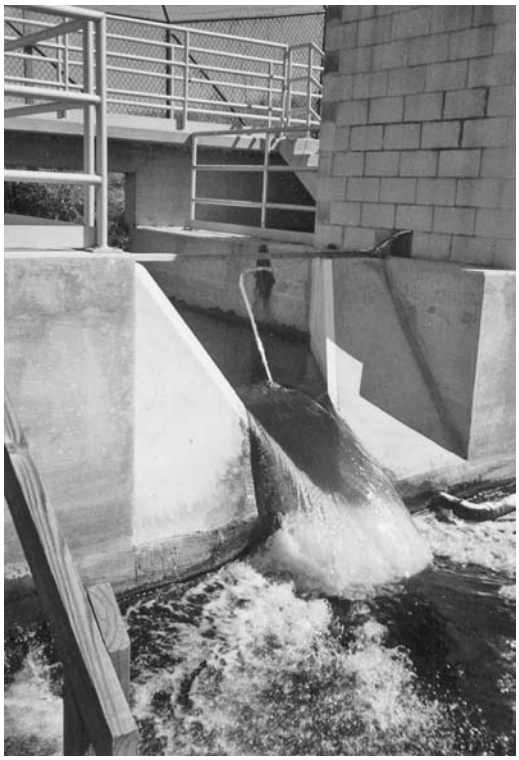


**FIGURE 18.28** The effluent structure at the 500-ha Lakeland, Florida, FWS treatment wetland. The average flow is approximately 35,000 $m^3$/d. A sampling pipe is positioned in the center of the flow.

electric power and remote control, and be combined with other flow measurement elements such as flumes (Figure 18.28).

Flows at the wetland outlet device are usually calibrated to stage and can be monitored with a staff gauge or water level recorder. For a variable outlet configuration or for a hybrid weir design, a stage-discharge curve can be prepared by using periodic volumetric flow measurements at various stage values.

### Trash Racks

Outlet structures are prone to accumulation of debris. The combination of water direction and the litter production of the wetland ensures the potential for an accumulation at outlets (Figure 18.29). A large-mesh debris fence, placed a meter or two from the structure, can alleviate this problem, but it will become clogged and form a barrier to flow upstream of the discharge structure. The use of floating skimmer boards can keep floating debris from the outlet. Its use is coupled with water withdrawal at a subsurface location. In very large systems, automated cleaning devices have been used to good advantage.

### Emergency Overflows

If flood conditions are anticipated, there should be appropriate provisions, possibly including emergency overflow spillways and back-flood gates. Emergency spillways are simply armored low sections in the containment levees with a preset elevation of their bottom (Figure 18.30).

### Pumps

Pumps are required for one of four reasons: supplying water to the wetland for treatment, removing water from the wetland to downstream uses, recycling water back to the inlet, and returning groundwater leakage to the wetland. Some systems have none of these; no North American system presently has more than three. Pumping incurs a capital cost, which is repeated on a time scale shorter than the life span of the ecosystem. Further, the operations and maintenance costs add to the capitalized total cost of the facility.

In very flat terrain, a low-head inlet pump may be needed to transfer water from the source to the wetlands at an elevation sufficient to move the water cross-country to the outlet. For the same reason, it may be necessary to utilize pumping to lower the water level at the system outlet (see Figure 14.2).



**FIGURE 18.29** Both live and dead plant material can collect at wetland outlets. Photo from the Imperial, California, treatment wetland.



**FIGURE 18.30** An emergency spillway at the Onaway, Michigan, treatment wetland. This low spot in the levee has been armored with coarse riprap.

Redundancy is usually built into these pumping facilities so that a portion of the capacity is normally out of service and accessible for maintenance.

If the wetland leaks an appreciable amount of water, it is a common practice to surround the wetland with a perimeter ditch to collect lateral seepage. Seepage water is then pumped back to the wetlands. This is the case in South Florida, where water conservation wetlands and stormwater treatment areas are so configured.

Recycle inevitably requires pumping. Treatment advantages that may be gained by this procedure are, therefore, expensive and sometimes are more economically accomplished by increasing the size of the wetland.

## 18.5 STORMWATER WETLANDS

The physical design of event-driven wetlands differs in some respects from continuous flow wetlands because they are focused on capture rather than efficient flow hydraulics. Nonetheless, it should be apparent that the general concepts of continuous flow wetlands mostly carry over to event-driven systems. However, the desire for ancillary benefits seems stronger among those who design, build, and permit stormwater wetlands. Therefore, these systems rarely are depicted as having regular geometry, and the configuration is more likely to reflect habitat than optimization of treatment.

### Bathymetry

Because runoff water is usually contaminated by TSS, most treatment wetland systems contain a wet pond or forebay at the inlet (Figure 18.31). This process element serves as a repository for settled solids, preventing them from clogging the vegetation in the remainder of the system. This deep-water area also provides storage for captured events and is typically sized as a fraction of the overall footprint. For instance, Schueler (1992) shows options with 10–45% of the area allocated to the deep inlet pool. The balance of the footprint is usually devoted to shallow and deep emergent marsh, and possibly to deep cross-zones (Figure 18.31). This patterning of the bathymetry is completely in agreement with the principles elucidated for good hydraulic efficiency for continuous flow systems. However, some literature goes on to add channelization of the marsh from inlet to outlet, presumably in an effort to emulate natural riverine wetlands (Schueler, 1992; Interstate Technology and Regulatory Council, 2003). That addition is detrimental to treatment and can only be justified by a need to avoid head loss during extreme high flows. However, those extreme flows would normally be bypassed.

Because it is not reasonable to expect to treat those extreme flows, there is an implied need to provide an alternate routing for those low-return frequency events.

### Control Structures

Because of the erratic arrival of water to the wetland, it is usually not feasible to consider operation at a fixed depth. The outlet structure for the system is designed to provide a workable hydropattern for the treatment system. A fixed weir or orifice is normally not a good choice (Somes and Wong, 1997), but there are simple passive alternatives that do work well. A siphon can act to regulate the water level between two limits with a well-modulated discharge during interim level conditions (Figure 18.32). In fact, an outlet siphon has also functioned well for the continuous flow wetland at Lake Nebagamon, Wisconsin (see Figures 3.23 and 18.19). The lower limit reflects a volume of water termed the *permanent pool*, whereas the upper limit is *extended storage*. A second alternative utilizes multiple outlets (Figure 18.33). Again, a permanent pool is maintained by a first (small) overflow outlet. During high flow events, that outlet is not capable of passing the full flow, and the water level rises until a second (larger)

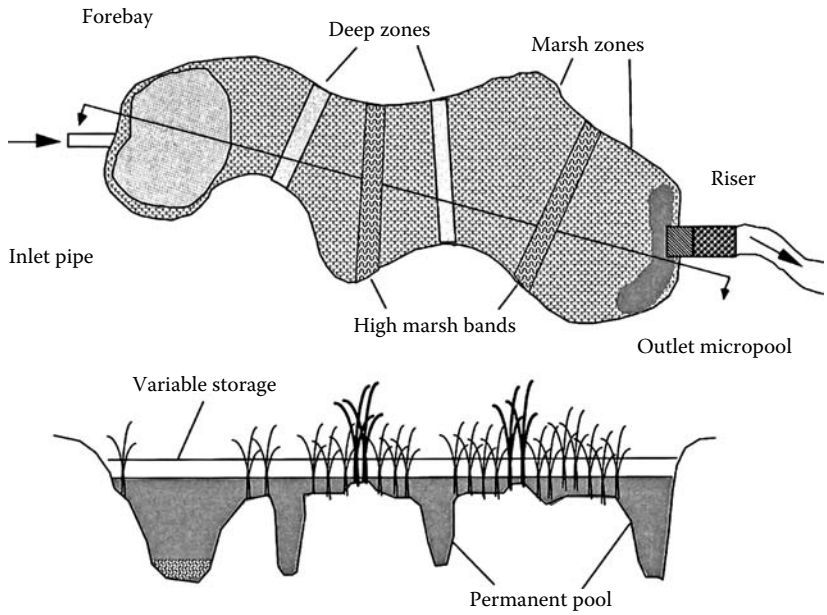



**FIGURE 18.31** Conceptual configuration of an urban stormwater treatment wetland.

outlet begins to function. During extreme flow events, both outlets may be overwhelmed and levels would rise to the emergency overflow spillway. A drain may also be provided so that all water can be removed from the wetland.

### Supplemental Water Source

Because interevent durations can be large, with dryout conditions, provision is needed for a water supply to maintain the permanent pool. There are two ways that this can be accomplished: by positioning the wetland bottom properly with respect to the regional water table or by providing a source of irrigation water. Clearly the least bothersome is to rely upon the regional groundwater table to maintain the permanent pool but that is not always possible.

Some stormwater wetlands are allowed to dry out. For instance, STA6 in South Florida drys for several months of the year. Many varieties of wetland vegetation can withstand, or even prefer, that hydropattern, and so the existence of that wetland is not threatened. However, there are small penalties for water quality improvement, as discussed in Chapter 14 (see Figure 14.9).

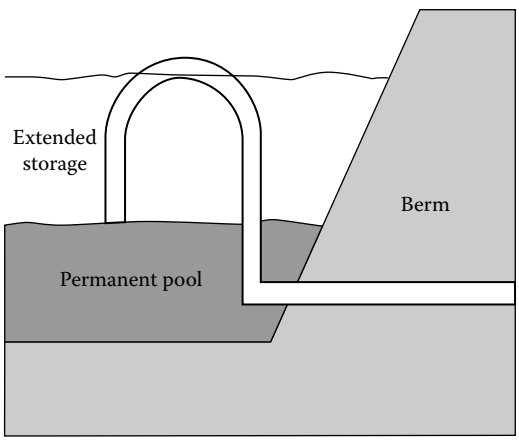



**FIGURE 18.32** Cross section of a siphon outflow structure.

## 18.6 WETLAND CONSTRUCTION

At one extreme, wetland construction can be an enjoyable experience of creation, and at the other extreme it can be a living hell of frustration and financial ruin. Almost any consultant or contractor who has built a wetland will tell you that there is no substitute for actual experience. The purpose of this section is to provide information that will increase chances for successful wetland construction the first time around, and to help with understanding and overcoming difficulties that may arise during construction of wetland treatment systems.

Wetland treatment system construction can be broken into two topics: site civil construction and vegetation establishment. Site civil construction is a broad discipline of engineering and has been refined through centuries of experience. This chapter does not attempt to discuss all aspects of this engineering practice but rather describes the highlights of site civil construction that are most relevant to wetland treatment system construction. A general discussion of wetland site civil construction is provided by Tomljanovich and Perez (1989).

### Site Preparation

Wetland treatment systems are usually constructed in nonwetland sites. The greatest amount of site preparation is typically required when an upland site has existing vegetation or structures. In some cases, these vegetation and/or structures must be removed to allow construction of wetland cells. Clearly,

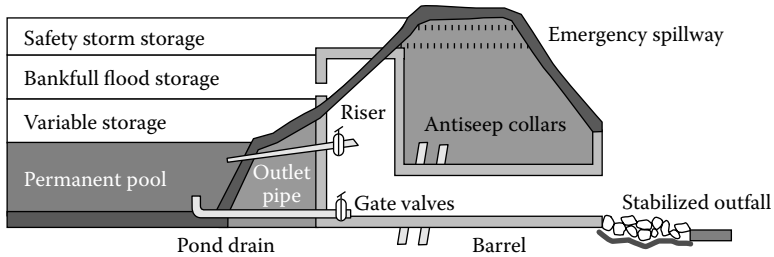



**FIGURE 18.33** Cross section of a typical wetland outflow structure. (Adapted from Schueler (1992) *Design of Stormwater Wetland Systems: Guidelines for Creating Diverse and Effective Stormwater Wetlands in the Mid-Atlantic Region*. Metropolitan Washington Council of Governments: Washington, D.C.)

construction costs can be reduced if existing vegetation and structures are avoided during project design. Assuming that design cannot avoid these features, the first step in construction is typically site clearing. Site clearing might include demolition and removal of rubble from existing structures or alternatively involve clearing of trees, rocks, or other obstacles present on the site. Tree clearing may involve the recovery of valuable timber followed by the uprooting and toppling remaining trees and stumps with a bulldozer.

As part of site clearing for a constructed wetland project, it is important to remove any belowground obstacles that might interfere with site grading or with the integrity of site soils or with plant growth. Grubbing refers to the removal of stumps and roots following removal of trees and shrubby growth on a site. Grubbing should also remove any unwanted, buried structures such as utilities and foundations on previously developed sites. Cleared vegetation may be chipped and temporarily staged with other debris in an upland area. Chipped material can serve as mulch for terrestrial planting operations. Nonrecyclable material should be disposed at a landfill.

Erosion on-site and off-site should be prevented as much as possible during wetland construction. Before any construction activities that might result in the disturbance of the existing soils at the proposed location of the constructed wetlands, soil erosion and sediment control measures should be constructed in accordance with the appropriate state standards for soil erosion and sediment control. The objective is to prevent sediment from being washed from the excavated areas into undisturbed areas. All soil erosion and sediment control measures should be maintained in good condition and left in place until permanent vegetation cover is established. Silt screens can be used to minimize off-site erosion, which consist of PE fabric or hay bales anchored by use of wooden or metal stakes. Geotextiles and fabric cloth can also be used to protect berms and high-flow areas within the constructed wetlands from erosion.

### Grading and Subgrade Preparation

Following site clearing and grubbing, basic landforming can begin. Using soil grading and hauling equipment, the contractor will generally remove and stockpile topsoils from the footprint of future wetland cells to be reused for final wetland-planting media. Following removal of the topsoil, the general contours of the wetland site are graded and all site features are horizontally located by standard surveying methods. The constructed wetland site is excavated to the contours, according to the detailed specifications prepared during the design process, in the form of cross-sectional and contour drawings showing the current and finished grades. Following removal of the topsoil, the subsoil material should be removed to a rough grade, approximately 15–30 cm below the final grade consistent with the planned elevations. This allows room in the excavation for placement of the liner (if required).

Preparation of the subgrade is an important part of the construction process. It is important that the subgrade be properly compacted to a design density because this will reduce settling under the liner and avoid the associated stress. Grading tolerance specifications, before placement of liner materials, should be kept within ±5 cm. Heavy construction equipment should be kept off rough grades if the subgrade is unable to support the vehicle weight without deformation. Final site grading follows construction of berms, deep zones, and islands. Because a small amount of erosion can negate this work, in rainy climates it is advisable to establish ground cover on berms before final grading. Final grading typically consists of leveling the wetland cell bottom to allow for a uniform thickness of rooting soil in the completed wetland. Generally, bottom elevation is constant from side to side within the wetland cell and decreases slightly or is flat from the upstream to the downstream end of the wetland. Finish subgrades of the wetland bottom may have tighter tolerances, ±3 cm, which can be accomplished by use of a laser-leveling system or by hand raking. Hand raking works well in small cells by pulling a level string across the width of the cell and working downgradient. Laser leveling with laser-equipped bulldozers and scrapers works best for larger sites.

Wetland cell berms are typically constructed in lifts or successive steps of placing and compacting soils. Interior berm cores and any wire meshing needed for muskrat burrow prevention should be placed in the berm during this process. Soil moisture is critical during compaction of berms

and internal cell liners, and construction may be delayed by excessively moist weather conditions.

As soon as berms have been shaped and dressed with any topsoils that are specified, they should be protected from erosion by use of straw and/or cover crops of grasses. Typically, a mixture of grass species is specified to allow establishment under variable climatic conditions. Hydroseeding with a mixture of grass seeds, shredded newsprint, and fertilizer has been found to be an effective method for rapid cover establishment on wetland berms. All cover crops should be irrigated as needed to insure good germination and growth of plants.

### Liner Placement

Wetland liners are placed following final cell grading. If large or sharp rocks are present in the subgrade, then liner protection is required. Geotextile mats are recommended if the subgrade cannot be prepared without jeopardizing liner placement. Geotextiles are easier to place than sand and have the advantage of staying on the sloped berms. If a sand layer is used, 15–30 cm is typically used.

The installation and testing of liners is an important element of a successful wetland project (Interstate Technology and Regulatory Council, 2003). This activity requires specialists and would usually be subcontracted. Specifications should include some reference requiring written approval of the subgrade by the liner installer before its installation. Written acceptance of subgrade preparation by experienced liner installers is usually a condition for maintaining manufacturers' warranties. Site preparation is very dependent upon the type of liner but will usually include compaction before to installation. Following installation of liners, wetland cells should be leak-tested before further installation of topsoils, media, or wetlands vegetation.

A protective layer of sand is sometimes placed to protect the liner from sharp objects in the rooting soil or from activities involved in rooting soil placement and planting (Figure 18.34).

### Rooting Soil Placement

Following site preparation and grading, a layer of appropriate soil is placed to serve as a medium for plant establishment and growth. Suitable substrate (at least 20–30 cm deep) is brought into establish the topographic contours of the constructed wetlands to final grade. Topsoil for backfilling should consist of a soil suitable for supporting plants in that region. Rooting soil may vary from topsoil to subsoil, depending on the application and the design requirements. Generally, hydric soils need not be used as fill material, because in wetlands, hydrologic regimes are such that hydric parameters will develop over time. The organic content, typical of wetland soils, will develop as a function of both aboveground and belowground plant production. However, the use of hydric soils as fill can assist in the introduction of wetland vegetative species into the system, depending on the location from where the soils were obtained. Because of the need to protect natural wetlands, it is necessary, in many cases, to obtain regulatory permission for the borrowing of wetland soils from off-site. The original topsoils from the site may be usable and may be transferred from a stockpile to the wetland floor.

Emergent wetland plants require suitable soil conditions for rapid initial growth, and long-term propagation and survival. Loamy soils, containing a mixture of sand, silt, and clays, are optimal for growth of most plants. These soils have adequate texture and organic matter to retain moisture, allow diffusion of oxygen and carbon dioxide, and retain nutrients for absorption through the plant roots. Although sandy soils provide good gas exchange and allow for relatively unencumbered root development, they do not retain nutrients or moisture well. As long as nutrients and moisture are adequate through continual flooding with enriched wastewaters, sandy soils can be used for constructed wetland development. Clays



**FIGURE 18.34** Covering the liner at the Lake Nebagamon, Wisconsin, treatment wetland.

or other fine-textured soils have poor characteristics for gas exchange and root development, and may limit water movement to the wetland plant roots. However, clays typically have high cation exchange capacities and provide excellent binding for important plant growth nutrients. In tight clays, the nutrient advantage is overwhelmed by the poor texture for root development, resulting in very slow plant growth. When mixed with sands and silts, organic soils are frequently the best medium for growth of wetland plants. Good agricultural topsoils frequently provide an excellent medium for wetland plant growth and should be stockpiled for use during wetland construction.

In those applications where the wastewater is nutrient poor (i.e., groundwater remediation or tertiary effluents), this topsoil may be amended with agricultural fertilizer at a rate typically used to cultivate a grass crop, which can be determined by soil analyses and recommendations from the local agricultural extension agent.

If the wetland is constructed on a compacted clay or plastic liner, soil will have to be placed on the liner so that the finished surface is level and free of vehicular tracks. Wetland cell floors may also be seeded with a cover crop of grasses if final construction or wetland planting will be delayed due to the need for a seasonal-planting window.

### Piping and Structures

Wetland water conveyance systems may be fabricated off-site, and then brought to the site and installed, or in some cases may be totally fabricated on-site. Wastewater conveyance systems typically include piping and pump stations, and water control structures such as splitter boxes, weirs, and flumes. Wastewater may be conveyed to a wetland treatment system site by use of a pressurized forcemain, by a gravity pipeline, or by an open channel. Construction of pump stations, pipelines, and open channels are not described in this chapter, and the reader should consult general references on hydraulic design and construction for more information. Once the wastewater is brought to the wetland site, it is typically distributed into the wetland via some distribution system consisting of pipes, channels, or coarse rock beds. Construction related to these wetland distribution systems is briefly discussed.

As described previously, effective flow distribution is critical to optimizing wetland treatment performance. Wetland distribution system construction is important to increase the ability of the system to achieve this potential. Gravity distribution systems rely on firm foundations and level installation across the width of the wetland cells. Ample footings should be provided for pipe-support structures. Frost protection must be considered for cold climate systems. If wetland design allows for outlet height adjustment, then vertical control tolerances are less critical.

Weirs and flumes should be constructed using standard site civil construction techniques. Heavy structures require adequate footings on noncompressible soils and adequate reinforcement. If organic or high-clay soils are present in locations where footings are to be constructed, they should be removed and replaced with better geotechnical materials.

## 18.7 VEGETATION ESTABLISHMENT

Wetland systems owe much of their public acceptance to the vegetative communities they support. Plants play an extremely important role in FWS wetlands. They shade the water column, provide surface area for biofilm growth, and aid in the cycling of nutrients and organic carbon. Thousands of plant species are adapted to grow in wetlands. However, only a few plant species have been widely used in constructed wetlands. Plants in treatment wetlands should be tolerant of the water quality of the project and be able to propagate. They also must be tolerant of continually saturated soil conditions and be able to thrive in the soil environment created by the particular wastewater under consideration. A number of references are available to provide guidance with wetland plant establishment (Gearheart *et al*., 1989; Hammer, 1992; Thurnhorst, 1993; U.S. Army Corps of Engineers, 2000; Allen *et al*., 2002; Wallace and Knight, 2006).

### Selection of Plants

Thousands of plant species are adapted to grow in wetlands. However, only a few plant species have been widely used in constructed wetlands. Ideally, plants in treatment wetlands should be tolerant of high-nutrient levels and be able to propagate through rhizome spread. They also must be tolerant of saturated soil conditions and be able to thrive in an anaerobic environment. In addition, wetland plants must be readily available from local plant nurseries. When selecting plant species for treatment wetlands, designers should consider the following factors:

- *Hydrology.* Natural wetlands develop a plant community that self-designs in response to the frequency and duration of saturation, and/or inundation (U.S. Army Corps of Engineers, 1987). Constructed wetlands usually have a simplified hydrology because they treat a relatively constant flow of wastewater. However, designers should identify areas that will normally be inundated and the depth of water within these areas. It is important to evaluate the water levels at both normal and peak flow rates to ensure plant survival.
- *Climate and latitude.* Selected plants must be able to survive in the local climate. Because some species such as cattails (*Typha spp.*) have very wide geographic distributions, it is important to specify planting stock that is available within a reasonably close proximity to the project site (e.g., 500 km) and/or produced in the same climatic region.
- *Method of introduction.* Vegetation may be planted or natural colonization may be allowed to occur.

Natural regrowth takes longer and is subjected to the availability of regional sources of propagules and their transport. Planting allows initial control of the species mix and usually takes less time to develop a mature wetland ecosystem.

- *Cost and availability of planting stock*. Plant species native to the region are preferred. Introduction of exotic plant species can result in penalties, ranging from significant permitting delays to violation of state/provincial and federal laws. Designers need to understand which plants are native to the region and, among those native plants, which ones are available in reasonable quantities at a reasonable cost.
- *Plant size*. Wetland vegetation should be selected based upon the project budget and the size of plant that is available from local suppliers. Plants with larger root biomass will have a higher survival rate. However, larger plants cost more, so the designer must balance the plant cost (and thus, plant size) against the project budget. If small plant stock is chosen, plant maintenance may need to be increased to prevent weeds and unwanted plant species from dominating the system.
- *Rate of colonization of the selected plant species*. Plant species that spread rapidly require less planting stock, which will reduce the cost and minimize the potential for weed invasion. However, aggressively spreading species will likely crowd out other plants, resulting in a less diverse plant community in the wetland.
- *Water quality (including salinity, pH, ammonia, sulfide, and oxygen demand)*. For successful establishment, the selected plant species must be able to tolerate the water quality characteristics within the wetland.
- *Project goals*. Is the wetland expected to provide high-quality wildlife habitat? If so, a more diverse planting would be appropriate. If treatment is the primary project goal, a simplified planting scheme, resulting in less diversity, may better suit project goals.
- *Maintenance*. Vegetation management is crucial during the first year of plant establishment, primarily via water depth control.

## Treatment Potential

As discussed in Chapter 3, the structure of the plant community is a more important determinant of pollutant removal performance than the species. There are numerous studies that look at the initial uptake of various nutrients and contaminants; for instance, see Tanner (1996) or Iamchaturapatr *et al*. (2007). But, to the extent that vegetation is directly involved in removals, it is the entire biogeochemical cycle that matters, not initial uptake. Further, it is the presence of physical structures below water that determines the site densities for microorganisms and algae, which in many wetlands are a major contributing factor to removals. Emergent wetland plants provide a wide range of treatment mechanisms in FWS wetlands, (Sinclair Knight Merz, 2000) including:

- Increased sedimentation by reducing wind-induced mixing and resuspension
- Additional surface area for increased biofilm growth and uptake of soluble pollutants
- Increased surface area for particle interception
- Shade from the plant canopy over the water column to reduce algae growth
- Induced flocculation of smaller colloidal particles into larger, settleable particles

Most of these mechanisms are structural in nature. Consequently, selecting the "perfect" plant species is not nearly as important as establishing a functional plant canopy. Thus, performance results often show little difference among species mixes of the same general structure (see Table 18.3, for example).

The designer has wide latitude in the selection of plant species and mixtures thereof. Greenway (2003) studied seven wetlands in Queensland, Australia, and concluded that a large variety of aquatic plants were suitable for surface flow constructed wetlands to polish wastewater effluent. However, species such as *Typha, Phragmites, Paspalum*, and *Ipomoea* were capable of displacing other species and forming dense monotypic stands. Floating and submerged species have higher nutrient content than emergent species; they also remove nutrients directly from the water column. Greenway (2003) recommended a mixture of species to maximize nutrient removal and enhance the ecological value of treatment wetlands.

Genetically engineered plants are on the horizon for consideration for future applications. Nandakumar *et al*. (2005) have demonstrated that it is possible to genetically engineer *Typha latifolia* to achieve the long-term objective of introducing candidate genes for phytoremediation.

## Diversity

Establishing diverse wetland vegetation is a stated goal in most wetland mitigations and, to the extent practicable, should also be a goal for treatment wetlands. This is desirable for a variety of reasons. A relatively large number of species means an array of environmental tolerances is represented. As the new wetland experiences fluctuations—such as water level, temperature, and herbivory—in various environmental conditions over time, some plants or species will not survive, but others may thrive. If the project is dominated by a few species, there is a possibility that the project will fail with the death of only one species. Planting a variety of species increases the chances for success of at least a few species. A more diverse mix of plant species will be better able to accommodate changes in water quality and flow.

**TABLE 18.3**
**Percent Removals for the Oregon State Side-by-Side Animal Waste Treatment Cells with Different Vegetation Types**

| Vegetation Type | | BOD | COD | TSS | TKN | $NH_4$-N | TP |
|---|---|---|---|---|---|---|---|
| Grass/Cattail/Bulrush | mean | 57 | 46 | 52 | 48 | 39 | 48 |
| (60:30:10) | s.e. | 7 | 6 | 12 | 5 | 7 | 6 |
| Grass/Cattail/Bulrush | mean | 57 | 46 | 60 | 47 | 40 | 51 |
| (50:30:20) | s.e. | 5 | 4 | 7 | 4 | 5 | 6 |
| Grass/Cattail | mean | 52 | 44 | 55 | 42 | 37 | 39 |
| (80:10) | s.e. | 6 | 5 | 10 | 5 | 6 | 6 |
| Cattail/Hydrocotyle/Bulrush | mean | 46 | 40 | 44 | 35 | 29 | 35 |
| (35:30:20) | s.e. | 6 | 5 | 10 | 5 | 5 | 6 |

*Note:* Data are averages of monthly measurements during 1993–1994.

*Source:* Data from Moore and Niswander (1997) *Constructed Wetlands for Animal Waste Treatment*. Payne Engineering, CH2M Hill (Eds.), Gainesville, Florida, pp. II-31–II-33; NADB database (1998) *North American Treatment Wetland Database (NADB), Version 2.0*. Compiled by CH2M Hill. Gainesville, Florida.

Monocultures, or communities with a single dominant species, are often considered to have a limited ecological value. The ecological benefits of diverse communities are numerous, given that the number of plant species and structural complexity of natural ecosystems generally correlate with wildlife species richness, particularly for birds. Consequently, a diversity of wetland plants within a system increases wildlife value. Various plant species association and hydrological conditions provide required habitats for different life history phases of animals such as feeding, winter cover, and breeding.

Table 3.1 summarizes the number of plant species reported for the treatment wetlands in the North American Treatment Wetland Database (NADB database, 1998; U.S. EPA, 2000a; Knight *et al*., 2001). More than 800 species of macrophytic plants have been reported in natural and constructed treatment wetlands. Of these observed plant species, 693 are emergent herbaceous macrophytes, 36 are floating aquatic species, 12 are submerged aquatics, 57 are shrubs, 55 are trees, and 18 are vines. The NADB v.2 lists 593 macrophytic plant species reported from constructed treatment wetlands and 427 species from natural treatment wetlands. Emergent herbaceous macrophytes account for 501 species in constructed treatment wetlands and 290 species in natural treatment wetlands. A significant variety of tree and shrub species occur in a few constructed wetlands (primarily in the constructed cells of the Orange County, Florida, system). Tree and shrub species are well represented in natural treatment wetlands with 88 different species recorded. Although the NABD v.2.0 data collection effort provided some insight into the plant diversity at constructed and natural treatment wetland sites in North America, the level of investigation varied widely between sites. These numbers should be considered as the minimum number of plant species that actually occur in treatment wetlands.

## Natural Selection and Convergence

It may be argued that the vegetative community of a treatment wetland is not, in the long term, a choice of the designer. The plants that are best adapted to the hydrologic regime and water quality of the wetland will prevail eventually if not controlled by weeding or herbiciding. Studies by Mitsch and Zhang (2004) showed that the plant communities and performance of two wetlands converged after about three years of development. One wetland was planted with 13 species; the other was unplanted.

Most FWS treatment wetlands undergo a process of alteration and self-design after the initial planting. The plant community that develops over time is a function of organic and nutrient loadings, hydrology, and climate. FWS wetlands that are heavily loaded with organic matter and nutrients will typically develop a less-diverse plant community because fewer plant species are able to tolerate the reducing conditions that develop under these circumstances. In polishing wetlands with very high water quality, a diverse species composition may develop.

## Natural Recruitment

One option is to allow for natural colonization of the wetland basins. Natural colonization of wetland restoration and establishment projects can be a highly successful method of establishing vegetation if sources of seeds and plant propagules are nearby. Reinartz and Warne (1993) reported finding 142 species of vascular plants in naturally colonized created marshes of southeastern Wisconsin. They found that the diversity and richness of native wetland plants and the proportion of total plant cover that was comprised of native marsh plant species increased from one- to three-year-old wetlands. The diversity and richness of native wetland species increased with proximity to the nearest native marsh with a marked decrease in

species richness beyond 700 m to the nearest marsh. These results indicate that sources of seeds of wetland species must be relatively close to the project site for dispersal of a diversity of species in adequate quantities to vegetate a site. The decision of whether adequate natural sources of seeds or vegetative propagules are available depends on the type of desired plants. For example, marshes containing desirable species that occur within 500–700 m of the project site are likely to be good seed sources of herbaceous plants, assuming the presence of dispersal vectors for the various seed types. Seed densities of windblown or animal-dispersed species decline rapidly with distance from the source. Water-dispersed seeds can be carried great distances, presumably for many kilometers, but the sources must be upstream of the project site.

It is possible to supplement the natural importation mechanisms by use of rooting soils that contain wetland plant seeds. This approach to establishing a wetland plant community is to harvest the seed bank from a neighboring wetland (natural or constructed) that has a plant community similar to the one desired for the new constructed wetland. This seed bank is harvested by scraping the top 10–20 cm of topsoil or muck from the donor wetland, and then redistributing this muck in strips or over the entire surface of the new wetland. An adequate moisture regime, resulting in saturated but not flooded soil conditions, is absolutely essential for the success of this approach. It is difficult to predict the development of vegetative cover from a transferred seed bank. Sometimes seed bank studies in greenhouses provide reasonable forecasts, but this is not always the case. The postrecruitment vegetative composition and structure from a seed bank cannot be predicted. Although many site factors, such as erosion, inundation, and drying, will affect emergence, results of a seed bank study can be used with caution to estimate the types of vegetation that will emerge on the project site.

Natural recruitment begins with the sprouting in the spring season, provided that moist soil conditions prevail (see Figure 18.35). The key to encouraging development of the existing seed bank is to maintain moist soil conditions, not deep-water conditions.



**FIGURE 18.35** This stand of cattails developed without assistance in a wet, fallow corn field near extensive *Typha* marshes associated with Wind Lake, Wisconsin. The row of dead corn stubble is seen in the foreground, proceeding from left to right.

In tropical climates, this process can be complete within one growing season but may require two or more seasons in colder temperate climates. In the event that the project schedule does not permit the time needed for natural recruitment, wetland vegetation can be established using any one of several types of propagules. A one-year delay in the imposition of permit requirements allows for natural regrowth and can save a considerable amount of money.

### Plant Propagules and Sources

When considering planted wetlands as a treatment option, designers must take steps to ensure that planting stock will be available in the quantity and quality needed for the project at a cost-effective price. Options for plant materials include:

- *Commercially available planting stock*. This is the most common method of obtaining planting stock for small to mid-size FWS projects; large projects may require contract growing with a commercial nursery.
- *Donor plants from another treatment wetland*. This option is often chosen for small projects when the plant species involved are very easy to propagate, e.g., common reed (*Phragmites australis*) or reed canary grass (*Phalaris arundinacea*).
- *Onsite plant nursery.* For large projects, a plant nursery can be created at the project site to provide the required planting materials.
- *Natural wetlands.* Field-harvesting materials from roadside ditches and native wetlands are labor intensive if done manually; often, plant collection permits are required.

Propagation and sale of wetland plant species has become a big business in several areas of the United States, and in Great Britain and Europe. Wetland plant nurseries supply thousands of plants that are used to renovate altered landscapes, such as phosphate and coal mining areas, and to create landscaped wetlands and ponds (aquascaping) for wildlife habitat. Although most of these plants are currently being propagated for use in habitat creation projects, this market has attracted suppliers who can propagate the types and quantities of plants required for constructing large wetland treatment systems. Plant propagules that are frequently used to establish constructed wetlands for wastewater treatment include seeds, bare-root seedlings, rhizomes, greenhouse-grown potted seedlings, and field-harvested plants.

If the wetland is to be planted, the cost and availability of plant materials must be addressed early in the design process. If suitable planting stocks cannot be obtained from local nurseries, the option of establishing an on-site wetland plant nursery must be decided very early because mature plants (one to two years old) are preferred. Mature plants have the energy reserves necessary to survive transplanting

and extend leaf tissues above the surface of the water, whereas very young plants may not.

### Seeds

Wetland plants can be established directly from seeds, given suitable seed stock, soil moisture, light, and temperature conditions. Seed germination is highly dependent on the soil/seed contact and the moisture regime of the soil. It is preferable to establish wetland plants using a native soil-rooting substrate. Some species have seeds that can be field harvested in very large numbers, whereas others produce few seeds. For example, a typical cattail seed head contains thousands of individual seeds, whereas mature bulrush culms may contain only 20–30 seeds each. Also, seed viability varies widely among plant species. If sufficient quantities can be obtained, most wetland seeds can be applied by using rotary seeders (or applied by hand) and then lightly raked into the surface soil layer. Broadcast methods have a disadvantage due to the development of areas that have poor cover, which may subsequently be dominated by undesirable species. Establishing new wetlands from seed has the advantage of lower initial cost. Seeds can generally be harvested and planted for a considerably lower cost than bare-root seedlings. However, it takes considerable time to establish a viable plant community that will provide adequate cover to allow system start-up and initiation of water quality treatment. Because bare-root seedlings pass their first several months of growth in greenhouse conditions, they have a comparable time advantage over seed germination in the constructed wetland.

### Bare-Root Plants

Both herbaceous and woody wetland plants may be provided as bare-root plants. Bare-root seedlings are typically planted in shallow, hand-dug holes. Their survival rate is significantly higher than for field germination of seeds and can generally be maintained at 80% or better with healthy plant stock and an adequate moisture regime. Initial success is largely dependent on the size of the bare-root plants; plants with small rootstock typically lack the energy reserve necessary to survive the transplanting process. To produce bare-root seedlings, nurseries plant germinated seeds in flats containing potting soil. The flats may hold dozens of individual plants that are later thinned. Bare-root seedlings are harvested from these flats after a few months when they have reached a height of 20–50 cm. Their roots are cleaned of potting soil, and the plants are wrapped together with moistened paper towels for temporary storage and eventual transportation to the planting site.

Bare-root seedlings are easily planted in the field in shallow individual holes prepared with a shovel, trowel, spike, or dibble. The survival rate of bare-root seedlings is significantly higher than for field germination of seeds, and can generally be maintained at 80% or higher with healthy plant stock and an adequate moisture regime. Use of bare-root seedlings at sufficiently close spacing can insure rapid establishment of a dense, monospecific plant cover in a short time. This success is critical for projects where minimal start-up time is of the essence.

### Rhizomes

Herbaceous wetland plants may be obtained as rhizomes or tubers. Rhizomes may be shipped to the project site while still dormant; plants that have broken dormancy often have the stems clipped back to facilitate shipment and planting. Larger rhizomes have more stored energy available to support plant establishment. Sprouting is typically from nodes along the rhizome, and cuttings should have at least two undamaged nodes. They are planted in moist soil.

The energy stored in the rhizome provides the energy to get the plant started; larger tubers or rhizomes offer a much better chance of success than small stock. Rootstock in the form of tubers should be at least 1.5 cm in diameter. Rootstock in the form of rhizomes should be at least 5 cm long. Plant count should be based on the number of roots or tubers, not the number of buds or shoots, because each tuber may have multiple buds or shoots.

### Potted Plants

To establish planting materials with a developed root structure, seedlings can be planted and grown in containers filled with potting soil. Potted plants have a greater advantage for initial growth over seeds and bare-root seedlings, primarily because they have larger stored energy reserves (if properly sized), and damage to the root structure during planting is minimal. However, the disadvantage of potted plants is the high initial cost. They are not only more expensive to grow but also expensive to transport to the newly constructed wetland.

Potted plants should be in at least 10-cm pots. The bound soil/root mass should be the same size as the container (otherwise, the product could be a 5-cm plant in a 10-cm pot). Plants in peat pots should be well rooted through the sides and bottom of the pot, and firmly anchored. If plants are growing at the time of installation, they should appear healthy with no leaf spots, insect damage, or wilting.

### Field-Harvested Plants

In some cases, herbaceous wetland plant stock can be field harvested and used to plant a newly constructed wetland. This planting technique is used in areas with abundant natural wetlands and high regional water tables, where wetland plants such as cattails and bulrush are common in roadside ditches, man-made ponds, and along canals. A plant collection permit may be required, depending on local regulations. Existing treatment wetlands are another potential source for planting stock. Field harvesting can occur by hand harvesting or by using a backhoe to scoop wetland plants from the ground and spread them on an open, upland area where they are separated by hand into manageable units. This method tends to be the most labor-intensive of any of the plant stock sources.

Field-harvested plants have several advantages over nursery grown stock. Because herbaceous wetland plants store most of their growth reserves in their roots, rhizomes, corms, and tubers, these plant parts can produce shoots and mature

plants faster than newly developing seedlings. In fact, a carefully tended rhizome generally produces numerous daughter plants in a few weeks, each of which in turn produces new rhizomes and a second generation of daughter plants. This growth results in faster development of adequate plant cover so that water quality treatment can begin earlier than otherwise possible. A second advantage of using plants that have been locally harvested is that these plants are more likely to be adapted to local climatic conditions, including their phenology and genetic tolerance to temperature and moisture regimes.

Invariably, other wetland plant species are introduced into a newly constructed wetland in the rootstock and attached soil of field-harvested plants. These additional plant species come in as small seedlings or as seeds in the existing seed bank from the donor wetland. For example, at the West Jackson County, Mississippi, constructed wetland near Ocean Springs, use of field-harvested cattails and the existing seed bank in the site soils resulted in the presence of 48 different plant species growing in the constructed wetlands one year after construction. This introduction of volunteer plant species is generally an advantage because it provides diversity in a constructed wetland that might otherwise be nearly monotypic.

### Planting Density

Only a small fraction of the ultimate plant density is planted in a new wetland. Planting densities range from 1,000–40,000 plants per hectare, depending on the rate of spread of the selected plant species in the climatic region, and the acceptable time frame and economics of plant establishment. As a reference, one plant per square meter, (a one-by-one meter spacing), corresponds to 10,000 plants per hectare. Through vegetative reproduction, these plants will eventually spread to much greater densities. The counting process after vegetative development usually involves stem counts rather than plant counts with densities of 100–800 stems per square meter being common. As the first round of plants mature and die, rhizomes send up new shoots. Consequently, the rate of rhizome spread is a factor that should be considered in design because rapidly-spreading species will eventually occupy much larger areas within the wetland than slow-spreading species, even if the initial planting quantities were the same. For instance, *Phragmites* and *Typha* are capable of spreading 10–20 m/yr (see, for example, Figure 3.12). Some species, such as *Schoenoplectus*, spread concentrically with ever-increasing clump diameters.

The potential for plants and seeds to grow and increase their coverage on the project site depends on several factors. If hydrology, nutrients, competition, and herbivory are not limiting to plant growth, it is not uncommon in wetland projects for rhizomatous grasses, other herbs, and some shrubs planted to less than 5% coverage to reach 100% coverage in less than three years by spreading vegetatively.

### Planting

After delivery to the job site, plant material should be stored in the shade and kept cool and moist. Plant stock left out in the sun can dry out and be useless within a few hours. Most wetland plant species have extremely poor drought tolerance. Field-harvested plants must be protected from excessive heat or cold, kept moist but not wet, and must be used within a few days to minimize mortality.

Planting is usually done by hand on small to moderate projects. Attempts to plant field-harvested rhizomes by mechanical planting have had varied success. Holes may be prepared with a shovel, a posthole digger, or a hand or tractor-mounted auger. In all cases, the hole must be deep enough to allow the rhizome or root material to be completely buried without excessive folding, and with the node between rhizome and shoot at the ground surface. Planting field-harvested plants is more difficult than planting nursery-grown seedlings because of the size of the propagules. Rhizomes and attached shoots and leaves are usually larger than bare-root seedlings and require a larger hole for planting.

Plants have a growing season and are dormant for the rest of the year in temperate and subtropical climates. Annual plants die in each fall or winter and must be reestablished from the seed in the following spring or summer. Fortunately, most plants used in treatment wetlands are perennials that lose their aboveground tissues and become dormant during the cold season, and regrow from storage reserves in belowground tissues during the next growing season. In some cases, new growth is timed to take advantage of predictable rainfall and moisture conditions rather than specific day length or temperatures. Knowledge of these climatic influences is important in planting specifications and during wetland planting.

The best time to establish new plants in a constructed wetland is in the beginning or height of the growing season, which usually coincides with spring or early summer. During this period, available light is increasing daily, and competition from previously established weeds or damage from pests and pathogens is minimal. Most plants are genetically adapted to grow during this period. Also, planting at the beginning of the growing season provides ample time for full plant development and attainment of suitable plant cover before the onset of cold temperatures and declining plant growth rate. A wetland treatment system can typically begin operation when the plant cover is at least 60–80%. If this percent cover can be obtained during the first growing season after planting, the wetland system can begin operating during the ensuing fall.

### Plant Establishment

The best technique for establishing rapid plant cover is to maintain saturated soil conditions without surface flooding. The higher soil oxygen conditions resulting from the absence of flood waters allow maximum root metabolism, effective nutrient use, and rapid development of the original plant propagules and their daughter plants within the wetland. This soil condition should optimally be maintained until the plants achieve complete cover (100%) or at least the minimum cover required for system start-up (about 60–80%).



**FIGURE 18.36** After planting, the Columbia, Missouri, wetland was allowed to dry because of disagreements about responsibility for watering. The initial planting was lost.

Once the soil is in place and the plants have been planted, water may be increased gradually to match plant growth. The best method to maintain soil saturation without excessive flooding is to start planting at the downgradient end of the wetland and continue planting upgradient while gradually raising water levels using the wetland outlet water level control weirs. When planting is complete, water levels can be dropped or raised as needed to maintain saturated soil conditions.

Incorrect control of soil moisture is the most frequent cause of failure to establish wetland plants (Figure 18.36). Inadequate soil moisture results in desiccation of roots and shoots, and causes wetland species to be replaced by weedy upland plant species that may be in the seed bank of the constructed wetland soils. Too much water results in oxygen depletion in the root zone and slow growth or plant death because of insufficient oxygen for root metabolism—otherwise known as drowning. The correct amount of moisture can be maintained through adequate planning and attention during the construction period.

To maintain suitable soil moisture during plant establishment, there must be a reliable and adequate supply of water for site irrigation. This requirement can be quite large when constructing wetland systems more than a few hectares in size. When practical, the best water source for site establishment is usually the wastewater that will be treated in the wetland. This water can generally be used if an existing treatment system is being retrofitted or expanded by the addition of a constructed wetland. If the wetland influent water is not available, then another irrigation supply must be provided. This irrigation source might be a potable water supply, an existing or temporary well, a pond or lake, or a nearby stream or river. Adequate pumps, piping, and sprinklers or hoses must be provided to allow even-flow distribution.

Design specifications for vegetation establishment in constructed wetlands should clearly delegate responsibility for plant maintenance from the time of initial planting through plant establishment. Successful establishment requires periodic inspections to document soil moisture conditions, plant survival, and plant growth. The frequency of these inspections is project-specific but must be often enough to deal proactively with potential problems.

Initial plant inspection should examine the viability of the planted propagules, whether seeds, seedlings, or field-harvested mature plants. Subsequent plant growth is monitored by estimating percent cover and average plant height. These nondestructive techniques are used to ascertain the status of plant development before and during wetland treatment system operation. Plant cover is an estimate of the percentage of the total ground area covered by stems and leaves. This parameter can be estimated by walking through or next to a plant stand and visually determining a cover category for the plants.

A controversial aspect of plant establishment is dealing with wildlife grazing. FWS wetlands can attract flocks of waterfowl, such as Canadian geese (*Branta canadensis*), which can obliterate planting stocks if host populations are high enough (Figure 18.37). Successful plant establishment



**FIGURE 18.37** The Hayward, California, wetland experienced great problems with waterfowl grazing. The only plantings that remain in this photo were in exclosures built to prevent such eat-out.

requires dealing with these issues in a proactive manner, via acceptable methods of animal control (screening, deterrents, etc.).

## SUMMARY

The physical design of a FWS wetland requires numerous decisions concerning siting and layout. The compartmentalization and number of flowpaths are specified, as well as the aspect ratios of the cells. The internal design of a cell involves possible use of deep zones as well as patterns of vegetation that enhance treatment. All such decisions are subject to the constraint that the wetland must pass the design flows without excessive head loss and inlet flooding. The height of the berms and depth of excavation are balanced so that material is neither exported nor imported to the site. The type and thickness of a liner are chosen, if one is necessary. Structures are selected to control the depth of water in the wetland, and to spread and collect the water efficiently.

As long as a few simple guidelines are followed and common sense is employed by contractors and field engineers, wetland project construction can help insure the success of a good design. Site civil construction activities, pump stations, forcemains, and open-channel conveyance systems are standard experience for most wetland contractors. For wetland systems, it is important to not create conditions that will ultimately limit the wetland treatment system from accomplishing its water quality treatment or habitat creation goals.

The presence of a healthy, dense population of emergent plants is an important requirement for pollutant assimilation in wetland treatment systems. Plants provide the structural surface area occupied by the microbial flora that attack dissolved pollutants; they provide the physical filtration that reduces particulate pollutants; and they shade the water surface, preventing algal regrowth, and dampening temperature fluctuations. In addition, the wetland plants provide a vast amount of fixed organic carbon important for microbial growth and denitrification and are the basis for a complex food web of wetland biota.

Wetland plants can be established successfully by a number of different methods that have varying benefits and limitations. Designer preference and planting cost will guide selection of the appropriate plant propagation technique. Care must be taken with the health of the plant propagules, soil and site preparation, planting, and soil moisture maintenance. The time required for vegetation establishment is often critical to meet scheduled system start-up.

# 19 Ancillary Benefits

When wetlands are used to reduce pollutant concentrations and peak stormwater flows, ancillary benefits can be achieved through thoughtful site selection and design. This chapter summarizes the ancillary benefits of wetlands used for pollution control and recommends design features to optimize these benefits. The primary objective of most wetland pollution control projects is water quality enhancement through assimilation and transformation of sediments, nutrients, and toxic chemicals. Secondary benefits that can be incorporated in wetland treatment designs include (1) vegetative biodiversity; (2) protection and production of fauna; and (3) aesthetic, recreational, commercial, and educational human uses. Realization of these benefits usually requires extra land and extra expenditures. Some of the features that are added are illustrated in Figure 19.1. The potentials for including each of these benefits in treatment wetlands are described in the following text. Research and manuals on wetland evaluation techniques (Golet, 1978; Greeson *et al*., 1978; Richardson, 1981; Kusler and Riexinger, 1986) may be used to compare the additional benefits of treatment wetlands with natural wetland functions.

## 19.1 VEGETATIVE BIODIVERSITY

Because of the presence of ample water, the wetland environment is generally characterized by a high diversity and abundance of plants. In many cases, wetland plant communities include multiple vertical strata, ranging from groundcover species to shrubs, and subcanopy trees to canopy tree species. Wetland plant diversity is important in determining wildlife diversity because of the creation of niches associated with differing vegetative structure, reproduction strategies, flowering and seeding phenologies, gross productivity, and rates of decomposition (Mitsch and Gosselink, 2000b).

The wetland treatment system designer should not expect to create or maintain a system with just a few known species. Such attempts frequently fail owing to the natural diversity of competitive species and the resulting high management cost associated with eliminating competition, or because of imprecise knowledge of all the physical and chemical requirements of even a few species. Rather, the successful wetland designer creates the gross environmental conditions suitable for groups or guilds of species; seeds the wetland with diversity by planting multiple species, using soil seed banks, and inoculating from other similar wetlands; and then uses a minimum of external control to guide wetland development. This form of ecological engineering results in lower initial cost, lower operation and maintenance costs, and a most consistent system performance.

This section presents an overview of the floristic diversity that naturally develops in treatment wetlands as well as some details of the growth requirements of commonly occurring plant species in wetland treatment systems. These microbial and plant species are typically the dominant structural and visual components in treatment wetlands. An understanding of their basic ecology will provide the wetland designer or operator with insight into the mechanics of their "green" wastewater treatment unit.

### Wetland Plants

#### Algae

There are multiple designations of the algal component of wetland vegetation. The term *periphyton* describes the community of organisms that grows attached to emergent and submerged plants. Although periphytons usually begin colonization of new plant surfaces by attaching algal growth of filamentous and unicellular species, this functional component also includes mat-forming assemblages of algae, fungi, bacteria, and protozoans. Benthic algae are attached to the sediment–water interface. Planktonic or free-floating algae are generally not a large component of wetland ecosystems unless open or deep water areas are present. Plankton spend most of their life cycle suspended in the water column and are of potential concern as a source of total suspended solids (TSS).

Algae are often the first colonizers in sparsely vegetated, newly constructed wetland basins. They are capable of rapid colonization, and for some months may outcompete the macrophytes. Eventually, emergent plants will fill in, block the light, and reduce the algal populations.

Algae are only rarely selected as the dominant plants in treatment wetlands. Exceptions include the technology known as *algal turf scrubbing* (Adey *et al*., 1993; Craggs *et al*., 1996; Craggs, 2000) and the use of periphyton, known in Florida as *periphyton stormwater treatment* (Vymazal, 1989; Kadlec and Walker, 2004). Algae are not the target of attempts at planting or management and would not be considered a major factor in system diversity.

#### Macrophytes

The term *macrophyte* includes vascular plants that have easily visible tissues. A wide variety of macrophytic plants occur naturally in wetland environments. The U.S Fish and Wildlife Service has more than 6,700 plant species on its list of obligate and facultative wetland plant species in the United States. Godfrey and Wooten (1979; 1981) list more

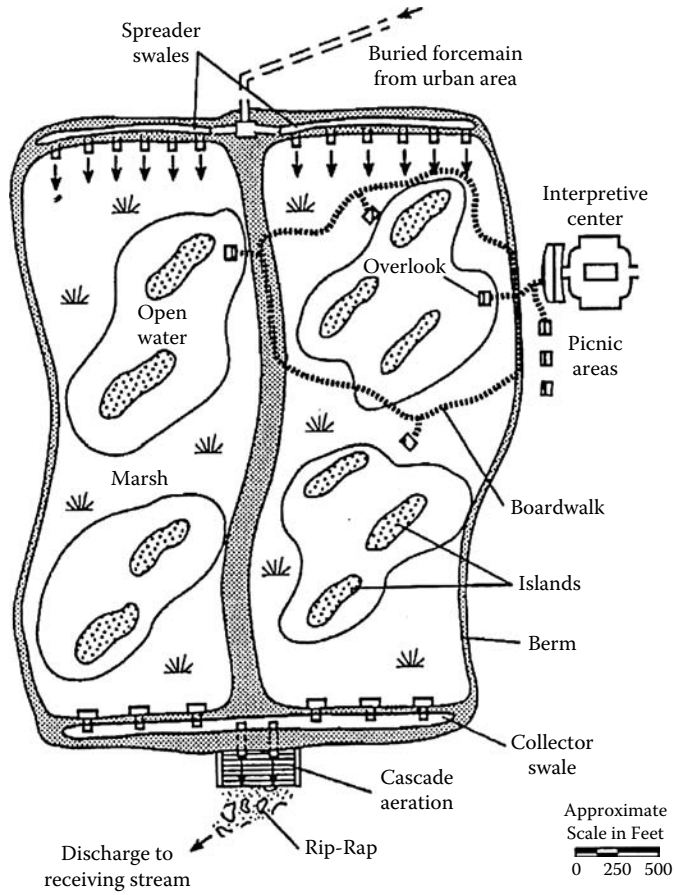



**FIGURE 19.1** Conceptual plan for treatment wetlands with ancillary benefits. (From Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

than 1,900 species (739 monocots and 1,162 dicots) of wetland macrophytes in their taxonomy of the southeastern United States. Obligate wetland plant species are defined as those that are found exclusively in wetland habitats, whereas facultative species are those that may be found in upland or in wetland areas. There are many guidebooks that illustrate wetland plants, for example, Hotchkiss (1972) and Niering (1985). Lists of plant species that occur in wetlands are available, for example, RMG (1992).

Wetland macrophytes are the dominant structural component of most wetland treatment systems. The terms *emergent*, *floating*, and *submerged* refer to the predominant growth form of a plant species. In the emergent plant species, most of the aboveground part of the plant emerges above the waterline and into the air. These emergent structures are generally self-supporting. Emergent wetland plant species are the primary concern in this section because they provide the dominant visual impression in wetland treatment systems. Floating and submerged vascular plant species may also occur in wetland treatment systems. Floating species have leaves and stems buoyant enough to float on the water surface. Submerged species have buoyant stems and leaves that fill the niche between the sediment surface and the top of the water column. Floating and submerged species are more typical of deeper, aquatic habitats within the wetlands, and they may dominate in wetlands when water depth exceeds the tolerance range for rooted, emergent species.

## Water Regime

The hydropattern, or water regime, of the water is the most important contributor to the wetland type or class (Gosselink and Turner, 1978; Gunderson, 1989). The concept of water regime includes two interdependent components: (1) the duration of flooded or saturated soil conditions (the hydroperiod as a percentage of time with flooding), and (2) the depth of flooding (Gunderson, 1989). Whereas *hydroperiod* refers to the duration of flooding, the term *water regime* refers to hydroperiod as well as to the combination of water depth and flooding duration (depth–duration curve). The importance of this factor as a determinant of wetland ecology cannot be overstated, because an incorrect understanding of the hydroperiod and water regime limitations of wetland plant species is the most frequent cause of vegetation problems and shifts in constructed wetlands.

Plant survival in flooded environments is a balance between the severity of oxygen limitation and the adaptations available to overcome this oxygen shortage. Thus, hydrophytic plants may be adapted to survive and flourish

in specific flooded conditions, for example, for three months each year, or in "clean" or flowing water, which might have higher *in situ* dissolved oxygen concentrations (Gosselink and Turner, 1978). However, these same plants may not be able to grow or survive during five months of flooding or in stagnant or "dirty" water conditions. Likewise, plants may have adaptations that allow prolonged survival in 0.3 m of water, but not in 0.6 m. It may be hypothesized that this balance is tilted unfavorably at higher water levels because of reduced aerial plant stem surface area to provide oxygen to the roots through the lenticels and aerenchymous tissues. This proposed explanation is supported by the finding that hydrophytes generally respond to flooding by growing taller—a growth response that allows a more favorable balance between emergent and submerged plant organs (Grace, 1989).

## Propagation

Wetland plants increase their numbers and density through asexual and sexual reproduction. Asexual reproduction refers to an increase in the number of individuals of a plant species through vegetative growth and typically occurs through the growth of roots or rhizomes, with subsequent emergence of new aboveground stems and leaves. Technically, a cattail bed that developed vegetatively from a single parent plant is a single plant. However, when these rhizomes are cut or decay, the individual daughter plants may remain viable and continue to spread vegetatively. A number of woody wetland plant species can spread vegetatively through coppice growth from viable root systems.

Most wetland plant species also may increase their numbers through sexual reproduction. In sexual reproduction, two individual plants, or the male and female flowers from a single plant, contribute gametes to form seeds with new combinations of genetic material. Sexual reproduction is important in providing alternative strategies for plants to survive from year to year through seasonal extremes, to propagate the species over large distances, to rapidly colonize new habitats, and to provide genetic variants that can adapt to changing environmental and competitive conditions.

Because of the potential for year-to-year hydrological variations in natural herbaceous wetlands with large numbers of annual plant species, many species produce seeds that remain viable for years. These seeds accumulate during productive years and remain dormant until conditions are favorable for germination, frequently after a period of desiccation and rewetting. This storage of viable seeds is known as a seed bank and has been studied in a number of marsh ecosystems (Pederson, 1981; Leck *et al*., 1989). In some cases, soil from a natural wetland with a seed bank can be used to establish a new constructed wetland. Most seeds in seed banks are annuals, but in some marshes, up to 50% may be perennials. Numbers of seeds range from <100/m$^2$ to >375,000/m$^2$ (Leck *et al*., 1989). A typical freshwater marsh in Manitoba had 4,582 seeds/m$^2$ and 34 species in emergent areas and only 93 seeds/m$^2$ in open water areas (Pederson, 1981).

## Biodiversity

Constructed treatment wetlands are typically dominated by emergent marsh, floating aquatic plant, or submerged aquatic plant communities. Emergent marsh species are frequently intermingled and codominant with populations of small floating aquatic plants such as duckweed (*Lemna* spp.). Most constructed treatment marshes in the United States are dominated by cattails (*Typha* spp.) or bulrush (*Scirpus* spp.) and, in Europe, by common reed (*Phragmites australis*) or grasses *(Phalaris arundinacea* or *Glyceria maxima*); however, many treatment marshes are dominated by other species or by a complex admixture of plant species, which include cattails, bulrush, grasses and/or reed as the dominant species. These plants are selected because they are robust and can survive in a wide variety of environmental conditions—in other words, they are survivors. They are inexpensive, plentiful in supply, and easy to propagate. However, they may also be viewed as a nuisance species owing to their ability to crowd out other species and assume visual dominance.

In the United States, species such as *Phalaris arundinacea, Lythrum salicaria, Typha* spp., and *Phragmites australis* are considered by many to be undesirable and aggressive invaders. These may take over natural wetlands and also be a component of treatment wetlands. Pollutant removal processes do not require these species. They are often selected on the basis of cost and survivability. Unfortunately, many people now erroneously equate free water surface (FWS) treatment wetlands with the low biodiversity and nuisance species.

Many constructed treatment wetlands undergo plant succession during their operational life. Constructed marshes tend to remain as marshes as long as flooding is nearly continuous and water depths exceed about 5 cm. However, if the water is too deep, then some emergent species will give way to submerged and floating plants. In any case, new plants find their way into the system. For instance, Schwartz *et al*. (1994) found numerous volunteer species in a Florida constructed wetland. The system was established by planting a diverse set of 21 species. After one year, 185 species were found in the system. A side-by-side study of wetland vegetation changes was conducted at the Olentangy River wetlands site in Ohio (Mitsch *et al*., 2004). As seen in Table 19.1, the effect of the initial planting strategy (planted or unplanted) was gone in three years. The volunteer cattails were then eaten out by muskrats and replaced by volunteer bulrushes in both wetlands. The Tres Rios, Arizona, wetlands were planted in an elegant mosaic with *Scirpus validus* (soft-stem bulrush) and *Scirpus olneyi* (three-square bulrush), having varying numbers of deep zones and islands (CH2M Hill, 1995). During the spring and summer of the third year after startup, almost all the vegetation died in all four wetlands for reasons that have not been fully resolved. The three possible causes were drowning, starvation, and sulfide toxicity (Kadlec, 2006b).

Thus, it is probable that the species patterns and types set by ecological or botanical designers will not survive in many circumstances. Such "designer" wetlands are likely to evolve

**TABLE 19.1**
**Progression of Two Side-by-Side Wetlands at the Olentangy River Wetlands Park over a Start-Up Period of Nine Years**

| | Wetland 1 (Planted) | Wetland 2 (Unplanted) |
|---|---|---|
| Year 1 (1994) | Algal pond with few macrophytes | Algal pond with few macrophytes |
| Year 2 (1995) | Plants developed, particularly around the perimeter, to about 13% macrophyte cover | Essentially no macrophyte cover |
| Year 3 (1996) | Continued to develop in vegetation cover, with about 39% cover | Developed to about 35% macrophyte cover by August, essentially catching up with the planted wetland within 3 growing seasons |
| Year 4 (1997) | About 54% cover | About 58% cover |
| Year 5 (1998) | Wider diversity of cover; not dominated by *Typha* | Began to be dominated by *Typha* |
| Year 6 (1999) | Dominated by *Typha* | Dominated by 3–4 of the planted species |
| Year 7 (2000) | Similar to 1999, except muskrats began to have a dramatic effect | Similar to 1999, except muskrats began to have a dramatic effect |
| Year 8 (2001) | Maximum muskrat impact, vegetation cover was lower than 1995 | Maximum muskrat impact, vegetation cover was lower than 1995 |
| Year 9 (2002) | Drawdown from April through June; developed cover of *Schoenoplectus tabernaemontani* | Drawdown from April through June; developed cover of *Schoenoplectus tabernaemontani* |

*Source:* Data from Mitsch *et al*. (2004) In *Olentangy River Wetland Research Park at The Ohio State University, Annual Report 2004*. Zhang and Tuttle (Eds.), Columbus, Ohio, pp. 59–68.

into other species mixtures and patterns. Wetlands that rely on antecedent seed banks and natural importation will develop more slowly, but are more likely to be regionally adapted. "Self-designed" wetlands may not be avoidable. It seems reasonable that new wetlands should be established with a species mix that provides a headstart for a number of regional species that can tolerate the water quality and hydropatterns deemed proper for water quality improvement.

Table 3.1 summarizes the number of plant species reported for the treatment wetlands in the North American Treatment Wetland Database, v.2 (NADB database, 1998). Almost 600 species of macrophytic plants have been reported in constructed treatment wetlands. Of these observed plant species, 501 are emergent herbaceous macrophytes, 31 are floating aquatic species, 12 are submerged aquatics, 17 are shrubs, 25 are trees, and 5 are vines. While the NADB v.2 data collection effort provides some insight into the plant diversity at constructed treatment wetland sites in North America, the level of investigation varied widely between sites, and these numbers should be considered as the minimum number of plant species that actually occur in treatment wetlands.

Wetland plant diversity is a poorly understood subject both in unaffected natural wetlands and in treatment wetlands. Wetlands are frequently dominated by a few plant species (for example, cattail, sedge, or sawgrass marshes) that are best adapted to stressful environmental conditions, such as low nutrient or soil oxygen levels and fluctuating water levels. Other unaffected natural wetlands have higher plant diversity and greater evenness between multiple dominant plant species. Constructed treatment wetlands cover the same range of plant dominance and diversity as unaffected natural wetlands. Information collected for NADB v.2 indicates that hundreds of plant species occur in a variety of treatment wetlands. Even when treatment wetlands are dominated by cattails or bulrush, dozens of other herbaceous and woody plant species are typically present. Thus, the issue is not so much of diversity as it is of dominance.

It is possible to start the treatment wetland with a variety of regionally prevalent plants so that these have an opportunity to compete during the successional development of the system. This philosophy has been implemented at a number of treatment wetlands, including 21 species planted at the Orange County Eastern Service Area in Florida (Schwartz *et al*., 1994), 20 species at Victoria, Texas (Reitberger *et al*., 2000), and 14 species at Wakodahatchee, Florida (Bays *et al*., 2000). The Oregon Gardens treatment wetland used dozens of native wetland plants (Figure 19.2), and the Lapeer, Michigan, wetland used four species that did not include treatment wetland species, although bulrushes were used on the fringes (Figure 19.3).

## 19.2 WILDLIFE

Numerous wildlife species of all taxonomic orders depend on wetlands as habitat. Plant structures that are lost into the aquatic portion of the food chain are typically degraded by a complex assemblage of small aquatic organisms that include invertebrates (protozoans, worms, molluscs, arthropods, and others). These organisms, in turn, serve as the basis of the food chain for other invertebrates, and for diverse vertebrate groups such as fish, amphibians, reptiles, birds, and mammals. Above-water plant materials may also be processed as food by nonaquatic fauna that have no direct contact with the water in the wetland.

We know from qualitative observation that a diversity of microscopic invertebrates such as protozoans and



**FIGURE 19.2** The Oregon Gardens treatment wetland, near Silverton, Oregon. A tremendous diversity of vegetation was introduced. The wastewater is better than secondary at the wetland inflow.

microarthropods is present, but there are no comprehensive lists of even the most common species in these groups. The macroinvertebrate groups most commonly studied are annelids (worms), arachnids (spiders and mites), crustaceans (amphipods, isopods, copepods, water fleas, and crayfish), insects, and molluscs. Fish are an obvious component of most aquatic ecosystems, but knowledge of their role in wetlands is limited. Amphibian and reptile (herptile) populations in wetlands tend to be variable and secretive and, therefore, difficult to quantify. Birds are extremely mobile, and many species include wetlands in some part of their life history. Because birds are one of the fauna groups that people appreciate the most, study of this wildlife group in treatment wetlands has been more extensive than for any other. Many mammal species use wetlands as an important component of their total habitat. Some depend greatly on wetlands, whereas others use wetlands for specific and narrow life history requirements.

### Macroinvertebrates

Information concerning wetland macroinvertebrate populations is infrequent in the published literature. A number of specific findings from wetland treatment systems are described in the following text to provide an indication of typical macroinvertebrate population densities and diversities in these systems.

A total of 518 species of aquatic invertebrates have been recorded from treatment wetlands in NADB v.2 (Table 19.2). These include 14 species of aschelminthes (cavity worms), 55 species of crustaceans, 10 species of arachnids, 23 species of molluscs, and 343 species of insects. Thirty-two treatment wetland systems listed in NADB v.2 have invertebrate data, although, in most cases, only species lists are available. The average Shannon-Weiner diversity (H') of benthic macroinvertebrate collections was 1.36 units (with a total of 342



**FIGURE 19.3** The Durakon treatment wetland, near Lapeer, Michigan. Most of the plants selected for the wetland are not in common use in treatment wetlands (*Carex lacustris, Sagittaria latifolia, Alisma plantago-aquatica, Pontederia cordata,* and *Sparganium eurycarpum*). Only narrow fringe zones were planted with species with established treatment performance (*Scirpus* spp.) The wastewater inflow is lagoon effluent.

**TABLE 19.2**
**Number of Invertebrate Species by Group in the NADB v.2 Database for Constructed Wetlands**

| Benthos Subgroup | Number of Species Recorded |
|---|---|
| Annelida | 47 |
| Arachnida | 10 |
| Aschelminthes | 14 |
| Bryozoa | 1 |
| Coelenterata | — |
| Crustaceans | 55 |
| Entognatha | 2 |
| Insects | 343 |
| Mollusk | 23 |
| Platyhelminthes | — |
| Tardigrads | 1 |
| Unknown | 22 |
| **Total** | **518** |

*Source:* Data from NADB database (1998) *North American Treatment Wetland Database (NADB), Version 2.0.* Compiled by CH2M Hill. Gainesville, Florida.

species reported) for constructed treatment wetlands. Benthic macroinvertebrate evenness averaged 0.56 for constructed wetlands. Average benthic populations in the NADB v.2 are 6,083 individuals per square meter for constructed treatment wetlands. Total populations of mosquito larvae and pupae in treatment wetlands from a few projects had an average of 1,144 individuals per cubic meter for constructed treatment wetlands (Hemet and Sacramento, California).

## Fish

The number of fish families that typically occur in wetland environments in North America is small compared to pond, lake, and stream environments. The fish species typical of wetlands are either adapted to obtaining oxygen in low-oxygen waters or only visit wetlands on a seasonal or shorter basis. The most widely distributed fish species in warmer temperate, subtropical, and tropical wetland treatment systems is the mosquitofish (*Gambusia affinis*) in the topminnow family. This species has been widely disseminated in North America and abroad in shallow wetlands and ponds because of its noted ability to consume the larvae and pupae of mosquitoes. In more northern climates, the black-striped topminnow (*Notrophus fundulus*) is a mosquito predator, and the central mudminnow (*Umbra limi*) is an important wetland fish species.

Seventy-seven fish species are reported from 13 treatment wetland sites in NADB v.2 (70 species from constructed treatment wetlands and 15 species from natural treatment wetlands). Table 19.3 summarizes fish data by site and system in NADB v.2.

Fish enter treatment wetlands from inlet and outlet streams as well as from other sources, such as raptor drops. For example, smallmouth bass (*Micropterus dolomieui*) and sunfish (*Lepomis macrochirus*) swam up the outlet structures at the Hillsdale, Michigan, treatment wetland, and took up residence in all deep zones. Drawdown of the wetlands, for purposes of maintenance and retrofitting, killed the fish. They were effectively precluded after the deep zones were filled for muskrat control. Bass and sunfish also are found in the Incline Village, Nevada, constructed wetlands, which are abutted by natural wetlands with deep water habitats. Tilapia (*Oreochromis* spp.) found their way into the Tres Rios, Arizona, wetlands, although there appeared to be no route open to them from either downstream with entering water (WWTP) or upsteam from receiving water (drop boxes and cascades). Drawdown of the wetlands, for research purposes, killed the fish. Tilapia also are prolific in the Lakeland, Florida, constructed wetlands—again with no obvious route for them to have emigrated.

**TABLE 19.3**
**Number of Fish Species by System in the NADB v.2 Database for Constructed Wetlands**

| Site Name | System Name | Number of Species |
|---|---|---|
| West Jackson County, Mississippi | WJC System | 2 |
| Brookhaven, New York | Meadow Marsh Pond System 1 | 2 |
| Brookhaven, New York | Marsh Pond System 2 | 2 |
| Hayward, California | Hayward | 16 |
| Hillsboro, North Dakota | American Crystal Sugar Co. | 3 |
| Everglades, Florida | ENRP | 18 |
| Sacramento, California | Sacramento Demo Wetlands | 2 |
| Champion, Florida | Champion Pilot | 3 |
| Mt. View Sanitary District, California | Mt. View Marsh | 33 |

*Source:* Data from NADB database (1998) *North American Treatment Wetland Database (NADB), Version 2.0.* Compiled by CH2M Hill. Gainesville, Florida.

Fish may be problematic in treatment wetlands, particularly those that feed or nest on the bottom. For instance, at the Des Plaines constructed wetlands in Illinois, carp (*Cyprinus carpio*) were unintentionally introduced into the wetland from the pumped source water (Des Plaines River), presumably as very small individuals. Over the course of two years, the fish grew in size to about 15 cm. These larger fish were observed to stir bottom sediments and impair suspended solids removal. The carp were removed from the wetland through a winter drawdown and freeze. A similar phenomenon occurred at the Tarrant County constructed wetlands in Texas, where floodwaters from the source river brought carp (*Cyprinus carpio*), shad (*Dorosoma cepedianum*), and bullheads (*Ictalurus* spp.) into the system. The activity of the fish caused severe bioturbation of the sediments during feeding and spawning activities (APAI, 1994). The cells were dewatered, and egrets and herons effectively removed the stranded fish.

A 21-ha constructed wetland has been used for final polishing of an industrial effluent since 1999 in Victoria, Texas. Fish populations in the eight wetland cells have been used as an indicator of environmental condition at or near the top of the wetland aquatic food chain (Reitberger *et al*., 2000). This constructed treatment wetland includes three different types of cells arranged in parallel and in series. Treated effluent from a conventional activated sludge treatment facility is distributed into the stage 1 wetland cells, which are colonized primarily by cattails (*Typha* spp.) and contain transverse deep water zones colonized by submerged and floating aquatic plants. Water from these cells flows into the stage 2 cells containing a diverse marsh plant community and transverse deep water zones. Water flows finally into a habitat cell with diverse water depths (up to 3 m). Six fish species have been collected; mosquitofish (*Gambusia affinis*) have the highest density, and populations of other species are variable from year to year, presumably in response to climatic conditions and water quality conditions. Sunfish (*Lepomis macrochirus* and *Lepomis gulosus*), bluefin killifish (*Lucania goodie*), silversides (*Menidia beryllina*), and gizzard shad (*Dorosoma cepedianum*) populations are sporadic in the habitat cell.

### Amphibians and Reptiles

Amphibians and reptiles (known jointly as herptiles) are part of the top consumer structure of wetlands, including treatment wetlands. The four most important groups are frogs (anurans), turtles, snakes, and alligators. Twenty-one anuran species are reported from six constructed and three natural treatment wetlands in NADB v.2 (ten from the constructed treatment wetlands). Turtles are found primarily in wetland treatment systems that have areas of deeper, open water, and especially in pretreatment lagoons. Species of turtles frequently found in these habitats include snapping turtles (*Chelydra serpentina*), mud turtles (*Kinosternon subrubrum*), and painted turtles (*Chrysemys picta*). A wide variety of snakes are associated with wetlands and wetland treatment systems and play an important ecological role by feeding on fish, invertebrates, birds, and small mammals. In North American treatment wetlands, the snakes that receive the most interest are the venomous species, including the cottonmouth or water moccasin (*Agkistrodon piscivorus*), the copperhead (*Agkistrodon contortrix*), and timber rattlesnakes (*Crotalus horridus*), which occasionally inhabit wetlands. In fact, most snakes in wetlands are nonpoisonous species of water snakes (Genus *Natrix*), swamp snakes (*Liodytes* and *Seminatrix*), rat snakes (*Elaphe*), and king snakes (*Lampropeltis*). Alligators (*Alligator mississippiensis*) have been observed at relatively high densities in wetland treatment systems having open water areas and high fish populations (Figure 19.4). Densities in the Florida stormwater treatment constructed wetlands have reached proportions sufficient to allow hunting.

### Birds

About 600 different bird species (one third of the total resident bird species) are either partially or wholly dependent on wetlands for some part of their life history in North America (Kroodsma, 1978). The diversity and abundance of birds in and around wetlands attract many bird watchers, who repeatedly have observed that their species lists are longer and their counts higher when they include wetlands in their counting areas. Where these water bodies are enriched by nutrients and



**FIGURE 19.4** Alligators are at the top of the food chain in wetland systems across the American south. (Photo courtesy J. Bays.)

**TABLE 19.4**
**Results from Constructed Wetland Treatment System Bird Surveys by U.S. EPA during 1991**

| Site | Constructed Wetland Area (ha) | Total Species | Density (#/ha) [Average (Range)] |
|---|---|---|---|
| Incline Village, Nevada | 198 | 47 | 19.1 (0.8–42.2) |
| Show Low, Arizona | 284 | 42 | 13.8 (7.8–21.7) |
| Collins, Mississippi | 4.5 | 35 | 7.2 (5.9–8.5) |
| Ocean Springs, Mississippi | 22 | 35 | 10.4 (6.4–14.5) |
| Orlando Easterly, Florida | 494 | 141 | 0.29 (0.11–0.54) |
| Lakeland, Florida | 498 | 63 | 0.38 (7.7–13.5) |

organic matter from wastewater and stormwater discharges, bird watchers usually find even better success in their sport.

A number of wetland bird species are game species and are hunted. Treatment wetlands frequently provide good waterfowl habitat. In a few cases, these treatment wetlands have been used for duck hunting as a recognized secondary benefit. The stormwater treatment constructed wetlands in Florida are open to duck hunting and are deemed the best hunting spots in the entire state. In other cases, wastewater has been found to improve the waterfowl breeding and feeding habitat as a secondary or nonessential side effect (Wilhelm *et al*., 1989).

A few quantitative studies of bird populations in wetlands treating wastewaters are available. If the wetland is constructed on previous upland, increases in bird species and abundance have been observed, compared to the previous upland habitat (Hickman, 1994). During the summer of 1991, the U.S. EPA conducted a study of the environmental condition of six constructed wetland treatment systems in the United States (McAllister, 1992; 1993a; 1993b). This inventory included two systems in the arid west (Incline Village, Nevada, and Show Low, Arizona), two systems along the coastal plain of Mississippi (Ocean Springs and Collins), and two systems in peninsular Florida (Orlando Easterly and Lakeland). Bird surveys were conducted by local ornithologists using slightly different methods. Table 19.4 summarizes the major findings of this research.

The EPA bird surveys at constructed wetland treatment systems confirmed that these systems have high species richness and population densities compared to control wetlands. The total number of bird species observed at each site during the 1991 study ranged from 33 to 63, with average daily population densities of all wetland-dependent species ranging from about 7 to 19 birds/ha. Densities of wading birds at the two central Florida constructed wetland treatment systems averaged 0.29 birds/ha at Orlando and 0.38 birds/ha at Lakeland. These densities were as high as, or higher than, comparison data from marshes along the St. Johns River, the central Everglades, and Central America. Highest total bird densities were noted at the arid region sites where there are few natural wetlands available to compete as alternative habitats. The EPA studies also concluded that the constructed treatment wetlands are important habitats for a number of endangered or threatened wetland-dependent bird species.

Bird usage of the 16-ha DUST (Demonstration Urban Stormwater Treatment) marsh in Coyote Hills Regional Park near Fremont, California, was studied by Duffield (1986). Weekly or biweekly bird censuses were conducted in the DUST marsh and a nearby 18.3-ha control from mid-January 1984 through mid-June 1985. The mean abundance of wetland birds in the marsh areas ranged from 100 to 300 (6.25 to 18.75 birds/ha) in the DUST marsh and 90 to 420 (4.91 to 23.0 birds/ha) in the control marsh. The mean number of species per census ranged from 14 to 23 in the DUST marsh and 10 to 18 in the control area. Highest mean species counts were observed in both marshes during the winter and spring seasons. During all seasons, the dabbling and diving ducks preferred the control marsh, and shorebirds, gulls, and terns preferred the DUST marsh. Individual fish-eating wading birds were generally present in greater numbers in the DUST marsh than in the control area. These populations were enhanced by the presence of roosting black-crowned night herons (*Nycticorax nycticorax*). Shorebirds preferred the mudflat areas of the overland flow cell in the DUST marsh. The most common shorebirds were American avocets (*Recurvirostra americana*), black-necked stilts (*Himantopus mexicanus*), and marbled godwits (*Limosa fedoa*).

Bird populations at the Victoria, Texas, constructed treatment wetland have been quantified by designed studies and by birdwatcher counts (Reitberger *et al*., 2000). A total of 188 bird species had been observed using the wetland as of June 2000. Annual species numbers ranged from 106 to 144, with average numbers of species per count from 34 to 53; average annual bird densities ranged from 19 to 26 birds per hectare. Bird population numbers were dominated by grebes, cormorants, ibis, waterfowl, vultures, coots, rails, stilts, sandpipers, gulls, doves, swallows, wrens, warblers, and blackbirds. This site is open to school groups and the public, and bird watching continues to be one of the most important ancillary benefits of the wetland.

The Wakodahatchee, Florida, wetlands are attractive to wildlife (Bays *et al*., 2000). The numbers of bird species observed within the wetlands totaled 119 and 142 in 1997 and 1998, respectively. Thirteen of these species are considered commercially exploited, threatened, or endangered by state and federal agencies. Bird densities estimated for 1997 averaged 53 per hectare. At least 13 species of birds were observed to have nested on the site, indicating that the



**FIGURE 19.5** Thousands of blackbirds move in and out of the Sweetwater constructed wetlands in Tucson, Arizona.

system is providing suitable avian nesting habitat for certain species.

Bird use and water quality treatment are not always totally compatible uses of constructed wetlands. There may be conflicts between maintenance activities and the lifestyles of threatened or endangered species. For example, the nesting habits of stilts and burrowing owls lead them to prefer the levee areas of drained or dewatered wetland cells. This has interfered with system restarts in the stormwater treatment constructed wetlands in Florida. Huge flocks (thousands) of yellow-headed blackbirds (*Xanthocephalus xanthocephalus*) have been implicated in recontamination of water by pathogens in Arizona (Orosz-Coghlan *et al*., 2006) (Figure 19.5). Waterfowl have been implicated in turbidity increases at the Columbia, Missouri, constructed wetlands (Knowlton *et al*., 2002). Waterfowl, particularly geese, can also be devastating on new transplants and can effectively wipe out an entire planting.

## Mammals

A wide variety of mammals reside in or visit treatment wetlands. Some of the more common for North America are listed in Table 19.5. Shrews (Family Soricidae) are found along the edges of wetlands and moist fields and feed on insects. But by far the largest and most important group of mammals associated with wetlands are rodents. Small rodents are also found in FWS systems, such as mice and voles, most of which are herbivorous species that graze on plants and seeds and are prey to fish, wading birds, and raptors. However, it is the larger rodents that have proven problematic in many treatment wetlands. These rodents are briefly discussed herein.

### Muskrats

Muskrats (*Ondatra zibethica*) cut large numbers of emergent herbaceous plants, primarily cattails (Latchum, 1996), and build feeding platforms and nests (mounds). This grazing can change treatment wetland areas from densely vegetated to a patchwork of open and emergent areas (Kadlec *et al*., 2007). Muskrats consume a portion of the annual net primary productivity, principally rhizomes, but their mounds represent a greater share of this production. Densities of 20 or more animals per hectare have been found, which can destroy the majority of the macrophyte standing crop in a given year. At such an exacerbated scale, muskrat herbivory may be termed as an "eatout," and it is evidenced by the removal of essentially all emergent plant parts (Figure 19.6). Destruction of the wetland vegetative infrastructure may create an attendant loss of some water quality functions, but may not harm others. No designed research studies have been conducted to quantify the effects of muskrat eatout on wetland water quality performance, but potential impacts may

**TABLE 19.5**
**Mammals in Treatment Wetlands in North America**

| | |
|---|---|
| **Rodents** | |
| Shrews | *Sorex* spp. |
| Mice | *Peromyscus* spp. |
| Voles | *Microtus* spp. |
| Muskrat | *Ondatra zibethica* |
| Beaver | *Castor canadensis* |
| Nutria | *Myocastor coypus* |
| **Herbivores** | |
| Rabbits | *Sylvilagus* spp. |
| Deer | *Odocoileus virginianus* |
| Elk | *Cervus canadensis* |
| **Carnivores** | |
| Raccoon | *Procyon lotor* |
| Opossum | *Didelphus marsupialis* |
| Skunks | *Mephitis* spp. |
| Mink | *Mustela vison* |
| Otter | *Lutra canadensis* |
| Bobcat | *Lynx rufus* |
| Coyote | *Canis latrans* |



(a) (b)

**FIGURE 19.6** Hillsdale, Michigan, wetland before (July 18, 1999) and after (June 29, 2001) eat-out by muskrats.

be speculatively identified. Two water quality measures are very liable to change because of denudation: dissolved oxygen (DO) and total suspended solids (TSS). Increased potential for atmospheric reaeration is present in the plant-free wetland, and plankton can generate oxygen within the water column because there is sufficient light. A 2-ha wetland in Commerce Township, Michigan, provided data to support these intuitive concepts. In an early, fully vegetated phase, this wetland had an annual average outflow of 6 mg/L TSS and 9.8 mg/L DO. After a 100% muskrat eatout, TSS rose to 12 mg/L and DO rose to 10.4 mg/L. There were also possible muskrat impacts at the Olentangy, Ohio, wetlands. An eatout occurred in the year 2000, five years after system start-up (Mitsch *et al*., 2004). Thereafter, turbidity increased upon passage, in contrast to decreases that occurred prior to the eatout. A similar response for total phosphorus was observed. However, such results are not conclusive because of other factors in system operation.

The integrity of berms may be threatened by burrowing. Impacts on wetland hydraulics are also possible. In all cases, loss of emergent vegetation has been viewed with dismay by owners, regulators, and the general public. The damage caused by muskrats, primarily by burrowing in containment and separation berms, is not a matter of conjecture. There are several examples of compromised parallel cells in which divider berms have been breached by burrows: Estevan, Saskatchewan (Duncan *et al*., 1999); Corcoran, California (Gao *et al*., 2003); and Sacramento, California (Nolte and Associates, 1998b). At the Manitoba Interlake site 1, it was found that muskrat burrows were extensive and were threatening to breach the dikes at several locations. Roads have been damaged by burrow collapse at Saginaw, Michigan (unpublished), and at Sacramento.

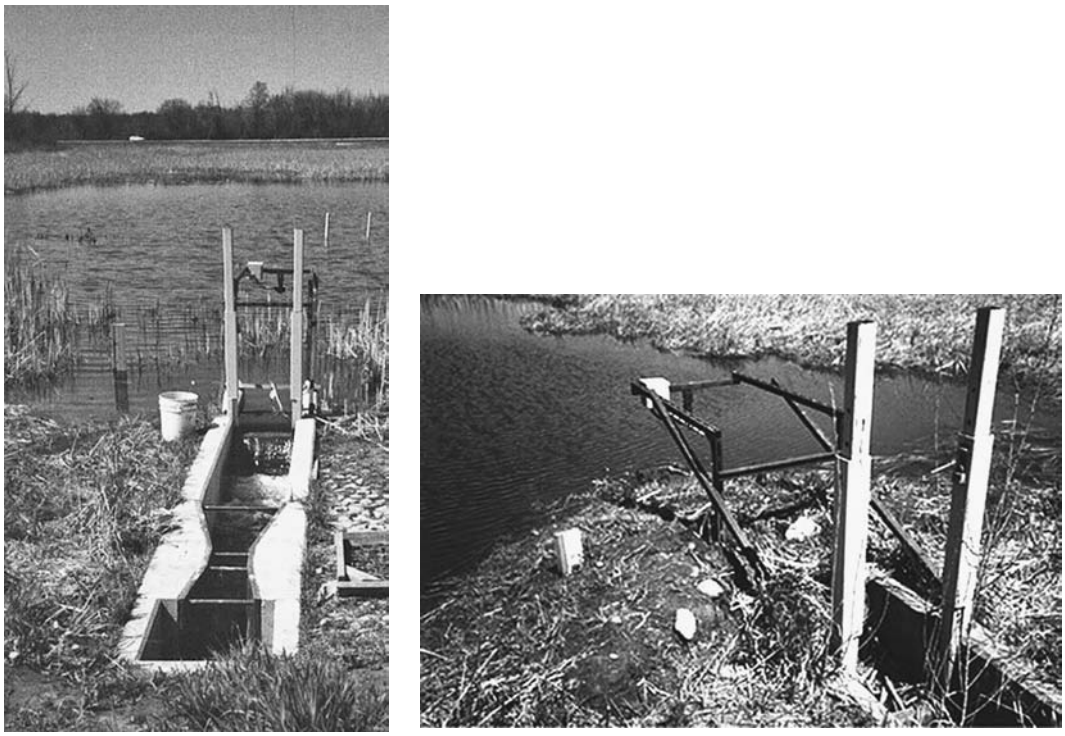
When muskrats eat all the emergent plant parts, there is a large visual impact, regardless of physical or water quality damage. A common perception is that wetlands should be lush and green to be effective in treatment. This is an intuitive carryover from agriculture, where lush green fields are a strong indicator of a good crop. Consequently, owners, regulators, and the general public equate good emergent stands of plants with the highest treatment capability. This may or may not be the case, but the urge to "fix it" is irrepressible in many cases. From the point of view of treatment efficiency, an eatout calls for careful scrutiny of performance to ascertain if the damage has, in fact, impaired pollutant removal. If not, then muskrat control becomes an expense associated with the ancillary benefit of a green emergent appearance. At Brighton, Ontario, where water quality was relatively unaffected by extensive muskrat herbivory, the aesthetic aspect became a concern. Residents of the community viewed the partially vegetated wetland as ineffective and expressed concern that there might be environmental consequences.

### Nutria

Nutria (*Myocastor coypus*) are an introduced species from South America, but they now range across the southern and northwestern United States. Nutria cause all the same problems as do muskrats, except that the animals are larger, and each individual is capable of more damage to vegetation. Mature nutria are large and have fewer natural predators. Nutria are strictly herbivorous and feed on a broad range of plants in treatment wetlands, including cattails, grasses, water hyacinth, duckweed, and young tree seedlings. Nutria cut vegetation in a manner similar to muskrats and build feeding platforms and nest mounds. They are prolific and commonly reach damaging population densities unless they are controlled by trapping or shooting. Nutria have occurred in problematic numbers in many treatment wetlands, including Halsey, Oregon; Victoria, Texas; and West Jackson County, Mississippi.

### Beavers

Beavers (*Castor canadensis*) are found in wetland treatment systems from Texas to the Canadian provinces. Beavers feed



(a) (b)

**FIGURE 19.7** Beaver activity caused complete closure of this outlet Parshall flume structure at Des Plaines River, Illinois, wetland EW3. A dam of soil, rocks, and sticks was constructed. Note the staff gage in the left panel is incorporated into the dam in the right panel.

on bark, twigs, and leaves from a variety of trees and, consequently, may cause undesirable herbivory damage in and around wetland treatment systems. For instance, the Roblin, Manitoba, constructed wetland was integrated into the adjoining terrestrial landscape with poplar tree (*Populus* spp.) plantings (PFRA, 2002). With an abundance of their favored aquatic habitat and favorite food (poplar), the beavers built lodges and decimated the tree plantings.

However, beaver damage to constructed wetlands does not stop with herbivory. When beavers find flowing water, such as in a treatment wetland structure, they construct a dam to contain that water, and create a deep-water habitat that will protect their nesting lodge from predators. The animals are large, up to 40 kg body weight, and can move stones of several kilograms, small logs, sticks, and mud to create stoppages in flows (Figure 19.7). Plugging of structures is clearly inimical to the proper performance of the wetland. Problems with beavers plugging structures have been experienced at many locations, including Fort Worth, Texas; Tres Rios, Arizona; and Des Plaines River, Illinois. Removal of the obstructions is not effective, because the animals can rebuild a dam literally overnight. Removal of the animals is the only recourse.

Beavers construct travel trails to facilitate movement with minimum risk from predators. These may be water-filled ditches leading to and from deeper water areas in upland margins or deepened areas in marginal shallows. In treatment wetlands, these travel trails may connect deep zones, thus providing increased opportunity for hydraulic short-circuiting.

Beavers burrow into banks as an alternative to lodges made of branches and small logs, or as an adjunct to such lodges. These burrows may create significant leakage from the wetland or compromise the integrity of containment berms. For instance, beaver burrows caused failure of the containment berm of a constructed infiltration bed/wetland at Houghton Lake, Michigan, causing over 1,000 $m^3$ of soil to wash out in one event.

These several effects of the presence of beavers are of considerable concern and often warrant removal of the animals.

## Predators

The richness of the invertebrate, avian, and small mammal populations in treatment wetlands provides good hunting for a number of predators. Mink (*Mustela vison*) prefer muskrats as food and will harvest a sizeable fraction of the resident population in the constructed wetland. Coyotes (*Canis latrans*) will prey upon waterfowl nestlings, provided the water is not too deep. As a consequence, wetlands that are subject to seasonal drawdown and dryout during the nesting season are vulnerable to coyote predation. This situation exists for the Incline Village, Nevada, wetlands, because water is diverted to crop irrigation starting in the nesting season.

Raccoons (*Procyon lotor*) and bobcats (*Lynx rufus*) are frequent visitors to wetland systems, from Arizona to Michigan in the United States. Alligators (Figure 19.4) are extremely numerous in the South Florida stormwater treatment areas.

### Incidental Visitors

There are numerous animals that may, from time to time, visit treatment wetlands and, in many instances, create problems. The Florida stormwater wetlands are very large and have very large inlet structures through which an occasional manatee (*Trichechus manatus*) may pass into the system. These large, gentle aquatic mammals are not a threat to the treatment system, but the habitat is wrong, and efforts are made to move them back to their natural habitat. The Tarrant County wetlands in central Texas were periodically visited by wild pigs (*Sus scrofa*), an abundant introduced species (more than one million animals estimated for Texas), with pernicious effects on the native ecosystems (Pimentel *et al*., 1999). The tubers of wetland plants were rooted up and eaten in the pilot wetland cells. SSF wetlands are also susceptible to these grazing problems; damage to plantings due to whitetail deer (*Odocoileus virginianus*) have occurred near Lutsen, Minnesota, and by wild pigs (*Sus scrofa*) near Nassogne, Belgium. On a worldwide basis, the list of potential troublemakers grows to include animals such as monkeys, which ruined a pilot treatment wetland vegetated with *Cyperus immensus* in Kenya (Abira *et al*., 2005).

## 19.3 DESIGN AND WILDLIFE USE

Each new treatment wetland situation brings a need to balance the wildlife value added and the potential impairment of the structure and function of the system for water quality improvement. To some extent, wetland design can influence this balance, but management can still be an important feature of wildlife control and enhancement. Management strategies are discussed under operations and maintenance in Chapter 22.

### Design to Encourage Wildlife

The general concept of maximizing wildlife values in treatment wetlands was elucidated by Sather (1989):

> In as much as plant community diversity is the key to faunal diversity, primary attention should be devoted to basic factors responsible for plant community diversity. The magnitude of ancillary benefits realized from wetlands constructed for wastewater treatment will be primarily dependent upon the degree of complexity and the amount of interspersion of the hydrophytic plant communities. Engineering design, therefore, should be sensitive to the hydrologic regimes and soil characteristics required to produce these types of plant communities.

Worrall *et al*. (1997) concluded that if constructed wetlands were to support wildlife as an integral function, they should be structurally diverse, with variations in plant types and cover, water depth, and substrate. Further, the wetland should be of sufficient scale and extent to support target species and populations: bigger is better.

The U.S. Environmental Protection Agency (U.S. EPA, 2000c) recognized that multiple values could be obtained from treatment wetlands. The degree of wildlife habitat provided by constructed treatment wetlands varies broadly across a spectrum. At one end of the spectrum are those systems that are intended only to provide treatment for an effluent or other water source and little to no wildlife habitat. At the other end are those systems that are intended to provide water reuse, wildlife habitat, and public use, besides providing a final polishing function for a pretreated effluent or other water source. The guidance provided by the U.S. EPA (2000c) primarily addresses the latter end of this spectrum. To promote wildlife values, constructed treatment wetland designs should avoid rectangular basins and straight channels and rather use diverse and sinuous edges and native vegetation whenever possible. It is desirable to incorporate the constructed wetland into the existing natural landscape with the natural transition zones as margins, perhaps including woody vegetated buffer areas around the site. Where appropriate, the constructed treatment wetland should provide habitat with a diversity of native species comparable to similar wetlands in the region. To the extent practicable, vegetative species diversity should be maximized without increasing the proportion of weedy, nonindigenous, or invasive species at the expense of native species. The biological diversity of the project is likely linked to, or dependent on, physical heterogeneity. This could include providing some areas of open water, creating nesting islands for waterfowl, and leaving some upland and buffer areas for other nesting species.

Waterfowl populations are enhanced if open water areas are interspersed with deep emergent marsh and upland islands. An approximate 1:1 ratio of wetland area devoted to marsh and to open water will provide maximum habitat for a variety of waterfowl species (Weller, 1978). Generally, wetlands adjacent to upland habitats will have greater floral and faunal diversity in proportion to the amount of the edge between adjacent ecosystems. Extending the perimeter does, however, carry added cost. Wading birds prefer a different habitat mix. These species require shallow, sparsely vegetated, littoral areas or perching substrates adjacent to open water areas. To benefit wading bird populations, constructed wetland treatment systems can be designed to provide a broad shelf of emergent marsh with water depth <20–30 cm. Deep, open water areas next to a shallow marsh provide additional foraging habitat. Open water areas and the transitional ecotones among marsh, open water, and adjacent uplands also help promote growth of herptile and fish populations which, in turn, provide food for wading and diving birds.

Depending on regional occurrence and available habitats, other bird varieties will colonize constructed wetlands, too. For example, if living or dead trees are included in the wetland, they will serve as perching and possible nesting sites (Hair *et al*., 1978). Nesting boxes might attract wood ducks and owls. Upland islands surrounded by open water



**FIGURE 19.8** Human and animal exclusion may necessitate fencing. At this Hillsdale, Michigan, remediation wetland site, there is a tall fence to exclude humans and a short fence to exclude muskrats. Whitetail deer (*Odocoileus virginianus*) have been observed jumping over the two-meter chain-link and barbed wire.

provide protection for ground-nesting bird species such as waterfowl.

### Design to Discourage Incompatible Wildlife

Project plans may include mechanisms to control or eliminate undesirable species. Design techniques vary, depending on the species of animal perceived as a possible threat. For instance, beavers may be prevented from plugging structures if the intakes and discharges are diffuse and noiseless and under more than a meter of water. Inlets and outlets may be screened to block entry by larger animals, such as turtles. Chain-link or coarse mesh fencing can block some intruders, such as domestic livestock.

A lack of predator habitat protects muskrats from predators such as hawks and owls. Erecting dead tree stubs or utility poles in an upright position that are of sufficient height to attract raptors can serve as a perch for these aerial predators. Such structures have been very effective in the Show Low, Arizona, wetland environments. Mink, the primary muskrat predator, is ubiquitous across the range of muskrats. These small animals can and will enter treatment wetlands in search of prey and are not deterred by chain-link fencing. Attempts to allow access to larger predators, such as foxes and coyotes, proved ineffective at a remediation wetland in Hillsdale, Michigan.

Exclusion fencing is an effective, but often expensive, option for controlling access by some animals. For example, muskrats have been effectively excluded from the Ducks Unlimited treatment wetlands at Oak Hammock, Manitoba. The exclusion fencing was erected prior to vegetating the site. Ducks Unlimited, Canada, uses 5 cm × 5 cm galvanized stucco wire mesh (hardware cloth), with about 0.30 m of the mesh vertically buried, about 0.5 m upright, and another 0.3 m set to a 45° angle sloped away from the wetland. It is important to ensure that the fencing is placed such that the muskrats cannot burrow underneath or between connected sections. Based on reports from the Ducks Unlimited domestic wastewater treatment wetland at Oak Hammock, Manitoba, similar fencing was installed at the Hillsdale, Michigan, treatment wetland, where it was also successful (Figure 19.8) (Kadlec *et al*., 2007).

Riprap placement along berms and bank slopes can be very effective at preventing burrowing. This involves the placement of up to 50–150 mm riprap in a 200–300 cm layer that extends from the wetland bottom to the top of the berm. Note that although this approach reduces the potential for burrowing into the berm directly at water level, experience at Nanticoke, Ontario, showed that the muskrats burrowed down into the roadway at the topmost edge of the riprap, which was 3 m from the water edge along the three to one side slope. An alternative is chain-link fencing laid into the final deep zone of the wetland and extended to the top of the berm, a technique that has been used successfully at the Lambton, Ontario, wetlands. There is probably a trade-off between mound building and bank burrowing, because both mounds and burrows are acceptable refugia. Thus, by denying bank access, mound building will be encouraged, and vice versa.

### Ecological Risk Analysis

The expanding use of treatment wetlands for remediation of contaminated groundwaters and for various industrial effluents poses a potential threat to resident or migratory animals. Bluntly put, the treatment wetland should not be a health threat to wetland creatures. For many specific chemicals there are water quality standards set to protect the aquatic environment, and these may serve as guidelines for FWS treatment wetlands. There are also standards for sediments in freshwater systems in some cases. In the event that it is determined that a threat may exist, other treatment alternatives such as SSF wetlands should be considered.

The tools for evaluating potential hazards are collectively known as *ecological risk assessment*. Ecological risk assessment is a process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors. The process is used to systematically evaluate and organize data, information, assumptions, and uncertainties to help understand and predict the relationships between stressors and ecological effects in a way that is useful for decision making. This process consists of three elements, as indicated in U.S. EPA (1998):

1. Problem formulation, the first phase, has the purpose of articulating the assessment. The problem is defined, and a plan for analyzing and characterizing risk is determined. Initial work in problem formulation includes the integration of available information on sources, stressors, effects, and the ecosystem and receptor characteristics. From this information, two products are generated: assessment endpoints and conceptual models.
2. Analysis, the second phase, is directed by the products of problem formulation. During the analysis phase, data are evaluated to determine how exposure to stressors is likely to occur (characterization of exposure) and, given this exposure, the potential and type of ecological effects that can be expected (characterization of ecological effects). The first step in analysis is to determine the strengths and limitations of data on exposure, effects, and the ecosystem and receptor characteristics. Data are then analyzed to characterize the nature of potential or actual exposure and the ecological responses under the circumstances defined in the conceptual model(s). The products from these analyses are two profiles, one for exposure, and one for stressor response.
3. During risk characterization, the third phase, the exposure and stressor-response profiles are integrated through a risk estimation process. Risk characterization includes a summary of assumptions, scientific uncertainties, and the strengths and limitations of the analyses. The final product is a risk description in which the results of the integration are presented, including an interpretation of potential ecological adversity, descriptions of uncertainty, and lines of evidence.

As an example, suppose that a treatment wetland is proposed for removal of trichloroethylene (TCE) from the water. One of the endpoints of ecological risk assessment would be the effects on muskrats that inhabit the system. The model could be the transfer of TCE from water to soils, from soils to cattails, and from cattails to muskrats that eat them. Characterization of the exposure would be the computation of how much TCE might be ingested. Characterization of the effect would be a determination of the acute and chronic doses that the muskrat could stand. If the estimated ingestion was well below the harmful dose levels, say, by several orders of magnitude, then the risk would be deemed low and ranked accordingly. This brief example was, in fact, part of the ecological risk assessment for the Hillsdale, Michigan, project (Ecology and Environment, 1996).

Obviously, the use of partially treated water engenders some degree of risk for humans and wildlife within the wetland. The most conservative course of action might be to monitor all potential indicators of contamination that might contribute to a problem of the health of some sector of the ecology—invertebrates up to the carnivores. Wren *et al.* (1997) outline an example of a three-tier, ultraconservative monitoring program. Unfortunately, the cost of such a program would usually be the fatal flaw for the entire project.

## 19.4 HUMAN USE

Humans appreciate wetlands for their commercial values (plant harvesting, livestock grazing, hunting, and aquaculture) and nonconsumptive values (aesthetics, recreation, and research) (Nash, 1978; Reimold and Hardisky, 1978; Sather and Smith, 1984; Smardon, 1988).

### Consumptive Activities

Wetland plants have been used historically for a number of purposes. Various sedges are used for fodder (marsh hay), and cedar (*Thuja* spp.) timber is highly valued. Cranberries (*Vaccinium macrocarpon*) are a high-profit cash crop grown in bogs, and blueberries (*Vaccinium* spp.) are grown commercially in peatland soils.

In North America, at least 46 native tribes were reported to use *Phragmites* historically (Kiviat and Hamilton, 2001). There were approximately 75 different uses of *Phragmites,* of which the most frequent were arrowshaft (17 tribes), cigarette (13), flute (12), whistle (7), pipestem (7), and matting (6). The number of uses per tribe was highest in the southwestern quadrant of North America. In modern times, in Europe, *Phragmites* is used for roof thatching, and it provides a very high-quality structure.

Plant harvest can possibly provide an economic profit from constructed wetlands. Solano *et al.* (2004) assayed two macrophytes, cattail (*Typha*) and reed (*Phragmites*). High levels of biochemical oxygen demand (BOD), chemical oxygen demand (COD), and TSS removal for all treatments were obtained, but cattails showed the greatest production of biomass (22 tons of dry matter per hectare). Both cattails and reeds presented high heating values (17–20 MJ/kg). Ciri *et al.* (2005) speculated that *Typha*, harvested from treatment wetlands, could be an economic fuel. A new rice variety (*Oryza sativa* "Kusahonami") developed for livestock feed was utilized for treating nutrient-polluted river water (Zhou and Hosomi, 2007).

Consumptive uses of wetland animals via hunting and trapping are more easily quantified than nonconsumptive functions. For example, over 10 million ducks and $35 million worth of furs are harvested from wetlands annually (Chabreck, 1978). Constructed treatment wetlands at Orlando (Florida), the stormwater treatment wetlands of Florida, and the Incline Village (Nevada)



**FIGURE 19.9** The Tollgate urban stormwater wetland in Lansing, Michigan.

system are used for waterfowl hunting. An ancillary benefit of the stormwater treatment areas (STAs) is that vegetation and shallow waters have attracted many ducks, such as teal (*Anas discors*), pintails (*Anas acuta*), widgeon (*Anas Americana*), ringnecks (*Aythya collaris*), and canvasbacks (*Aythya valsineria*). The South Florida Water Management District (SFWMD) and the Florida Fish and Wildlife Conservation Commission have opened some of the STAs to limited duck hunting by permit, starting in 2001. The hunting has generally been excellent, with the STAs producing an average of four to five ducks per hunter, which is unheard of for a public waterfowl area. As a result, hunters have come from all over Florida, as well as other states, to enjoy the STAs. United Waterfowlers Florida President Newton Cook stated that "STA 5 is the #1 public waterfowling marsh in the United States and is nationally known."

STA-5 was opened for the first year for alligator hunting in 2006 (SFWMD, 2006).

### Passive Activities

Nonconsumptive uses of wetlands constructed primarily for water quality treatment include recreation, nature study, aesthetics, and education. Increasingly, designs of treatment wetlands have incorporated attractive and informative parklike areas. For instance, stormwater treatment wetlands in urban settings such as Greenwood Park in Orlando (Florida), the Tollgate Wetlands in Lansing (Michigan), and Coyote Hills east of San Francisco Bay are used frequently for field trips and other educational purposes (Figure 19.9; Table 19.6). Wetlands constructed for wastewater treatment in Arcata, California; Hillsboro, Oregon; and Orlando and Palm Beach counties in Florida, are vital recreational areas, which offer jogging and bird watching. In South Florida, STA 5 and STA1W are open for guided bird-watching tours. The visitor will typically not be disappointed because these treatment wetlands are frequented by very large numbers of subtropical birds, including roseate spoonbills (*Ajaia ajaja*).

These human uses of wetlands, including the satisfaction of having a wetland and wildlife at the edge of town, are perhaps the most important factors behind public support of protection and enhancement of existing wetlands.

### Conflicts

Although most reasonable human use is acceptable in treatment wetlands, there are some potential conflicts. These are generally in two categories: destructive human activities and liabilities of the project owner—both real and perceived.

Humans can be destructive, either maliciously or unintentionally. An illustration of the unintended damage issue

**TABLE 19.6**
**Use Estimates for the Arcata, California, Treatment Wetlands**

| Use | Density (persons/day) | Duration (hours) | Management |
|---|---|---|---|
| Picnicking, relaxing | 57–175 | 0.3–0.8 | Traffic control, parking, garbage cans, and cleanup |
| Birdwatching, nature study | 5–50 | 2–4 | Traffic control, parking, viewing blinds, and docent program |
| Walking, jogging | 20–80 | 2–4 | Designated trails, maintenance, mosquito control program |
| Education | 2–6 | 2–4 | Designated trails, maintenance, signage |
| Photography | 1–2 | 2–4 | None |

is the case of a small treatment wetland in Dixboro, Michigan, which treats runoff from a botanical garden. Elementary school children visit the facility frequently, and one of their activities was catching frogs and dragonflies in the wetland. But about 30 children running around in the 850-$m^2$ wetland caused severe damage to sediments and vegetation, and the activity had to be cancelled.

More problematic is malicious damage. Signage has been defaced with spray paint (Show Low, Arizona), distribution pipes cut with an axe (Houghton Lake, Michigan), and stolen autos driven into the wetland (Howell, Michigan).

Issues of public safety combine with issues of facility protection to lead to limitation of access to treatment wetlands. Insurance against public injury has led to exclusion fencing at many treatment wetlands. This is more important for wetlands treating waters that may be hazardous to human health, which include some remediation sites as well as wetlands treating primary effluents. Nevertheless, there are no known instances of human injury in a treatment wetland.

## 19.5 DESIGN FOR ANCILLARY BENEFITS

The previous sections described the secondary features of wetland treatment systems, which are useful for encouraging wildlife. This section focuses on design considerations to enhance ancillary benefits in general, while optimizing the primary functions of pollutant removal and flood attenuation. Only a scattering of papers on wetland design for ancillary benefits have appeared through the past 15 years—for instance, Knight (1992; 1997); Worrall *et al*. (1997); and Conner and Luczak, (2002). As noted in an ancillary benefit design review by Conner and Luczak (2002), the U.S. EPA Design Manual (U.S. EPA, 2000a) "pays little attention to the conservation aspects of these treatment systems." The same may be said for Crites and Tchobanoglous (1998), Water Environment Foundation (2001), and Crites *et al*. (2006). The book by Campbell and Ogden (1999) considers sustainable development and how wildlife considerations may be woven into design; however, Conner and Luczak (2002) note that "the lack of detail limits its usefulness for design purposes." Outside of North America, there is scant attention paid as well, with the exception of the book by Merritt (Ed.) (1994), which provides useful information on the conservation aspects of systems in the United Kingdom. However, virtually all treatment wetlands in the United Kingdom are subsurface flow with *Phragmites*, and contain no concessions to wildlife or humans. In summary, there is no comprehensive manual on how to blend treatment and ancillary benefit designs. Ony a few manuals attempt to blend the concepts of mitigation wetlands and treatment wetlands, such as Sutula and Stein (2003).

Wetland design considerations include siting, cell size and configuration, water flow and depth control, planting, and species stocking. Because this book assumes that the treatment wetland's primary function is pollution control for wastewater, the water source itself is not an ancillary design decision. However, the level of pretreatment is important to determine the potential for secondary benefits. Developing a wide variety of habitat type within a treatment wetland will enhance higher wildlife diversity. Controls in the design of the treatment wetland, which affect habitat and plant diversity, include seasonal hydroperiods, depth and flow changes, vegetative succession, and accumulation of sediments (U.S. EPA, 2000c). Habitat quality can be enhanced by providing for areas of open water, creating nesting islands for waterfowl, and leaving some upland and buffer areas for other nesting species. Because the selection of plant palette is one of the major controls on habitat quality in a treatment wetland, it is important that the following recommendations be considered in the design:

- Provide habitat with native species mixtures comparable to similar wetlands in the region (U.S. EPA, 2000c).
- Create gentle slopes to allow for plant establishment and diversity. Incorporate water level fluctuations in the operation plan.
- Maximize vegetative species diversity without increasing use of weedy, invasive species at the expense of native species. Project plans may include the means to control or eliminate undesirable species.

Figure 19.1 shows some of the features that might be incorporated in a constructed wetland to control water pollution and to provide secondary wildlife and recreation benefits. In this conceptual plan, the wetlands are sized and configured to meet specific water quality treatment goals, and operation is based on meeting effluent criteria. However, the wetland also may be designed to take advantage of the potential for wildlife habitat creation by incorporating deeper, open water areas in the marsh and providing habitat islands as a refuge from human access and predators. Constructed wetland treatment systems providing a multitude of ancillary benefits now exist at a number of project sites.

### Siting

Siting wetlands to treat municipal and industrial wastewater is relatively flexible because the effluents are often pumped to the wetland. However, wetlands to control stormwater must be sited close to stormwater sources or further downstream in a watershed, intercepting a tributary. The location of a stormwater treatment wetland will determine the quantity and timing of the influent, which will affect the ancillary functions of the wetland. Generally, wetlands located in headwater areas will receive more irregular and less dependable inflows, resulting in prolonged dry conditions unless soils are impermeable or groundwater is normally high. This relative lack of flooding will limit ancillary benefits, such as primary and secondary production. In these cases, maintaining a healthy stand of wetland-dependent vegetation may be difficult, and upland or transitional species may eventually predominate. This type of system certainly will have some upland values

and may support faunal assemblages seasonally. However, overall production of wetland-dependent species likely will be lower than in a perennially flooded wetland.

Siting the constructed stormwater treatment wetland further downstream in the watershed may yield a different constraint: too much water during stormwater runoff periods. To control high stormwater volume, the downstream wetland can be designed to be off-line (out of the main flow path) so that it captures only some flood flow, preventing the washout of vegetation and berms. A series of off-line constructed wetlands, each capturing a portion of the storm flows, can be used to deal with high volumes. Wetlands located downstream in a watershed are more likely than wetlands at headwaters to have perennial water because of more constant base flows and higher groundwater levels. In turn, downstream stormwater treatment wetlands generally will have more wildlife and more food-chain support.

The siting of wetland treatment systems to optimize ancillary benefits may also consider the advantages of locations with adjacent donor wetlands (sources of plants and wildlife to colonize the area), adjacent undeveloped uplands (habitat diversity), or proximity to humans (aesthetics).

### Cell Size and Configuration

The wetland cell size depends primarily on water quality treatment needs and cost considerations. Because large cells require less berm construction per unit area and fewer inlet and outlet structures, per-area project costs are lower. For example, a 100-ha constructed wetland may cost about $10,000/ha to construct, whereas a smaller constructed wetland may cost about $50,000/ha. Although cell size may influence the use of wetlands by larger wildlife, it has minimal affect on plant productivity or secondary production of most wetland animals (Sather and Smith, 1984). A higher berm-to-cell area ratio, typical of smaller wetland cells, may result in increased beneficial edge effects. As long as berms are infrequently mowed or visited, the larger edge area provides nesting and feeding habitat for more mammal and bird species.

Islands surrounded by marsh or open water provide excellent habitat for nesting waterfowl. In many wetlands, islands with trees are preferred nesting habitat for wading bird rookeries (see Figure 19.1). Nesting islands for waterfowl should be about 0.6 m above the normal high water, whereas higher and lower islands may also be valuable for other species for feeding, resting, or nesting.

Open water areas may improve the water quality treatment potential of constructed wetlands, but they definitely enhance their ancillary benefits for wildlife. Mallard duck production is maximum in wetlands with almost even areas of marsh and open water (Ball and Nudds, 1989). Open water areas can be created by excavating at least 1.5 m below the normal water level. Deeper excavations can also provide fisheries habitat. To prevent hydraulic short-circuiting, open water areas should not be connected along the flow path but rather interspersed with densely vegetated shallow marsh habitat (average water depth of about 0.3 m or less). In general terms, areas will be larger when ancillary benefits are desired. Ponds, islands, and edge amenities are not required for treatment.

Cell number and configuration in series or in parallel are major considerations when determining treatment capability and operational flexibility. These design considerations primarily affect ancillary wetland benefits because of their importance in depth control.

### Water Depth and Flow

Water depth and the flow rate are important factors affecting dissolved oxygen in wetlands. Higher flow rates resulting from shallow water tend to provide higher dissolved oxygen concentrations in marshes because of atmospheric reaeration. These higher dissolved oxygen levels generally result in higher secondary production of aquatic invertebrates and vertebrates, thereby increasing ancillary wetland benefits. Although deeper water in a marsh may increase hydraulic residence time, this longer reaction time does not always result in enhanced water quality treatment (oxidation of organic matter and ammonia) because of the resulting reduction of dissolved oxygen concentrations.

Water depth is one of the main factors that affect wetland plant growth. High water levels will stress the growth of emergent macrophytes and encourage the dominance by floating or submerged plants or algae. The hydrological tolerance range and optimum hydroperiod of any desired vegetation type should be known and closely adhered to when water level control structures are designed. Ideal design allows water levels to be varied from zero (drained) to the maximum depth tolerance of desired wetland plant communities. Stop logs or weir plates should seal the leaks to help maintain water levels during periods of limited inflows. Multiple inlet and outlet weirs between adjacent cells allow greatest hydroperiod control flexibility.

### Vegetation

The plant species selected for a constructed wetland will greatly influence ancillary benefits such as primary and secondary productivity. The selection of improper plant species will result in low productivity, and a lengthy adaptive period may be necessary until available plant species, either planted or natural volunteers, rearrange themselves according to hydrologic and water quality factors. High plant diversity can be initially achieved by assisting existing soil seed banks to colonize areas that are graded and shallowly flooded or by spreading muck and propagules from a donor wetland area (Gilbert *et al.*, 1981). Wetland vegetation establishment is most rapid when plants are closely spaced, less than 1 m on centers, and planted during the growing season (Lewis and Bunce, 1980; Broome, 1990). Wildlife use all marsh plant species directly for food or shelter or, indirectly, through the detritus food chain, so expensive management to exclude "noxious species" or to select favored species may lower

overall wildlife use in favor of optimizing specific wildlife species. Burning marshes may be good management for waterfowl species but poor management for fish, small mammals, or other bird species.

### Pretreatment

In addition to wetland hydroperiod, water quality is a key determinant of a wetland's form and function. Primary water quality characteristics, which affect wetland plant communities, include nutrients (especially nitrogen and phosphorus), suspended solids, salts, pH, and temperature. Except for nutrient concentrations, these same water quality characteristics also influence faunal populations. Inflows of biodegradable solids and the ammonia form of nitrogen can affect wetland flora and fauna indirectly by their impact on dissolved oxygen concentrations.

If high loads of suspended solids are released in the constructed wetland, the solids can smother plant growth in inflow areas (Kuenzler, 1990). Generally, solid loads can be controlled by conventional pretreatment. In stormwater systems, the problem can be minimized with a forebay or pond prior to the wetland (Livingston, 1989). If the pretreatment area traps the mineral-suspended solids (clays, silt, and sand), the wetland treatment system will need to be maintained less frequently.

Inlet ponds also can reduce nutrient levels, although nutrient reduction may not be the primary goal in constructed wetlands designed to enhance wildlife. As noted earlier, higher nutrient levels generally result in higher primary production of wetland plants which, then, support increased wildlife populations.

As a consequence, the wetland should not be used as the first interceptor of urban runoff, particularly in industrial or highly urbanized sites (Schueler, 2000). Specific recommendations include the following (Sutula and Stein, 2003):

- Control source of pollutants where possible (e.g., street sweeping, removal of sediment and debris from storm drain inlets, improving infiltration, and other land-based best management practices (BMPs).
- Use oil and grit interceptors in industrial sites or highways.
- Install floating berms to trap trash and large debris.
- Utilize sediment collection/settling forebays. Design and locate the forebays for ease of maintenance and removal of sediment.
- Segment the wetland such that the primary treatment is provided in the forebay or initial pond and the remainder of the wetland is used for polishing and/or wildlife enhancement.

### Human Access

Recreational facilities that provide public access to the STAs of South Florida include asphalt parking areas, composting toilets, landscaping, and information kiosks. Pedestrian gates, signage, and fencing are included as needed to define public access areas. Associated road improvements are sometimes needed, such as an acceleration and deceleration lane in the vicinity of the entrance to the parking lot. Some facilities include a canoe launching site, access footbridges, and boardwalks.

Although public access to a created wetland might disturb wildlife populations, disturbances can be minimized with controlled access to certain areas and design features such as islands for roosting and nesting. Visitors typically appreciate guidance, and signage is therefore an important aspect of design for human use (Figures 19.10 and 19.11). Although clearly marked walking trails suffice for wetlands of limited size (Figure 19.10), boardwalks are the preferred means of protecting both the visitors and the wetlands in large systems (Figure 19.11). These are not restricted, however, to large systems (Figure 19.12). Boardwalks facilitate nature study and outdoor recreation, and enhance the scientific research benefits of a constructed wetland. Observation towers are valuable because they provide an opportunity for visitors to get an overview of what is generally a flat landscape (Figure 19.13). Wildlife viewing blinds can greatly enhance the ability to observe wildlife with minimal disturbance to the animals. An interpretive center allows small groups to assemble with shelter from sun and rain (Figure 19.14). In some very large treatment wetlands, boat launching facilities are provided (Figure 19.15).

It is not uncommon for scientists to visit treatment wetlands. This human use is typically not accompanied by any special facilities; such visitors are deemed knowledgeable enough to fend for themselves (Figure 19.16).

It is noteworthy that there are no reports of accidents involving humans and treatment wetlands.

### Examples

The popularity of treatment wetlands for public use is very high. Perhaps the best way to illustrate the design features that contribute to public use is to describe examples.

#### Palm Beach County, Florida

Originally conceived in 1992 as part of a cooperative effluent reuse study between the Palm Beach County Water Utilities Department, the City of West Palm Beach, and other municipalities, the Wakodahatchee Wetlands have been designed to demonstrate the following ancillary benefits of reclaimed water reuse (see Figure 19.11):

- Creation of significant wildlife habitat accessible to the public in a parklike setting
- Maintenance of biological diversity and open space in a highly developed landscape
- Recharge of local water supplies through infiltration to the surficial aquifer
- Provide limited public use for passive recreational activities




(a)



(b)

**FIGURE 19.10** The Whangarei, New Zealand, treatment wetlands are very visitor friendly.



(a)




(b)

**FIGURE 19.11** The Wakodahatchee Wetlands in Palm Beach County, Florida, has a very user-friendly boardwalk.



**FIGURE 19.12** Even this short boardwalk at the University of Michigan Matthaei Botanical Gardens treatment wetland is a popular place to stop and look.



**FIGURE 19.13** Wetland observation tower at the Hennepin–Hopper (Illinois) site.



**FIGURE 19.14** Public viewing shelter at the Tucson, Arizona, Sweetwater wetlands.



**FIGURE 19.15** Boardwalks and boat launching sites are features of the South Florida Stormwater Treatment Areas.



**FIGURE 19.16** Scientific visits are an important human use of treatment wetlands, such as this at the leachate treatment wetlands at Reguengo, Portugal.

The Wakodahatchee Wetlands are part of the Southern Region Water Reclamation Facility, completed in 1997. The wetland area totals 16 ha on the 23-ha site, with individual wetland cells ranging from 0.9 ha to 4.4 ha. The marsh area is designed to operate at an average depth of 15 cm but may be operated normally at depths to 45 cm. High rainfall can elevate wetland water levels to even greater depths of 60 cm or more for short periods. Deep zones, consisting of ponds 1.5 m or more in depth, variable in width, and oriented transverse to the direction of flow, are interspersed throughout each wetland. A total of 28 deep zones and 8 habitat islands are included. Nominal detention time varies from a maximum of 40 days at 2,000 $m^3$/d of flow to ten days at 8,000 $m^3$/d.

Marshes comprise about 70% of the wetland area, vegetated by native emergent, forested, and transitional wetland species and designed to emulate native South Florida wetland plant communities. Emergent marsh zones are composed of bulrush (*Scirpus validus, S. californicus*), duck-potato (*Sagittaria lancifolia*), arrowhead (*Sagittaria latifolia*), spikerush (*Eleocharis cellulosa*), fireflag (*Thalia geniculata*), and pickerelweed (*Pontederia cordata*). The wetlands are maintained to be free of nonnative and nuisance plant species, and the surrounding berms and entry areas are landscaped to maintain an aesthetic and comfortable experience for wetland visitors. The combined efforts to manage the vegetation, maintain the boardwalk facility, monitor water quality, and maintain the grounds cost on the order of $150,000/yr.

Snags and perching posts were added for wildlife use. The numbers of bird species observed within the wetlands totaled 119 and 142 in 1997 and 1998, respectively. Thirteen of these species were considered by state and federal agencies to be commercially exploited, threatened, or endangered. Reptile diversities were high and increased over the first two years. Mosquitofish densities were high and consistent between the two years. Mammal and amphibian diversities were low.

The public makes extensive use of the Wakodahatchee Wetlands. Parking facilities are available, and there is a boardwalk through the wetlands, with associated gazebos, which provide an attractive and comfortable setting for walking, bird-watching, nature photography, environmental education, and related passive activities (Figure 19.11). The Audubon Society of the Everglades has identified the Wakodahatchee as one of the birding hot spots within Palm Beach County. Available data on site use indicate that the numbers of daily visitors to the wetlands have increased from 165 people in 1997 to over 1,500 through the first half of 2000. This high level of interest has led the Water Utilities Department to construct another wetland facility with an interpretive center on 70 ha of farmland adjacent to the Wakodahatchee Wetlands for treatment and disposal of up to 38,000 $m^3$/d of treated effluent.

### Victoria, Texas

The DuPont Victoria constructed wetland was built to treat wastewaters from a petrochemical plant that manufactures precursors for nylon and polyethylene. The 21.5-ha wetland receives about 12,000 $m^3$/d of water that has been pretreated in a biological plant. The wetland is configured in three stages (Figure 19.17). Stage 1 consists of five cells in parallel, which discharge into two parallel Stage 2 cells, both stages containing deep zones. Water then passes to Stage 3 wetlands, which are primarily intended for wildlife habitat. There are deep open-water zones for waterfowl, and islands add protected upland features to the habitat. Twenty species of native plants were used. Public access features include a parking area, an outdoor education center suitable for classroom groups, an observation platform, information kiosk, bird watching blind, walking trails, a boardwalk across the wetland, and a water sampling pier.

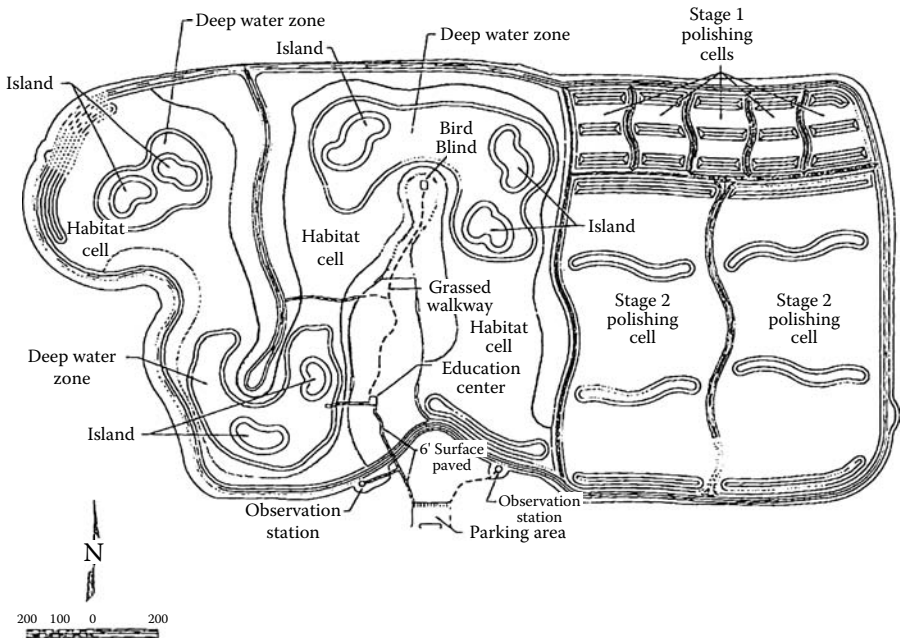



**FIGURE 19.17** Layout of the DuPont Victoria, Texas, treatment wetland. (From Reitberger *et al*. (2000) *Achieving Multiple Benefits from a Constructed Wetland*. Reddy and Kadlec (Eds.), Proceedings of the 7th International Conference on Wetland Systems for Water Pollution Control, 11–16 November 2000; University of Florida and IWA: Lake Buena Vista, Florida, pp. 749–758. Reprinted with permission.)

The wetland provides valuable habitat for migratory and resident birds. In excess of 180 species were identified in a 1999 survey, which increased to over 195 by 2002. An additional 28 species of plants had volunteered by 1998, and by 2002 there were 85 species of plants. Five species of fish live in the wetland and are just as healthy as their counterparts in the reference pond. Many larger animals have been attracted by the abundance of the wetland, but a problem was the invasion by nutria (*Myocastor coypus*), a nonnative aquatic rodent. The wetlands suffered significant loss of vegetation because of herbivory by the nutria. Population control methods are being employed, including attracting alligators to the site. Hunting and trapping methods were developed that provided effective population control without using firearms. Revegetation efforts have taken place despite continued nutria activity. Measures include replanting, nutria exclosures, and water level adjustment, which have successfully reestablished vegetation in most impacted areas.

Education programs are provided to students from Grade 4 through college via the Wetland Environmental Science Education Encounter (WE SEE). The program encompasses treatment benefits along with the inherent natural benefits of a wetland. A full-time on-site educator is the first such person located at an industrial wetland. The program is housed at a state-of-the-art education center and offers a wide variety of science education learning opportunities to area schools. Given that many young persons will be in a natural environment, there have to be rules. There is an inherent danger in any "natural" setting. To minimize the chance of injury, the students are cautioned to do the following:

- Do not leave trails or enter any area of high grass or brush.
- Do not attempt to capture any animal known to sting or bite. This includes such insects and arachnids as spiders, scorpions, bees, wasps, hornets, ants, and caterpillars.
- Do not attempt to capture or harass any mammal, bird, amphibian, or reptile in the wetland. Be on the lookout and avoid snakes.
- Watch for and do not attempt to touch plants with stems bearing leaves clustered in threes (poison ivy or poison oak). This plant has not been found in the wetland yet. If not sure, ask. Some plants in the wetland have thorns, so be careful of the brush.
- Walk, and do not run anywhere in the wetland, including on the trails, pier, boardwalk, and inside the education center.
- Follow proper field and laboratory safety procedures.

As of August 2006, over 27,000 students had participated.

### Renton, Washington

The Waterworks Garden is a stormwater treatment system with emphasis on not only the usual natural features of wetlands but also on elements conceived for their artistic value. An environmental artist (Lorna Jordan) worked with landscape architects and engineers to develop the combination of wetland treatment and art. Visitors pass through "rooms" that



**FIGURE 19.18** The water enters the Waterworks Gardens urban stormwater wetlands in Renton, Washington, under a paved colonnade.

symbolize a journey from the civilized to the wild (Campbell and Ogden, 1999). The walking path parallels the course of the water, moving from the "Knoll" inlet structure through more natural water forms ("Funnel," "Grotto," "Passage," and "Release") before discharge to the river (Figure 19.18). The planting scheme is symbolic; plants at the top of the garden where the runoff is still contaminated are species Native Americans consider bad omens (France, 2003). The entire system is just a few hectares, but the cost of the art was $1.3 million in 1995, the year of completion.

## SUMMARY

Wetlands for water quality improvement are being constructed throughout North America at an accelerating rate. However, many wetland designs do not incorporate ancillary benefits to the extent possible. With thoughtful design, constructed wetlands can provide benefits beyond effective water treatment, such as wildlife enhancement and recreational opportunities. In fact, with the decline of the total area of natural wetlands, constructed treatment wetlands are a viable and cost-efficient way to compensate for the loss of productive wetland habitat.

Pretreated wastewater and stormwater can nourish hydrologically altered natural wetlands or create natural wetland functions in constructed marshes. Wastewater constituents can be controlled to enhance a wetland's ancillary benefits. For example, nutrients promote plant growth at the bottom of the food chain, which then supports other wildlife species. By considering certain design features such as cell configuration, hydroperiod control, vegetation and wildlife stocking, and human access, wetlands can be designed to meet water treatment goals as well as provide additional benefits.

# 20 Sizing of SSF Wetlands

The design of subsurface flow (SSF) wetlands may be roughly divided into two categories: sizing calculations and physical specifications. Sizing requires characterization of the incoming water and regional climate as well as the goals of wetland treatment, as discussed in Chapter 16. This chapter discusses different sizing methodologies for SSF wetlands. Chapter 21 deals with the physical considerations, including the number of cells, layout, liners, bed depth, media size, plants, and water level control. It is recognized that SSF wetlands are not stand-alone treatment devices but rather form part of an overall treatment process.

For HSSF wetlands, primary treatment, at a minimum, is required to remove settleable and floating solids prior to the wetland bed. This is typically provided through the use of settling tanks. Some projects may employ a higher level of pretreatment through the use of activated sludge systems, lagoons, or other treatment processes. Vertical flow (VF) wetlands may or may not have such preliminary treatment stages, and biosolids systems will receive material of different quality depending on the nature of the overall treatment process. This chapter focuses on the sizing of SSF wetland components, recognizing that this will be only one part of the overall treatment process.

To date, most SSF wetlands have been sized using prescriptive criteria. The most common approach has been to tie the required wetland size to a parameter that presumably defines the influent conditions. Influent loadings, population equivalents, detention time, and number of bedrooms have all been used to create these scaling factors. Prescriptive criteria represent a reasonable approach to wetland sizing when there is a large body of preexisting performance data for well-defined pollutants (loading charts), and a more detailed understanding of the internal dynamics of the wetland is not necessary. As a result, prescriptive criteria are used mainly for the design of SSF wetlands that treat domestic wastewater.

However, for other projects, there is limited information, and prescriptive criteria cannot be developed. Pilot-scale treatability studies may have to be conducted to measure rate constants and temperature effects. First-order modeling is often employed to interpret these results and make predictions on the anticipated performance of a full-scale wetland. Also, first-order modeling can be used to represent internal profiles of pollutant reduction within the wetland. This book employs the *P-k-C** model for performance-based wetland design.

## 20.1 PRESCRIPTIVE SIZING CRITERIA

Prescriptive criteria have been the most popular method for sizing SSF wetlands because these methods are quick and mathematically simple. This has led to the widespread use (and sometimes misuse) of prescriptive methods.

Prescriptive criteria are only reasonable when they are based on the observed performance of large numbers of wetlands built to the same physical specifications treating the same type of wastewater under the same climatic conditions. If treatment is acceptable under a specified condition, and the data set is large enough to capture inter- and intra-system variability, it may reasonably be expected that future wetlands operated under the same conditions will also provide acceptable treatment (and that deviations from the mean are acceptably small or infrequent). Unfortunately, prescriptive criteria have been adopted in many situations where there is insufficient data to support their use.

### Loading Charts

Loading charts are design tools, which can be used to size wetland systems. An influent loading rate is selected to produce a targeted effluent concentration. The wetland area is calculated from the influent mass load. Recent examples include U.S. EPA (2000a) and Wallace and Knight (2006).

Loading charts that can be used in this manner are also included in this book. They are composed, mainly, of wetland data from Europe, North America, the United Kingdom, and New Zealand; so the vast majority of the systems represented in these charts is from cool temperate climates. For HSSF wetlands, these charts include

| | |
|---|---|
| TSS: | Figure 7.30 ($C_{out}$ versus $Load_{in}$) |
| BOD: | Figure 8.26 ($C_{out}$ versus $Load_{in}$) |
| Organic N: | Figure 9.18 ($C_{out}$ versus $Load_{in}$) |
| Ammonia N: | Figure 9.38 ($C_{out}$ versus $Load_{in}$) |
| TKN: | Figure 9.24 ($C_{out}$ versus $Load_{in}$) |
| Total N: | Figure 9.30 ($C_{out}$ versus $Load_{in}$) |
| Total P: | Figure 10.36 ($C_{out}$ versus $Load_{in}$) |
| Fecal coliforms: | Figure 12.12 ($C_{out}$ versus $C_{in}$) |

For VF wetlands, these charts include

| | |
|---|---|
| TSS: | Figure 7.32 ($C_{out}$ versus $Load_{in}$) |
| BOD: | Figure 8.31 ($C_{out}$ versus $Load_{in}$) |
| TKN: | Figure 9.25 ($C_{out}$ versus $Load_{in}$) |

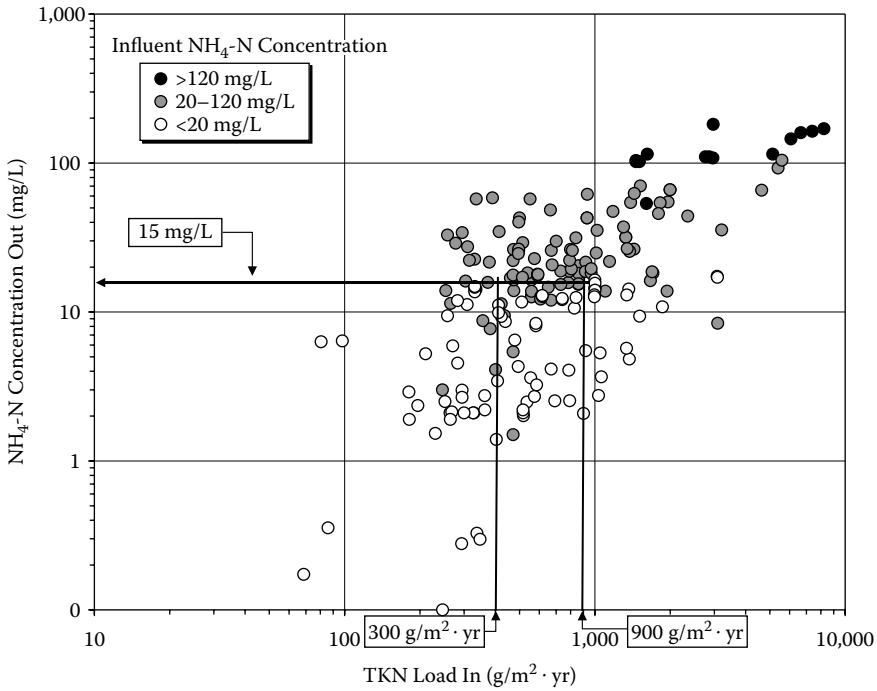



**FIGURE 20.1** TKN–Ammonia loading chart, with forecasted treatment performance.

Because there is considerable scatter in the loading of chart data, selection of the wetland area will likely be an iterative process; selection of a lower influent loading (larger wetland area) will result in lower estimated effluent concentrations. For instance, consider the case of ammonia in Figure 20.1 (replotted from Figure 9.39).

A TKN loading rate of 900 g/m$^2$·yr (approximately 5 m$^2$/PE; $C_i$ approximately 120 mg/L) results in a predicted effluent ammonia concentration of approximately 15 mg/L. Because of scatter in the data, this estimate of effluent concentration can only be considered a central tendency. In other words, approximately half of the wetlands loaded at this rate will return ammonia concentrations greater than 15 mg/L.

If the loading rate is decreased to 300 g/m$^2$·yr (approximately 15 m$^2$/PE) for the same influent concentration, the probability that the wetland will produce an effluent ammonia concentration equal to or less than 15 mg/L is increased considerably. This is consistent with the operational experiences with HSSF wetlands, where loadings less than 600 g/m$^2$·yr (10 m$^2$/PE) are necessary to return low-ammonia effluents (Geller, 1996).

It is tempting to estimate the probability that the wetland will not perform to this standard by counting the number of data points above the targeted performance goal. For instance, there are seven system·years above the targeted performance level (15 mg/L at 300 g/m$^2$·yr) out of 47 system years at or below this loading rate, so the probability that the wetland will meet the median effluent performance goal is roughly 85%. This predictor only describes median performance over the data averaging period used to construct the loading chart. Because these data averaging periods are often long (annual or period of record), important aspects of system performance, such as seasonal or stochastic variations in effluent concentrations, are lost. More importantly, this "point counting" approach does not take into account the effect of the influent concentration ranges. For instance, many data points in Figure 20.1 are for $C_i$ less than 20 mg/L, which is considerably lower than the 120 mg/L influent concentration used in this example.

Because the data averaging period for Figure 20.1 is annual (198 system·years of data are represented), no information on seasonal or monthly variation is contained in the chart. This is a problem when the regulatory compliance interval is short (weeks or months). Some design references have attempted to address this by using shorter data-averaging periods, such as system·months (Wallace and Knight, 2006).

However, tools such as those shown in Table 9.35, Chapter 9, can be used to assist the designer in this regard. As indicated in the table, the 90th percentile trend multiplier on the effluent concentrations ($1 + \Psi$) is 1.76. Figure 20.1 indicates that at an influent loading of 300 g/m$^2$·yr, the central tendency in the effluent ammonia concentrations is approximately 5 mg/L. If the wetland performs at or near this central tendency, 90% of the effluent results should be less than 8.8 mg/L ($1.76 \times 5$ mg/L). The probability that the wetland will have poor annual and stochastic results can be crudely estimated by multiplying the probability factors.

It should be noted that the amount of data available on the performance of treatment wetlands is much less than statisticians are used to working with; thus the application of statistical tools to wetland performance is inherently limited by the lack of current performance data.

Tools such as those shown in Figure 9.50, Chapter 9, indicate that some of this variation is likely to be seasonal.

There are obvious limitations to the loading chart method. First of all, there must be a functional relationship between the inlet loading and effluent concentration. Figure 7.29 in Chapter 7 is an example where the effluent TSS concentration is essentially independent of the influent loading. This is because removal of the influent TSS occurs in only a small region at the inlet of the HSSF wetland bed, and the overall wetland size (and hence, loading) does not play a significant role in determining effluent TSS concentrations. Secondly, there is considerable scatter in loading chart data, because wetland design, climatic, and loading parameters are not implemented in the same, exact fashion for all systems comprising the data set. The resulting data scatter means that a large component of the wetland design must necessarily rely on the best professional judgment of the designer. Some performance features, such as the role of influent concentration (independent of influent load), are also lost.

Because loading charts require few mathematical calculations, they are an attractive design tool. They have also been misused by designers who do not understand the inherent limitations of the loading chart method. It is important to remember that loading charts will not predict system performance if the physical configuration of the wetland is changed from the parent data set, the pollutant type or concentration changes, or the wetland is in a different climatic region. For instance, loading charts based on the BOD in domestic wastewater will not necessarily apply to industrial effluents where the organic compounds making up the BOD are not the same, and they will likely have different degradation rates (*k*-rates). Also, a loading chart based on a parameter such as BOD will not predict effluent performance for another pollutant, such as ammonia. Also, as loading charts are based on inlet–outlet data, predictions about pollutant reductions within the wetland (internal profiles) cannot be made.

## Scaling Factors

Many efforts have been made to reduce the information contained in a loading chart down to a single parameter—a prescriptive scaling factor. This is done to further simplify the design process. In general, these scaling factors attempt to describe two aspects of system performance:

1. Median system performance at a specified inlet condition.
2. A "margin of excursion containment" that is intended to allow for intersystem (seasonal and stochastic) as well as intrasystem (differences in the performance response of different wetland systems) variabilities.

This margin of excursion containment is rarely defined in the treatment wetland field; the only exception to date is Wallace and Knight (2006).

There is often an underlying assumption that *bigger is better* (larger wetland area for a given influent condition), and this has often been used as an excuse to not rigorously collect or analyze performance data. As a result, many prescriptive criteria are overly conservative and hinder, rather than help, the reliability of the wetland. There are numerous examples of "bad" or outmoded concepts that are imbedded in various "prescriptive rules." Examples include making the wetland excessively large (exacerbating freezing problems in cold environments and water loss to *ET* in arid climates), specifying long length:width ratios (maximizing the cross-sectional loading and associated clogging problems) for HSSF wetlands, and specifying deep beds to increase the hydraulic retention time (exacerbating vertical stratification of the flow).

### Scaling Factors for HSSF Wetlands

Scaling factors provide the simplest approach to wetland design; thus it is not surprising that this is the most commonly used design method for HSSF wetlands. Examples of some existing criteria include

- 5 $m^2$/PE (BOD loading equivalent to 8 $g/m^2 \cdot d$) to produce an effluent BOD of less than 30 mg/L for primary-treated domestic wastewater
  *Source:* (EC/EWPCA Emergent Hydrophyte Treatment Systems Expert Contact Group and Water Research Centre, 1990)
- 3.1 cm/d
  *Source:* (TVA, 1993)
- Bed sizing ≥ 5 $m^2$/PE (BOD loading equivalent to 8 $g/m^2 \cdot d$)
  *Source:* (ATV, 1998; ÖNORM B 2505, 2005)
- 6 $g/m^2 \cdot d$ of inlet BOD loading to produce an effluent with less than 30 mg/L BOD
  *Source:* (U.S. EPA, 2000a)
- 8 $g/m^2 \cdot d$ of inlet BOD loading to produce a median effluent concentration of 30 mg/L; 5 $g/m^2 \cdot d$ to produce a median effluent concentration less than 25 mg/L BOD
  *Source:* (Wallace and Knight, 2006)
- 10 to 13 days' nominal hydraulic retention time
  *Source:* (Gustafson *et al.*, 2001)
- 41 $L/m^2 \cdot d$; two to three days' nominal hydraulic retention time
  *Source:* (Lesikar, 1999)
- 28 $m^2$ per bedroom per day; trenches 30 cm wide and 92 m long per bedroom
  *Source:* (Iowa Department of Natural Resources, 2001)

### Scaling Factors for VF Wetlands

Scaling factors have also been widely employed to size intermittently loaded VF wetland systems, especially in Europe. Scaling factors that are currently in use are summarized in Table 15.1, in Chapter 15.

### Cautions Regarding the Use of Scaling Factors

These summaries are by no means a complete list of all the scaling factors existing in the treatment wetland field. Obviously, these examples bear little functional relationship to each other; all are clearly artifacts of the available data, regulatory situation, and climatic conditions where they were developed. The designer is thus cautioned that extrapolating these types of prescriptive rules beyond their local regulatory jurisdiction will likely result in inappropriate wetland sizing; also, many of the physical specifications contained in these prescriptive rules are obsolete and have not withstood the test of modern data analysis in the treatment wetland field.

### Empirical Equations

Empirical equations are another means of predicting system performance. These equations are developed based on a preexisting data set (similar to those summarized in the loading charts presented in this book). This data set is then analyzed to determine a mathematical relationship that has the highest level of statistical correlation. It is important to note that this method makes no attempt to describe the internal dynamics of the wetland system and is utterly dependent on the input–output data set being analyzed. Recent examples of this method are included in Crites *et al.* (2006) and Figure 8.27, Chapter 8, of this book.

Empirical equations can best be thought of as a sliding-scale "rule of thumb." All the limitations inherent in the loading chart method and scaling factors also apply to empirical equations. As a result, designers are cautioned that applying empirical equations to situations not representative of the parent data set will often result in inappropriate designs.

## 20.2 PERFORMANCE-BASED WETLAND SIZING

In performance-based design, the effects of degradation rate coefficients (*k*-rates), temperature (θ-factors), and the combined effects of internal hydraulics and pollutant weathering (*P*TIS) are considered discretely. The great advantage of this method is that it allows prediction of treatment wetland performance across different hydraulic and temperature regimes.

Historically, early HSSF designs were based on first-order modeling with the assumption of plug flow (EC/EWPCA Emergent Hydrophyte Treatment Systems Expert Contact Group and Water Research Centre, 1990). In the realm of proprietary wetland designs, this method was extended to waste-specific *k*-rates and pollutant weathering for soil-media HSSF wetlands (Kickuth, 2002), although these results were not reported to the international scientific community.

The first-order approach was largely abandoned in Europe in favor of simple scaling factors because the majority of HSSF wetlands were implemented for small flows of domestic wastewater under similar climatic conditions.

In North America, implementation of HSSF wetlands was less rapid than in Europe. However, there has been an expanding interest in using HSSF wetlands for applications other than domestic wastewater treatment, resulting in the continued development of sizing models using first-order kinetics (Kadlec and Knight, 1996).

This book includes a performance-based approach to HSSF wetland sizing using the first-order *P-k-C** model. Conceptually, this is a tank-in-series (TIS) model using a relaxed parameter, *P*, to account for both hydraulic and weathering effects. The *P-k-C** model is discussed in detail in Chapter 6.

This performance-based approach is in contrast to most of the currently existing wetland manuals, which advocate a prescriptive loading approach for HSSF wetland design. In this book, loading charts are proposed as a means to check the relative conservativeness of a proposed wetland sizing.

The *P-k-C** model has a number of advantages for HSSF wetland sizing, which include

- Loading charts cannot predict internal profiles of pollutant reduction, but this is easily done through first-order modeling.
- Differences in hydraulic efficiency or pollutant weathering can be explicitly addressed. This is especially important when designing full-scale wetlands based on pilot system data, which may have different hydraulic properties.
- The first-order approach allows the modeling of sequential removals, such as for nitrogen.
- Effects of a nonzero background concentration, *C**, may be explicitly addressed.

The procedure outlined here is for a constant influent flow situation. Different influent flow rates may need to be considered to deal with daily or seasonal peaking factors. Different seasons may need to be considered so that the design-limiting ("bottleneck") period of the year is identified and analyzed. The design-limiting period depends on the goals of the project and the climate of the project location. For instance, for projects in very cold climates, operation of the wetland without freezing during the winter months may be a design priority. In hot, arid climates, water loss due to evapotranspiration (*ET*) may be a major concern.

### Basic Application of the *P-k-C** Model to HSSF Wetlands

The simplest situation is where there are no gains of water from precipitation or losses of water due to *ET* or infiltration. This can be easily described using the *P-k-C** model (Figure 17.1, Chapter 17) and Equation 6.57 in Chapter 6, restated here as Equation 20.1:

$$\left(\frac{C - C^*}{C_i - C^*}\right) = \frac{1}{(1 + k/Pq)^P} = \frac{1}{(1 + k_V \tau/P)^P} \qquad (20.1)$$

**TABLE 20.1**
**Estimated Pollutant Reduction Using a First-Order (*P-k-C**) Model for Constant Flows (*P* = *ET*)**

| Input Parameters | | Calculated Values | |
|---|---|---|---|
| Flow rate, $Q$ | 20 $m^3$/d | Volume per tank | 14.3 $m^3$ |
| *P*TIS (system) | 4 | Area per tank | 125 $m^3$ |
| Area, $A$ | 500 $m^2$ | Influent flow, $Q_i$ | 20.0 $m^3$/d |
| Porosity, ε | 0.38 | Effluent flow, $Q_o$ | 20.0 $m^3$/d |
| Bed depth | 0.30 m | Average flow, $Q_{avg}$ | 20.0 $m^3$/d |
| $C_i$ | 200 mg/L | Effluent mass load | 551 g/d |
| $C^*$ | 8 mg/L | Nominal HRT | 2.85 d |
| $k$ | 45 m/yr | HRT based on $Q_{avg}$ | 2.85 d |
| $k$ | 0.123 m/d | HRT based on *P*TIS | 2.85 d |

| Calculated Values | | System In | Exit Tank 1 | Exit Tank 2 | Exit Tank 3 | Exit Tank 4 | System Out |
|---|---|---|---|---|---|---|---|
| Net flow | $m^3$/d | 20.0 | 20.0 | 20.0 | 20.0 | 20.0 | 20.0 |
| HLR, $q$ | m/d | 0.040 | 0.16 | 0.16 | 0.16 | 0.16 | 0.04 |
| Concentration, $C$ | mg/L | 200.0 | 116.4 | 69.2 | 42.6 | 27.5 | 27.5 |
| Nominal HRT | days | 2.85 | 0.71 | 0.71 | 0.71 | 0.71 | 2.85 |

where

$k$ = modified first-order areal rate constant, m/d
$k_V$ = modified first-order volumetric rate constant, $d^{-1}$
$P$ = apparent number of TIS

An example is detailed in Table 20.1. This hypothetical example is configured for a small HSSF wetland (100 PE) with a BOD loading of 40 g/PE and a flow rate of 200, L/PE ($C_i$ = 200 mg/L). A $k$-rate of 45 m/yr is selected based on the distribution shown in Table 8.7 Chapter 8. The wetland is assumed to hydraulically function as eight tanks in series (see Table 6.2, Chapter 6); a reduced value of $P$ = 4 has been selected here to account for weathering of the BOD mixture as it passes through the wetland. The background concentration, $C^*$, has been estimated at 8 mg/L based on Figure 8.27, Chapter 8.

Equation 20.1 can be solved directly, or the calculation can be completed one tank at a time using a computer spreadsheet, as shown in Table 20.1.

## Water Budget Effects

If there is a net gain or loss of water from the wetland, effluent concentrations will be different from those predicted by Equation 20.1. For HSSF wetlands operating under hot, arid conditions, water loss from *ET* may be a significant design concern, and the effects of the water budget should be considered.

The annual water budget forms the basis for a first approach to understanding pollutant reductions and the area required. Under the assumption of a constant water level, the flows from the wetland may be computed from the influent flow rate and meteorological data from the vicinity of the project site. Calculation procedures have been established in Chapter 2.

The inlet hydraulic loading will be increased by rainfall (≈0.5–1.5 m/yr) and decreased by *ET* (≈0.5–1.5 m/yr). However, there may be seasonal imbalances. These amounts are important if the wetland is to have a very low hydraulic loading, or correspondingly, a long detention time. Net evapotranspiration (*ET* > *P*) has two effects: lengthening of detention time and concentration of dissolved constituents. The inverse is true in net precipitation environments (*P* > *ET*). The use of an average flow rate compensates for altered detention time but not for dilution or concentration. Therefore, it may be prudent to consider worst-case seasonal conditions when calculating water-budget effects on the wetland treatment performance.

Pollutant mass balances are conducted on a cells-in-series basis (see Figure 20.2). Results of the overall water mass balance are apportioned to the cells according the chosen number of TIS. For the first unit in the series (Equation 6.66 in Chapter 6, restated here as Equation 20.2),

$$Q_1 = Q_i + A_1(P - ET - I) \tag{20.2}$$

where

$A_1$ = area of the first segment (tank), $m^2$
$ET$ = evapotranspiration, m/d
$I$ = infiltration, m/d
$P$ = precipitation, m/d
$Q_i$ = inlet flow rate, $m^3$/d
$Q_1$ = outlet flow rate from segment #1, $m^3$/d

The additional data input requirements for the water mass balances are

1. Inflow ($Q_i$)
2. Precipitation (P)
3. Evapotranspiration (*ET*)
4. Infiltration (*I*)
5. Area (*A*)

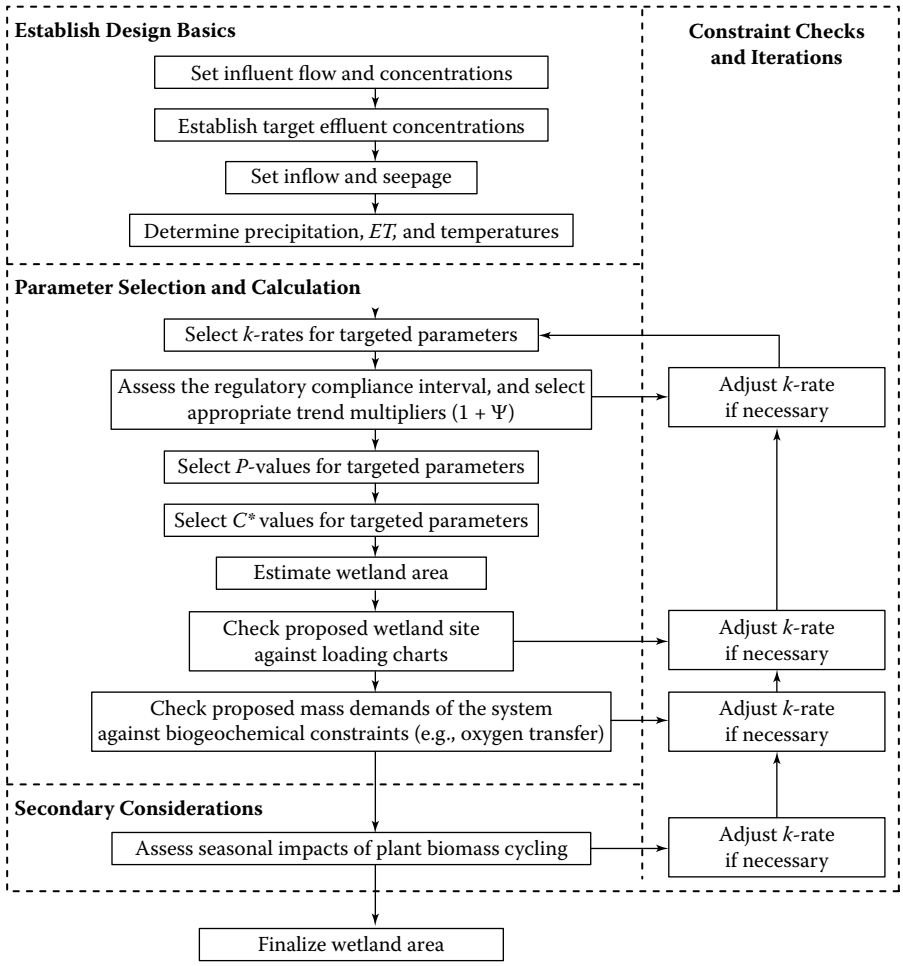



**FIGURE 20.2** Design process flow-chart for HSSF wetlands.

The example of Table 20.1 is continued forward to examine a situation where there is a net loss of water because of *ET*. Results of the hydraulic changes are summarized in Table 20.2. For ease of comparison, the parameter $P$ has been kept at $P = 4$ (even though a higher value could be justified in the absence of pollutant weathering).

As seen in Table 20.2, there is a net change in the flow rate, hydraulic loading, and detention time as water flows through the wetland reaction segments, because in this particular example, approximately 25% of the water flow is lost through the system. Averaging the inlet and outlet flows does not fully account for these effects.

## Pollutant Mass Balances

If there is a net gain or loss of water within the system, the flow in each reaction compartment is different, and it becomes necessary to calculate contaminant removals on a mass basis, as concentration alone no longer adequately describes system performance.

This computation is then repeated for the remaining segments, in each case using the outlet concentrations and flows from the preceding segment. The wetland outlet concentration is that exiting from the final reaction segment. The mass output is calculated from the effluent flow rate and output concentration.

As the flow rate is different for each reaction segment, Equation 20.1 cannot be used to directly calculate the result. Instead, calculations must be carried out sequentially for each reaction segment (tank). This is most easily done using a computer spreadsheet.

Comparison of Table 20.1 with Table 20.2 illustrates that the effluent concentrations predicted by these methods are almost identical. Water is lost to *ET*, which concentrates pollutants within the wetland. However, the volume reduction results in lower hydraulic loading rates (longer detention times), so a greater degree of treatment also occurs within the wetland reactor. Use of average flow accounts for only the longer detention and not the evaporative concentration.

Reviewing the system performance from a mass load basis reveals a very different picture. In the example of Table 20.1, the assumption is of no water loss, so the effluent mass load is 27.5 g/m$^3$ × 20 m$^3$/d, or 551 g/d. In the example of Table 20.2, the effluent flow has been reduced by 24%, so the effluent mass load is 27.6 g/m$^3$ × 15.2 m$^3$/d, or 420 g/d.

**TABLE 20.2**
**Example of BOD Reduction under Variable Flows (*ET* > *P*)**

| Input Parameters | | Calculated Values | |
|---|---|---|---|
| Flow rate, $Q$ | 20 $m^3/d$ | Volume per tank | 14.25 $m^2$ |
| Precipitation, $P$ | 0.5 mm/d | Area per tank | 125 $m^2$ |
| $ET$ | 10 mm/d | Influent flow, $Q_i$ | 20.00 $m^3/d$ |
| Infiltration | 0.02 mm/d | Effluent flow, $Q_o$ | 15.24 $m^3/d$ |
| *P*TIS (system) | 4 | Average flow, $Q_{avg}$ | 17.62 $m^3/d$ |
| Area, $A$ | 500 $m^2$ | Effluent mass load | 420 g/d |
| Porosity, ε | 0.38 | Nominal HLR | 0.0400 m/d |
| Bed depth | 0.30 m | HLR based on $Q_{avg}$ | 0.0352 m/d |
| $C_i$ | 200 mg/L | HLR based on *P*TIS | 0.0341 m/d |
| $C^*$ | 8 mg/L | Nominal HRT | 2.85 d |
| $k$ | 45 m/yr | HRT based on $Q_{avg}$ | 3.23 d |
| $k$ | 0.123 m/d | HRT based on *P*TIS | 3.74 d |

| Calculated Values | | System In | Exit Tank 1 | Exit Tank 2 | Exit Tank 3 | Exit Tank 4 | System Out |
|---|---|---|---|---|---|---|---|
| Net flow | $m^3/d$ | 20.0 | 18.8 | 17.6 | 16.4 | 15.2 | 15.2 |
| Precipitation | $m^3/d$ | 0.063 | 0.063 | 0.063 | 0.063 | 0.063 | — |
| $ET$ | $m^3/d$ | 1.250 | 1.250 | 1.250 | 1.250 | 1.250 | — |
| Infiltration | $m^3/d$ | 0.003 | 0.003 | 0.003 | 0.003 | 0.003 | — |
| HLR, $q$ | m/d | 0.040 | 0.150 | 0.141 | 0.131 | 0.122 | 0.034 |
| Nominal HRT | days | 2.85 | 0.76 | 0.81 | 0.87 | 0.94 | 3.74 |
| Concentration, $C$ | mg/L | 200.0 | 120.5 | 72.3 | 43.9 | 27.6 | 27.6 |

The difference in the predicted effluent mass load can be approximated by the ratio $a = (P–ET)/q_i$, which is the atmospheric augmentation or deficit. The fractional error in a first-order-model prediction of mass load is approximately equal to $a$, for $a > -0.5$. Thus, if 25% of the inflow evaporates, the effluent mass load will be overpredicted by about 25% unless flow corrections are made. If the wetland gains water through precipitation, the inverse is true. It should be noted that averaging the flow will not completely address this problem. Predicted effluent mass loads for the flow assumptions presented in Tables 20.1–20.3 are summarized in Table 20.4.

### Interconnected Pollutants

Some pollutants have sequential degradation processes and thus require a linkage of the mass balances between the parent and daughter processes. A classic example of this is nitrogen: nitrogen species interconvert, thereby linking the mass balances for organic, ammonia, and oxidized nitrogen. It is sometimes possible to disconnect these species, as, for instance, in the case of wetlands that receive nitrate but little or no organic or ammonia nitrogen. However, in many cases it is necessary to account for the (internal) production of ammonia from organic sources—the incoming water or the decomposition of wetland necromass—and the internal production of oxidized nitrogen. The reaction sequence has been presented in Equation 17.8, Chapter 17 (restated here as Equation 20.3). A simple presumed chemistry is

$$\mathrm{ORG-N} \rightarrow \mathrm{NH_4-N} \rightarrow \mathrm{NO_x-N} \rightarrow \mathrm{N_2} \qquad (20.3)$$

In a simplified version of this analysis, uptake and return from biomass is not included. The effects of biogeochemical cycling will be explored in a latter part of the design process. In the case of nitrogen, the three mass balances become

**TABLE 20.3**
**Effluent Pollutant Mass Loads (Based on the *P-k-C** Model) under Different Flow Scenarios**

| Method of Calculation | Predicted Effluent Flow Rate ($m^3/d$) | Predicted Effluent Concentration (mg/L) | Predicted Effluent Mass Load (g/d) |
|---|---|---|---|
| *P-k-C**, based on constant flow | 20 | 27.5 | 551 |
| *P-k-C**, based on average flow | 17.6 | 27.6 | 486 |
| *P-k-C**; *P-ET* corrections based on *P*TIS | 15.2 | 27.6 | 420 |

**TABLE 20.4**
**BOD Removal Based on Performance-Based Criteria; Constant Flow Assumption**

| Input Parameters | | Calculated Values | |
|---|---|---|---|
| Flow rate, $Q$ | 20 m³/d | Volume per tank | 18.5 m³ |
| *P*TIS (system) | 4 | Area per tank | 162.5 m³ |
| Area, $A$ | 650 m² | Influent flow, $Q_i$ | 20.0 m³/d |
| Porosity, ε | 0.38 | Effluent flow, $Q_o$ | 20.0 m³/d |
| Bed depth | 0.30 m | Average flow, $Q_{avg}$ | 20.0 m³/d |
| $C_i$ | 200 mg/L | Effluent mass load | 399 g/d |
| $C^*$ | 8 mg/L | Nominal HRT | 3.71 d |
| $k$ | 45 m/yr | HRT based on $Q_{avg}$ | 3.71 d |
| $k$ | 0.123 m/d | HRT based on *P*TIS | 3.71 d |

| Calculated Values | | System In | Exit Tank 1 | Exit Tank 2 | Exit Tank 3 | Exit Tank 4 | System Out |
|---|---|---|---|---|---|---|---|
| Net flow | m³/d | 20.0 | 20.0 | 20.0 | 20.0 | 20.0 | 20.0 |
| HLR, $q$ | m/d | 0.031 | 0.12 | 0.12 | 0.12 | 0.12 | 0.031 |
| Concentration, $C$ | mg/L | 200.0 | 103.9 | 55.9 | 31.9 | 20.0 | 20.0 |
| Nominal HRT | days | 3.71 | 0.93 | 0.93 | 0.93 | 0.93 | 3.71 |

linked, and the tank equations are

$$Q_1 C_{O1} = (Q_{in} C_{O,in}) - (I\ A_1 C_{O1}) - k_O A_1 (C_{O1} - C_O^*) \quad (20.4)$$

$$Q_1 C_{A1} = (Q_{in} C_{A,in}) - (I \cdot A_1 C_{A1}) - k_A A_1 (C_{A1} - C_A^*) + k_O A_1 (C_{O1} - C_O^*) \quad (20.5)$$

$$Q_1 C_{N1} = (Q_{in} C_{N,in}) - (I \cdot A_1 C_{N1}) - k_N A_1 (C_{N1} - C_N^*) + k_A A_1 (C_{A1} - C_A^*) \quad (20.6)$$

where

$C_O$ = organic N concentration, mg/L
$C_A$ = ammonia N concentration, mg/L
$C_N$ = oxidized N concentration, mg/L
$k_O$ = organic N rate coefficient, m/d
$k_A$ = ammonia N rate coefficient, m/d
$k_N$ = oxidized N rate coefficient, m/d

and the subscript "in" denotes parameters associated with the influent flow for each nitrogen form.

These may be rearranged to solve the outlet concentrations for each tank:

$$C_{O1} = \left( \frac{Q_{in} C_{O,in} + k A_1 C_O^*}{Q_1 + I A_1 + k_O A_1} \right) \quad (20.7)$$

$$C_{A1} = \left( \frac{Q_{in} C_{A,in} + k_A A_1 C_A^* + k_O (C_{O1} - C_O^*)}{Q_1 + I A_1 + k_A A_1} \right) \quad (20.8)$$

$$C_{N1} = \left( \frac{Q_{in} C_{N,in} + k_N A_1 C_N^* + k_A (C_{A1} - C_A^*)}{Q_1 + I A_1 + k_N A_1} \right) \quad (20.9)$$

Equation 20.7 represents the degradation of organic nitrogen. Equations 20.8 and 20.9 contain extra production terms from ammonification in the ammonia balance and from nitrification in the oxidized nitrogen balance. The three must be solved sequentially—20.7, followed by 20.8, followed by 20.9. This can be done through an expanded process of the computer spreadsheet illustrated in Table 20.3, with one sequence of calculations for organic nitrogen, another sequence of calculations for ammonia, and a final sequence of calculations for oxidized nitrogen.

### More than One Parameter

Multi-parameter wetland sizing has been previously summarized in Tables 17.4 and 17.5, Chapter 17, for FWS systems; the same mathematical process can be applied to SSF wetlands.

## 20.3 ACCOMPLISHING PERFORMANCE-BASED SIZING FOR HSSF WETLANDS

The procedure explored here analyzes the effect of changing the wetland area (or equivalently, detention time) on the forecasted effluent concentrations of contaminants. It is presumed that the designer will conduct the design calculations via a computer spreadsheet so that design changes may be easily explored as an iterative process.

The general performance-based sizing algorithm has been previously discussed in Chapter 15 for FWS wetlands. A modified version, more specific to the typical needs and priorities for SSF wetlands, is presented here:

### Establish the design basis

1. Set inlet concentrations.
2. Set target effluent concentrations (regulatory limits and exceedance factors).

3. Set inflow and seepage (typically zero if synthetic liners are used).
4. Determine precipitation, *ET*, and temperature for critical seasons and targeted outflow rates.

**Parameter selection and primary sizing calculations**

5. Select *k*-rates for the targeted parameters.
6. Assess the regulatory compliance interval, and select appropriate trend multipliers ($1 + \Psi$).
7. Select the *P*TIS value. Since *P* incorporates both the hydraulic efficiency (number of TIS) and pollutant weathering effects, the selected *P*-value may not be the same for all targeted parameters.
8. Select *C** parameters for the targeted parameters.
9. Adjust the wetland area until design goals are met.
10. Check the proposed wetland size against loading charts (if available) to assess the relative conservativeness of the design for the targeted parameters.
11. Check the proposed mass demands of the system against biogeochemical constraints (e.g., oxygen transfer).
12. Adjust the wetland area until design goals are met.

**Secondary design considerations**

13. Assess the seasonal impact of plant biomass cycling (potentially important if nitrogen is a targeted parameter).
14. Modify the wetland area, if necessary, to meet secondary considerations.

This is necessarily an iterative process, and the designer may have to iterate at several stages in the overall design process, as illustrated in Figure 20.2.

Of necessity, performance-based wetlands require more mathematical effort than scaling rules, which is easily accomplished using spreadsheets and judicious selection of design parameters. This book provides a variety of resources to assist the designer in using performance-based design tools. These include:

***k*-rate, *C**, *P*TIS**

| | |
|---|---|
| BOD | Table 8.8 |
| Organic Nitrogen | Table 9.12 |
| TKN | Tables 9.15, 9.33 |
| Total nitrogen | Table 9.19 |
| Ammonia (nitrification) | Tables 9.25, 9.33 |
| Nitrate (denitrification) | Table 9.39 |
| Pathogens | Tables 12.15–12.19 |

**Temperature effects (θ-factors)**

| | |
|---|---|
| BOD | Table 8.9 |
| Organic nitrogen | Table 9.13 |
| TKN | Table 9.17 |
| Ammonia (nitrification) | Table 9.32 |

**Seasonal and stochastic variability ($1 + \Psi$), based on currently available data**

| | |
|---|---|
| BOD | Tables 8.11, 8.12 |
| Ammonia | Table 9.35 |
| Total phosphorus | Table 10.16 |

## Conservatism in Design

Part of the inherent challenge of using a general design model (such as the *P-k-C** model) is that there are so many variables that can be adjusted by the designer. For instance, the designer can introduce conservatism into the design model at many stages, including:

- Choosing a *k*-rate at the lower range of the frequency distribution
- Incorporating effluent multipliers for seasonal variations (see Table 8.13, Chapter 8, for example)
- Choosing a lower value of *P* (lower hydraulic efficiency and/or greater effect of pollutant weathering)
- Selecting a high concentration for *C** (limiting the level of treatment the wetland can achieve)
- Choosing a higher θ-factor (more temperature sensitivity); this is a conservative design assumption in cold climates

A designer who selects the most conservative value for each of these parameters will have an overly conservative design that ignores the many thousands of successful SSF wetland systems operating around the world. The current state-of-the-art design knowledge requires judicious selection of design parameters. The recommended approach in this book is to select median parameters for *P*, *k*, *C**, and θ, predict the system performance, and adjust using variability tables (such as those presented in Tables 8.13, 9.35, and 10.16). Of course, if the physical configuration of the wetland is different than these parent data sets, then designers must use their best professional judgment to select appropriate design parameters, because existing performance data is unlikely to represent the altered wetland configuration.

Fortunately, loading charts (such as those presented in Figures 7.30, 8.26, 9.18, 9.24, 9.30, 9.38, 10.36, and 12.12) provide a simple means to check the relative conservatism of a wetland design against the body of knowledge accumulated from wetland systems already in operation. This iterative check against loading chart data is also described in Figure 20.3.

Let us consider the example wetland previously described in Tables 20.1 and 20.2; the wetland sizing will be determined through system performance rather than a prescriptive scaling factor. In Table 20.1, the HSSF wetland size was set at 5 $m^2$/PE, which is a scaling factor commonly used in Europe (ATV, 1998). In this example, we will assume that the inlet flow of 20 $m^3$/d and inlet BOD concentration of 200 mg/L remain the same and the wetland must achieve a 90% BOD mass load reduction. We will further assume that the *k*-rate

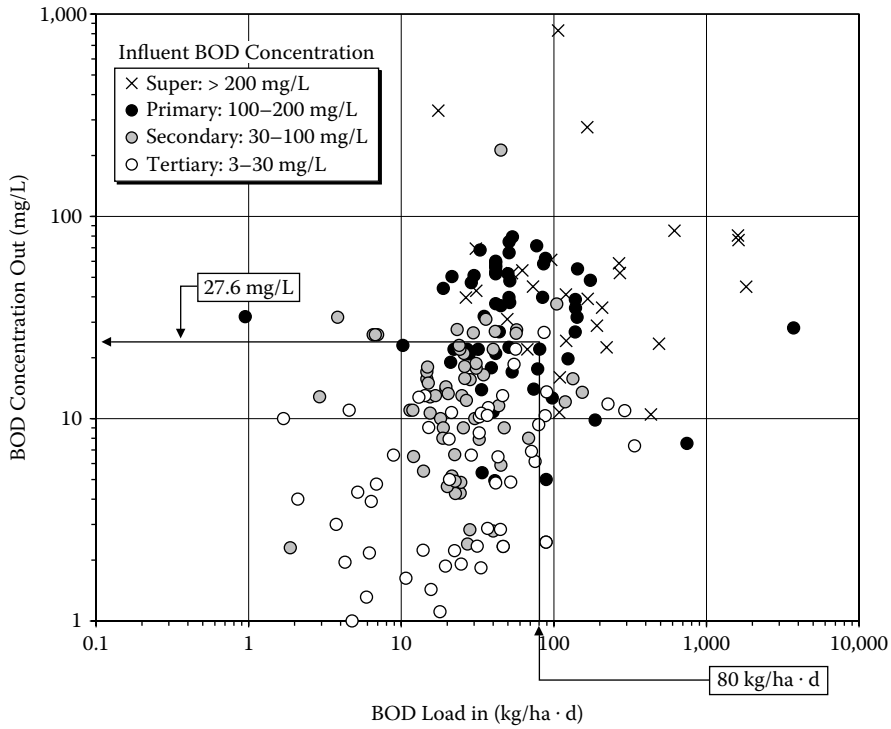



**FIGURE 20.3** Comparison of the *P-k-C** model against existing performance data based on Table 20.2.

(45 m/yr) and *P*TIS (*P* = 4) parameters remain the same as the previous examples for ease of comparison.

### Most Basic Case: Consideration of Concentration Reduction Only, No Change in Flow

If it is assumed that there is no change in flow, then a 90% reduction in mass load corresponds to a 90% reduction in concentration. Equation 20.1 can be used (with some algebraic manipulation) to directly calculate the area required, or a spreadsheet-based approach, such as the one presented in Table 20.1, can be adopted. The spreadsheet can be manually iterated by the designer, or the iteration process can be automated using functions such as the Solver™ routine in Microsoft Excel™. An example of this spreadsheet-based approach (for the assumption of no change in flow) is shown in Table 20.4. The assumption of $C^* = 8$ mg/L has been retained from the previous example.

Table 20.4 summarizes the manual iteration of Table 20.1. In just three manual iterations, a solution of 650 $m^2$ (6.5 $m^2$/PE) is arrived at. This is a prediction of the median annual performance of the wetland under a constant-flow situation ($P = ET$).

### Second Case: Pollutant Reductions under Variable Flow

If precipitation and *ET* are not equal, there will be differences between the calculations for concentration reduction and mass load reduction. The spreadsheet example of Table 20.2 has been utilized here to iterate a solution of 90% mass load reduction for the high *ET* season previously considered. Results are summarized in Table 20.5.

The spreadsheet of Table 20.5 was manually iterated (three times) to converge on a wetland size of 515 $m^2$ (5.15 $m^2$/PE) to achieve the required mass load reduction on median annual basis.

### Role of C* in Pollutant Reduction

The pollutant removal rate, *k*, and the background concentration, *C**, are independent parameters of the wetland under consideration. However, they are mathematically linked in the sizing process. To use the analogy of descending in an elevator through a building, *k* describes how rapidly you drop, and *C** determines what floor you arrive at. A high value of *k* implies a rapid drop from the initial inlet concentration, and a low level of *C** implies that the wetland can remove pollutants to low concentrations (if sized large enough).

Let us consider the case of Table 20.5, with the altered assumption that *C** is now 20 mg/L, not 8 mg/L. This situation could easily occur if there is a significant portion of BOD that is from nondomestic sources. Olive mill processing wastewater and landfill leachates are two examples where a higher level of *C** may be justified. The results are summarized in Table 20.6, which indicates that the wetland must be 700 $m^2$ (7.0 $m^2$/PE) to achieve the design goal of 90% mass load reduction on a median annual basis (even though the effluent concentration leaving the wetland

## TABLE 20.5
## Performance-Based Sizing of HSSF Wetlands under Variable Flows to Achieve Targeted Mass Load Reduction

| Input Parameters | |
|---|---|
| Flow rate, $Q$ | 20 $m^3/d$ |
| Precipitation, P | 0.5 mm/d |
| *ET* | 10 mm/d |
| Infiltration | 0.02 mm/d |
| *P*TIS (system) | 4 |
| Area, $A$ | 515 $m^2$ |
| Porosity, ε | 0.38 |
| Bed depth | 0.30 m |
| $C_i$ | 200 mg/L |
| $C^*$ | 8 mg/L |
| $k$ | 45 m/yr |
| $k$ | 0.123 m/d |

| Calculated Values | |
|---|---|
| Volume per tank | 14.68 $m^2$ |
| Area per tank | 128.75 $m^2$ |
| Influent flow, $Q_i$ | 20.00 $m^3/d$ |
| Effluent flow, $Q_o$ | 15.10 $m^3/d$ |
| Average flow, $Q_{avg}$ | 17.55 $m^3/d$ |
| Influent mass load | 4,000 g/d |
| Effluent mass load | 400 g/d |
| Mass % reduction | 90% |
| Nominal HLR | 0.039 m/d |
| HLR based on $Q_{avg}$ | 0.034 m/d |
| HLR based on *P*TIS | 0.033 m/d |
| Nominal HRT | 2.94 d |
| HRT based on $Q_{avg}$ | 3.35 d |
| HRT based on *P*TIS | 3.89 d |

| Calculated Values | | System In | Exit Tank 1 | Exit Tank 2 | Exit Tank 3 | Exit Tank 4 | System Out |
|---|---|---|---|---|---|---|---|
| Net flow | $m^3/d$ | 20.0 | 18.8 | 17.5 | 16.3 | 15.1 | 15.1 |
| Precipitation | $m^3/d$ | 0.064 | 0.064 | 0.064 | 0.064 | 0.064 | — |
| *ET* | $m^3/d$ | 1.288 | 1.288 | 1.288 | 1.288 | 1.288 | — |
| Infiltration | $m^3/d$ | 0.003 | 0.003 | 0.003 | 0.003 | 0.003 | — |
| HLR, $q$ | m/d | 0.039 | 0.146 | 0.136 | 0.127 | 0.117 | 0.033 |
| Nominal HRT | days | 2.94 | 0.78 | 0.84 | 0.90 | 0.97 | 3.89 |
| Concentration, $C$ | mg/L | 200.0 | 119.1 | 70.7 | 42.5 | 26.5 | 26.5 |

## TABLE 20.6
## Performance-Based Sizing of HSSF Wetlands under Variable Flows to Achieve Targeted Mass Load Reduction; High $C^*$ Assumption

| Input Parameters | |
|---|---|
| Flow rate, $Q$ | 20 $m^3/d$ |
| Precipitation, $P$ | 0.5 mm/d |
| *ET* | 10 mm/d |
| Infiltration | 0.02 mm/d |
| *P*TIS (system) | 4 |
| Area, $A$ | 700 $m^2$ |
| Porosity, ε | 0.38 |
| Bed depth | 0.30 m |
| $C_i$ | 200 mg/L |
| $C^*$ | 20 mg/L |
| $k$ | 45 m/yr |
| $k$ | 0.123 m/d |

| Calculated Values | |
|---|---|
| Volume per tank | 19.98 $m^3$ |
| Area per tank | 175.25 $m^3$ |
| Influent flow, $Q_i$ | 20.00 $m^3/d$ |
| Effluent flow, $Q_o$ | 13.33 $m^3/d$ |
| Average flow, $Q_{avg}$ | 16.66 $m^3/d$ |
| Influent mass load | 4,000 g/d |
| Effluent mass load | 400 g/d |
| Mass % reduction | 90% |
| Nominal HLR | 0.029 m/d |
| HLR based on $Q_{avg}$ | 0.024 m/d |
| HLR based on *P*TIS | 0.023 m/d |
| Nominal HRT | 4.00 d |
| HRT based on $Q_{avg}$ | 4.80 d |
| HRT based on *P*TIS | 6.00 d |

| Calculated Values | | System In | Exit Tank 1 | Exit Tank 2 | Exit Tank 3 | Exit Tank 4 | System Out |
|---|---|---|---|---|---|---|---|
| Net flow | $m^3/d$ | 20.0 | 18.3 | 16.7 | 15.0 | 13.3 | 13.3 |
| Precipitation | $m^3/d$ | 0.088 | 0.088 | 0.088 | 0.088 | 0.088 | — |
| *ET* | $m^3/d$ | 1.753 | 1.753 | 1.753 | 1.753 | 1.753 | — |
| Infiltration | $m^3/d$ | 0.004 | 0.004 | 0.004 | 0.004 | 0.004 | — |
| HLR, $q$ | m/d | 0.029 | 0.105 | 0.095 | 0.086 | 0.076 | 0.023 |
| Nominal HRT | days | 4.00 | 1.09 | 1.20 | 1.33 | 1.50 | 6.00 |
| Concentration, $C$ | mg/L | 200.0 | 111.0 | 64.4 | 41.1 | 30.0 | 30.0 |

is 30 mg/L). Thus, the effect of a higher background concentration (in this example) increases the wetland area 36% to meet the same annual performance goal.

### Seasonal and Stochastic Variability

The rate-constant procedure to this point can be considered complete if monitoring the effluent is not required, or the system is only required to meet an annual average limit. However, many systems (especially those discharging to surface waters) will have a more frequent compliance interval (typically monthly or weekly).

In these situations, designing the wetland based on median annual performance is not acceptable because half the effluent values will be above the targeted permit concentration. In these situations, some factors must be introduced into the design to account for seasonal and stochastic changes in effluent quality. There are three basic routes open to the designer to account for these effects:

- Use of temperature correction (θ) factors (e.g. Tables 8.8, 8.9, 9.13, 9.17, 9.32). These factors can be used to adjust removal rates based on temperature dependence; however, many removal processes in HSSF wetlands are not temperature-dependent (with the exception of the nitrogen species); or
- Use of cosine-fitting parameters to account for seasonal changes in effluent concentrations (Table 8.11, Table 10.19); plus
- Use of variability parameters ($1 + \Psi$) to account for short-term fluctuations in effluent quality (Tables 8.12, 9.35, and 10.20).

The example situation considered here is if the specified wetland performance (90% mass load reduction in BOD) must be met on a monthly (as opposed to an annual) basis. Use of variability tools such as those shown in Table 8.12 can assist in this matter; selection of a 90th percentile trend multiplier (1.78) can be used to account for stochastic variations.

The example spreadsheet of Table 20.5 ($C^*$ assumed to be 8 mg/L) is again manually iterated to determine the required wetland size; results are shown in Table 20.7.

Results of Table 20.7 (manually iterated four times) indicate that the required wetland area needed to achieve a 90th percentile of non-compliance (with $C^* = 8$ mg/L) is 719 $m^2$ (7.19 $m^2$/PE) under the variable-flow situation. This is in comparison to Table 20.4, which indicates that the required wetland area is only 5.15 $m^2$/PE to meet the annual load-reduction goals. In other words, as the regulatory compliance interval is shortened, the wetland size must be increased. In this particular example, the size of the wetland must be increased 40% to address seasonal and stochastic

**TABLE 20.7**
**Mass Load Reduction to Account for Seasonal and Stochastic Variability Including the Assumption of Variable Flows**

| Input Parameters | |
|---|---|
| Flow rate, $Q$ | 20 $m^3$/d |
| Precipitation, $P$ | 0.5 mm/d |
| $ET$ | 10 mm/d |
| Infiltration | 0.02 mm/d |
| $P$TIS (system) | 4 |
| Area, $A$ | 719 $m^2$ |
| Porosity, ε | 0.38 |
| Bed depth | 0.30 m |
| $C_i$ | 200 mg/L |
| $C^*$ | 8 mg/L |
| $k$ | 45 m/yr |
| $k$ | 0.123 m/d |

| Calculated Values | |
|---|---|
| Volume per tank | 20.49 $m^3$ |
| Area per tank | 179.75 $m^3$ |
| Influent flow, $Q_i$ | 20.0 $m^3$/d |
| Effluent flow, $Q_o$ | 13.16 $m^3$/d |
| Average flow, $Q_{avg}$ | 16.58 $m^3$/d |
| Influent mass load | 4,000 g/d |
| Effluent mass load | 225 g/d |
| Mass % reduction | 94.4% |
| Nominal HLR | 0.028 m/d |
| HLR based on $Q_{avg}$ | 0.023 m/d |
| HLR based on $P$TIS | 0.022 m/d |
| Nominal HRT | 4.10 d |
| HRT based on $Q_{avg}$ | 4.94 d |
| HRT based on $P$TIS | 6.23 d |

| Calculated Values | | System In | Exit Tank 1 | Exit Tank 2 | Exit Tank 3 | Exit Tank 4 | System Out |
|---|---|---|---|---|---|---|---|
| Net flow | $m^3$/d | 20.0 | 18.3 | 16.6 | 14.9 | 13.2 | 13.2 |
| Precipitation | $m^3$/d | 0.090 | 0.090 | 0.090 | 0.090 | 0.090 | — |
| $ET$ | $m^3$/d | 1.798 | 1.798 | 1.798 | 1.798 | 1.798 | — |
| Infiltration | $m^3$/d | 0.004 | 0.004 | 0.004 | 0.004 | 0.004 | — |
| HLR, $q$ | m/d | 0.028 | 0.102 | 0.092 | 0.083 | 0.073 | 0.022 |
| Nominal HRT | days | 4.10 | 1.12 | 1.24 | 1.38 | 1.56 | 6.23 |
| Concentration, $C$ | mg/L | 200.0 | 103.3 | 53.3 | 28.7 | 17.1 | 17.1 |

variability in order to meet the targeted load-reduction goal 90% of the time.

An alternate approach would be to construct the wetland as two cells in series. This would conceptually increase the hydraulic efficiency of the wetland, which could be approximated by increasing the *P*-value. If the *P*-value is increased to seven (instead of the original value of four), the required wetland size would be only 662 $m^2$ under the assumption of variable flows, $C^* = 8$ mg/L, instead of 719 $m^2$ (Table 20.7); the results are summarized in Table 20.8.

For this particular example, increasing the hydraulic efficiency of the wetland (by construction as two cells in series) reduces the required area by 8%; however, this increased treatment efficiency comes at a cost; additional berms and water level control structures must be constructed. So, at the end of the design process, the designer must balance the needs for treatment efficiency against the cost of construction. The designer could accept a larger (but less efficient) wetland area or construct a smaller, and more efficient, system, but at the added cost of additional berms and control structures.

## Cross-Checks against Existing Performance Data

As illustrated in Tables 20.4–20.8, application of the *P-k-C** model can yield performance predictions across a wide range of operating and hydraulic regimes. However, the indiscriminate use of this mathematical "tool kit" across all potential operating regimes cannot be recommended, because at present there is insufficient calibration data to address all conceivable operating conditions. Further, it will not be obvious whether the calculations exceed the experience envelope, that is, whether there have been extrapolations. Extrapolations can be extremely dangerous and must be avoided. The following sections of this chapter explore the constraints on the *P-k-C** model that a prudent designer would cross-check before finalizing the design sizing of a treatment wetland system.

Let us consider the example of Table 20.7. In this case, the designer has decided that a wetland of 719 $m^2$ (7.19 $m^2$/PE) should be sufficient to meet the targeted performance goal of a 90% mass load reduction in BOD under a variable-flow situation, and the size of the wetland has been further increased. How do real-world wetland systems perform in this loading range? Loading charts (such as shown in Figure 8.26) can be used to cross-check the forecasted wetland performance against operating data from existing wetland systems.

An example of such a cross-check is illustrated in Figure 20.3. In this example, the predicted wetland performance shown in Table 20.2 has been plotted on Figure 8.26. At a sizing of 5 $m^2$/PE, the equivalent influent BOD loading is 8 g/$m^2$·d (80 kg/ha·d) and the forecasted wetland concentration is 27.6 mg/L. The loading chart cross-check indicates that this performance forecast is close to the center of the data cloud; many existing wetlands operate in this performance

**TABLE 20.8**
**Decrease in Required Area if the Wetland Is Constructed as Two Cells in Series (Compare to Table 20.7)**

| Input Parameters | | Calculated Values | |
|---|---|---|---|
| Flow rate, $Q$ | 20 $m^3$/d | Volume per tank | 10.8 $m^2$ |
| Precipitation, $P$ | 0.5 mm/d | Area per tank | 94.6 $m^2$ |
| $ET$ | 10 mm/d | Influent flow, $Q_i$ | 20.00 $m^3$/d |
| Infiltration | 0.02 mm/d | Effluent flow, $Q_o$ | 13.70 $m^3$/d |
| *P*TIS (system) | 7 | Average flow, $Q_{avg}$ | 16.85 $m^3$/d |
| Area, $A$ | 662 $m^2$ | Influent mass load | 4,000 g/d |
| Porosity, ε | 0.38 | Effluent mass load | 225 g/d |
| Bed depth | 0.30 m | Mass % reduction | 94.4% |
| $C_i$ | 200 mg/L | Nominal HLR | 0.030 m/d |
| $C^*$ | 8 mg/L | HLR based on $Q_{avg}$ | 0.025 m/d |
| $k$ | 45 m/yr | HLR based on *P*TIS | 0.021 m/d |
| $k$ | 0.123 m/d | Nominal HRT | 3.77 d |
| | | HRT based on $Q_{avg}$ | 4.48 d |
| | | HRT based on *P*TIS | 5.51 d |

| Calculated Values | | System In | Exit Tank 1 | Exit Tank 2 | Exit Tank 3 | Exit Tank 4 | Exit Tank 5 | Exit Tank 6 | Exit Tank 7 | System Out |
|---|---|---|---|---|---|---|---|---|---|---|
| Net flow | $m^3$/d | 20.0 | 19.1 | 18.2 | 17.3 | 16.4 | 15.5 | 14.6 | 13.7 | 13.7 |
| Precipitation | $m^3$/d | 0.331 | 0.047 | 0.047 | 0.047 | 0.047 | 0.047 | 0.047 | 0.047 | — |
| $ET$ | $m^3$/d | 6.620 | 0.946 | 0.946 | 0.946 | 0.946 | 0.946 | 0.946 | 0.946 | — |
| Infiltration | $m^3$/d | 0.013 | 0.002 | 0.002 | 0.002 | 0.002 | 0.006 | 0.002 | 0.002 | — |
| HLR, $q$ | m/d | 0.030 | 0.202 | 0.192 | 0.183 | 0.173 | 0.164 | 0.154 | 0.145 | 0.021 |
| Nominal HRT | days | 3.77 | 0.56 | 0.59 | 0.62 | 0.66 | 0.70 | 0.74 | 0.79 | 5.51 |
| Concentration, $C$ | mg/L | 200.0 | 161.3 | 106.3 | 70.0 | 46.5 | 31.5 | 22.1 | 16.4 | 16.4 |

range, but the design is not particularly conservative (if the wetland being designed functions similar to the ones described by the performance data).

The design concept has here been illustrated for BOD, but most wetlands will be multifunctional and have multiple regulatory requirements. For instance, nitrogen, phosphorus, TSS, and pathogens may also need to be considered. Therefore, the design procedure is repeated for each design pollutant, and an area required for each determined. The largest area will be selected for the "bottleneck" pollutant. The design calculations for all other constituents must then be repeated for that area, resulting in overtreatment for all other constituents.

## Biogeochemical Cycle Constraints

Tools such as loading charts and *k*-rates can be used to predict system performance; however, it is important that the designer remains aware that the wetland is always an internally cycling biomass and is subject to other constraints, such as oxygen transfer and carbon supply, that can potentially limit system performance. These constraints are not explicitly addressed either in the loading chart or in the *P-k-C** approach; therefore, it is important to cross-check the wetland design against biogeochemical constraints.

### Biomass Cycling

The annual cycle of plant growth, death, and decay imposes a set of cycling processes that exchange carbon, nitrogen, and phosphorus between different compartments within the wetland, as discussed in Chapters 3, 8, 9, and 10. For the HSSF wetlands designed for simple removals, such as BOD, biomass cycling is probably not an important factor (Langergraber, 2005) and can be lumped into the seasonal and stochastic variability tables previously described. However, in other instances, plant biomass cycling may be a significant factor in system performance, and the use of biomass cycling parameters, such as those summarized in Table 17.7, may be used as a cross-check of system performance.

As an illustration, we will here consider the performance of a HSSF wetland cell at New Hanover County, North Carolina (Liehr *et al.*, 2000). This particular cell was supplied with a nitrified landfill leachate that had low BOD levels. Denitrification performance was poor and assumed to be limited by the availability of organic carbon. In 1997, the wetland was loaded at a rate of 628 g/m·yr of nitrogen, 92% of which was nitrate.

Table 17.6 in Chapter 17 suggests that approximately 30 g/m$^2$·yr could have been taken up by the vegetation (moderate second-year growth). The small amount of ammonia (TKN) was about sufficient to meet that need. The amount of organic carbon necessary to fuel conventional denitrification would be 691 g/m·yr of carbon (Table 17.7, Chapter 17). A review of Table 17.6 indicates that if the system was considered to be "nutrient moderate," then there would clearly be insufficient organic carbon from the plant biomass cycle (2,000 g/m$^2$·yr × 10% = 200 gC/m$^2$·yr). At higher plant densities, Table 17.6 indicates there still would not be sufficient organic carbon produced in the biomass cycle. Also, the designer must consider the spatial distribution of the biomass; aboveground plant materials are deposited on the surface of the HSSF bed, and may not readily degrade (within one annual cycle) and return carbon to the water column. (This phenomena is exacerbated under hot, arid conditions when aboveground biomass desiccates). The New Hanover County wetland in 1997 was still a developing system in a phase of plant grow-in during this study; so little biomass degradation products were presumably returned to the water column.

The BOD in the incoming water averaged only 18 mg/L and therefore could not support any high amount of denitrification.

In the case of the New Hanover County system, only 12% of the applied nitrogen was removed during the 1997 season (75 gN/m$^2$·yr), implying a net organic carbon supply of only 83 g/m$^2$·yr. In this particular case, wetland nitrate removal performance was clearly limited by the carbon supply, and neither the incoming BOD nor the biomass production of carbon was adequate to realize the full removal potential.

### Oxygen Transfer

In the normal mode of operation for HSSF wetlands, the water is kept below the surface of the granular media, which is typically covered with accumulated plant detritus (and potentially an insulating mulch layer). Air–water exchanges are discussed in Chapter 5; oxygen transfers can be implied for the removal of nitrogen species; Table 9.26, Chapter 9 is an example for HSSF wetlands.

Whereas the rate of oxygen transfer in HSSF wetlands is debatable (because there is insufficient data to settle the issue), there is clearly an upper limit to the rate of transfer; oxygen transfer is obviously influenced by the temperature and physical barriers between the atmosphere and the water column, thus implying a limitation on treatment reactions that require oxygen.

### Other Biogeochemical Cycles

The designer should keep in mind that the biomass cycle spins nitrogen, phosphorus, and micronutrients in addition to the organic carbon examples described previously. All of the cycle constraints discussed in Chapter 17 also apply to SSF wetlands.

Whereas biomass cycling and oxygen are some of the most common performance limitations encountered in HSSF wetland design, other limitations do exist, and the designer must always check the stoichiometry of the desired treatment reactions. A classic example is the treatment of mine drainage waters. If sulfide precipitation of copper and other heavy metals is a desired treatment outcome, there obviously must be enough sulfate (on a mol-to-mol basis) in the

influent waters and also enough organic carbon (often supplementally added to the system as peat or compost) to drive sulfate reduction within the wetland (Frostman, 1993).

## 20.4 VF WETLANDS (INTERMITTENTLY LOADED BEDS)

Vertical flow (VF) nitrification wetlands are vegetated, intermittent sand filters that are sufficiently wet to support obligate wetland plants. Figure 1.7 shows the section through a typical VF system. The bed consists of layers of graded gravel on top of large stones. Reeds are planted in the layer of coarse sand that provides the top surface. The *Phragmites* or other plants growing on the bed provide an improved microbial environment and help maintain the hydraulic conductivity of the bed. Bacteria that do the BOD removal and nitrification are in the biofilm growing on the sand and gravel particles in the bed. In addition, ammonia adsorption can occur during the downflow, followed by nitrification during the aerobic drainage period. The large stones at the bottom of the bed are placed around a drainage system consisting of drainage tiles or perforated agricultural drainage pipes. The wastewater to be treated is dosed intermittently to the surface of the bed.

A number of dosing systems have been used. In the flooding concept, water is dosed onto the bed in a large batch thus flooding the surface. The water then gradually drains down through the bed and is collected by a drainage network at the base. The bed drains completely, and this allows air to refill the bed. The next dose of liquid traps this air, and this, together with the aeration caused by the rapid dosing onto the bed, leads to good oxygen transfer and hence the ability to nitrify. The frequency of dosing should be timed such that the previous dose of liquid has had time to percolate through the bed so that the void spaces are refilled with air. Other concepts involve spray application and creation of anaerobic pools in the bottom of the bed.

Loading is typically intermittent to the active bed at frequencies that vary from project to project. Typically, loading occurs four to six times a day by means of valves and siphons or pumping devices installed in a feed sump or in a buffer tank. Multiple beds are used sequentially in time to allow a resting period. The number of beds ranges from two to six. Figure 20.4 shows an example of a layout used at Medmenham, United Kingdom, consisting of two units of four beds each in series.

### Development

The earliest VF constructed wetlands in Europe were the so-called infiltration fields in the Netherlands and the Seidel System, also known as the Krefeld System or Max Planck Institute Process (Seidel, 1976; Brix, 1994a). These infiltration fields have been in operation in Europe for approximately 25 years, to treat the wastewater from recreation sites outside the municipal sewer network in the Netherlands (Rijs and Veenstra, 1990). The system is typically comprised of a preliminary settling/distribution ditch, four infiltration compartments with soil/sand media, a discharge via drain, and an effluent ditch. The infiltration fields are planted with *Phragmites australis* and are provided with a drainage system 0.55 m beneath ground level. Raw domestic sewage is discharged into the preliminary settling/distribution ditch. After settling, the sewage is mixed with surface water from the effluent ditch and fed into one of the infiltration compartments every three to four days. The other three compartments are kept dry for a period of 10 to 11 days. The purification results and the composition of the groundwater indicate that extensive removal of pollutants takes place in the soil filter.

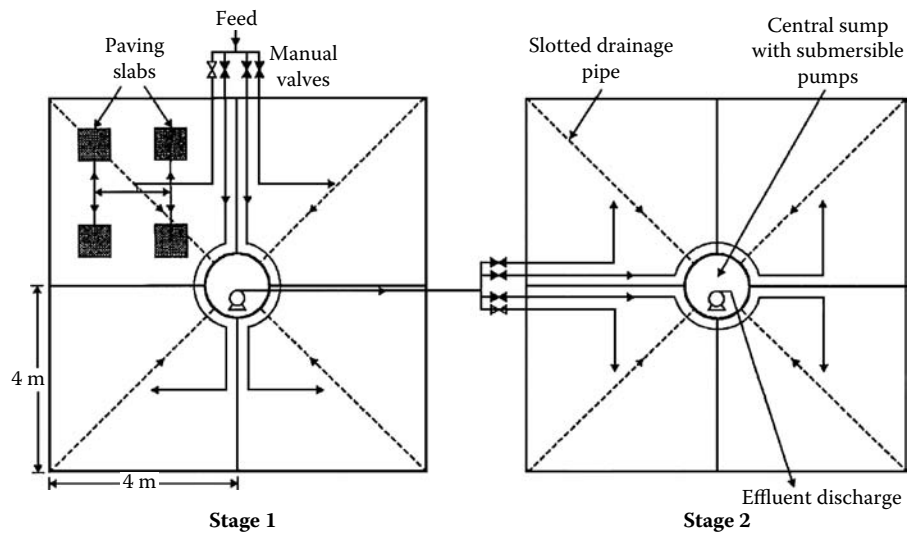



**FIGURE 20.4** Arrangement of the system of VF beds at Medmenham, United Kingdom. This represents a small and intricate set of beds and plumbing. (From Cooper *et al.* (1999) *Water Science and Technology* 40(3): 283–289. Reprinted with permission.)

The Seidel System (Seidel, 1976) consists of several beds laid out in parallel in one or several VF stages in cascades, followed by two or three horizontal flow beds in series. The VF stages are usually planted with *Phragmites australis*, whereas the horizontal flow stages contain a number of other emergent macrophytes, including *Iris, Schoenoplectus, Sparganium, Carex, Typha,* and *Acorus*. The media used in the beds is gravel, topped off with sharp sand in the VF beds to achieve an effective distribution of the wastewater across the bed. The lower, coarser gravel layers are kept aerobic by means of aerated drainage tubes. The VF beds are loaded with pretreated wastewater for one to two days, and are then allowed to dry out for four to eight days. The thin crust of solids that forms on top of the sand layer is mineralized during the rest period and achieves an equilibrium thickness. Variations of these systems have been in operation in different European countries for about 30 years (Burka and Lawrence, 1990; Liénard *et al.*, 1990). Austrian experiences have been very good during the last few years (Schönerklee *et al.*, 1997; Laber *et al.*, 1997).

The development of VF wetlands in the United States has followed more recently than in Europe, in the early 1990s. Watson and Danzig (1993) performed pilot mesocosm experiments that confirmed the European experience and subsequently retrofitted a lagoon-wetland system with a VF bed prior to the wetland. In 1993, the Lamb Weston company began pilot studies of a *Phragmites* VF system at Connell, Washington (Kadlec *et al.*, 1997). This study led to a full-scale facility in which 5,000 $m^3$/d are applied to 4 ha of VF beds, which have operated successfully since 1996 (Burgoon *et al.*, 1999). Other VF facilities have been developed at a variety of scales.

## Sizing

Usually, VF constructed wetlands are applied as biological stages (secondary treatment) for mechanically pretreated domestic wastewater. Experiences have shown that their performance is strongly dependent on pretreatment, i.e., settlement tank, septic tank, package plant, or others. VF constructed wetlands can also be applied as a polishing stage following a conventional biological treatment plant, such as activated sludge. The typical design parameters of VF constructed wetlands are the required area, the depth, the medium, and the application strategy.

### Scaling Factors

To date, scaling factors have been the most widely used sizing method for VF wetland systems. Table 15.1, Chapter 15, summarizes various scaling factors that have been proposed in the literature.

### First-Order Models

The first-order areal model may be used for the calculation of the wetland area as has been detailed for the FWS and HSSF systems. The *k*-values depend on the water-quality parameter and on the different environmental and operational circumstances. The ranges reported in Table 20.9 were deduced from measurements of an unspecified number of practically operated plants (Vymazal, 1998). These rate-coefficient ranges are plug flow values, and therefore are based upon the *P*TIS model with a very large value of *P*. The tracer measurements do not support this presumption, which is surprising, because the vertical unsaturated flows probably do not back-mix and are not subject to effects of wind as are FWS systems. In any case, the wide range of the *k*-values is also a result of the different designs and operations of the systems. It must be further recognized that the internal loading of ammonia, resulting from mineralization of organic nitrogen, has been ignored, and the rate coefficients are, therefore, only apparent values. They do not represent nitrification, which may indeed be much higher. Also, because of the periodic character of the process, the use of such a simple first-order model, as for other types of constructed wetlands, may be inappropriate (Platzer, 1999). It is possible to modify the application of the first-order model to a more complex form involving the dynamics of the filling and draining cycle (Sun *et al.*, 1998). To date, there are no calibrations of such tidal models to domestic sewage treatment systems.

**TABLE 20.9**
**Rate Coefficients for Vertical Flow Pulse-Loaded Wetlands Compared to Those for HSSF Systems; the VF Values Are for Plug Flow**

| Parameter | VF *k*-rate (m/yr) | HSSF Input (mg/L) | HSSF 20th Percentile *k*-rate (m/yr) | HSSF 80th Percentile *k*-rate (m/yr) | HSSF Table Number | HSSF *P*TIS |
|---|---|---|---|---|---|---|
| $BOD_5$ | 20–60 | 30–100 | 20 | 82 | 8.7 | 3 |
| $BOD_5$ | — | 100–200 | 12 | 62 | 8.7 | 3 |
| $NH_4$-N | 10–40 | — | 2.7 | 29.8 | 9.25 | 6 |
| TN | 12–20 | — | 3.3 | 18.1 | 9.19 | 6 |
| FC | 70–95 | — | 43 | 251 | 12.17 | 6 |

*Source:* For VF data, Vymazal *et al.* (1998) In *Constructed Wetlands for Wastewater Treatment in Europe*. Vymazal *et al.* (Eds.), Backhuys Publishers, Leiden, The Netherlands, pp. 95–121.

### *Biochemical Oxygen Demand*

The BOD rate coefficients for the VF systems in this book are given in Table 8.16, Chapter 8, separated by inlet concentration ranges. These differ widely from those for HSSF, and are considerably higher for VF than for HSSF (Table 8.8, with lower values if plug flow is assumed), by a factor of three to five. The current data set produces about the same *k*-values as those shown in the work of Vymazal *et al.* (1998), under the plug flow assumption (not shown). But Schwager and Boller (1997) conducted tracer tests that indicated *N*TIS = 5 for their VF system (see Chapter 2). When the BOD weathering effect is factored in, it seems probable that a value such as *P*TIS = 2 should apply to VF systems. Under this assumption, the VF BOD *k*-values are much higher.

### *Ammonia*

There seems to be little difference in the ranges of the ammonia reduction rate coefficients for HSSF and the Vymazal et al. (1998) VF wetlands (Table 20.9). Although this seems in conflict with the concept of better nitrification and BOD reduction in VF systems, it is not. Rather, it points to the need to consider hydraulic loadings and the supply constraints that limit the reactions that may occur, as detailed in constraint checks advocated in earlier chapters. The ranges of apparent plug-flow ammonia *k*-values, derived from the database of 72 systems used in this book, are similarly wide (Table 20.10). However, the ammonia rate coefficients are three to five times as high as those for HSSF wetlands (compare Tables 20.10 and 9.25).

## Loading Rates

There are two philosophies of hydraulic loading for pulsed VF wetlands: loadings comparable to HSSF wetlands, and loadings many times higher. Peaks in the reported frequency distribution occur at 4 cm/d and at 40 cm/d. Higher rate constants are associated with higher hydraulic loadings. The Brix *et al.* (2002a) design hydraulic loading is on the order of 20 cm/d. Hydraulic loadings may be fairly high and still yield good BOD/COD and ammonia reduction. Loadings ranging to over 100 cm/d have been employed.

The definition of hydraulic loading rate used here is the long-term average flow rate to the system divided by the total area of beds. Other definitions can be used, and the reader is cautioned to understand the terminology in use for a particular project. For example, beds may be loaded only one third of the time, thus tripling the per-bed loading when in use.

**TABLE 20.10**
**Apparent Rate Coefficients for Ammonia for VF Pulse-Loaded Wetlands**

**Stipulations**

1. Period of record averages are used in calculations.
2. For *k*-value calculations, the following *P*-*k*-*C** parameters are selected:
   a. $C^* = 0$ mg/L.
   b. $P = 8$ TIS.
3. Ranges of variables:

| | HLR (cm/d) | $NH_4$-N In (mg/L) | $NH_4$-N Out (mg/L) |
|---|---|---|---|
| **Mean** | 22.7 | 59 | 21.1 |
| **Median** | 4.3 | 49 | 4.9 |
| **Max** | 222 | 264 | 212 |
| **Min** | 1.00 | 0.20 | 0.03 |

**Results: N·*t* = 120 wetland-years; N = 72 VF wetlands.**

| Percentile | HLR (cm/d) | Apparent Implied Oxygen Utilization ($g/m^2 \cdot d$) | Apparent Rate Constant (m/yr) |
|---|---|---|---|
| 0.05 | 1.9 | 3.1 | 0 |
| 0.10 | 2.2 | 4.2 | 5 |
| 0.20 | 2.4 | 9.4 | 10 |
| 0.30 | 3.1 | 12.8 | 17 |
| 0.40 | 3.5 | 15.1 | 24 |
| 0.50 | 4.3 | 17.7 | 33 |
| 0.60 | 7.6 | 21.0 | 48 |
| 0.70 | 11.5 | 30.6 | 68 |

*Note:* The apparent implied oxygen utilization is calculated as $IOTR = \Delta BOD + 4.6 \cdot \Delta NH_4N$. Both numbers are apparent because they do not include internal ammonia production from organic nitrogen.

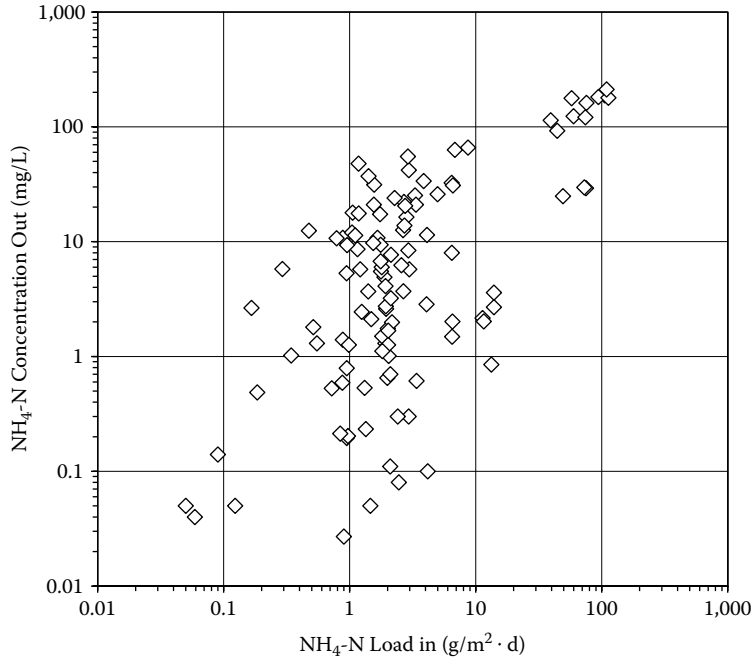



**FIGURE 20.5** Ammonia load response for pulsed VF wetlands. (N = 118 system-years of data.)

Further, pulse loading means that water flows to one of the three beds only a fraction of the time it is in service, meaning that there is an instantaneous, per-bed loading as well.

However, performance (i.e., lower outlet concentrations) increases as loading decreases (Figure 20.5). Thus, increasing bed size, or using the beds in series, is an effective design technique. However, the scatter on the loading chart is too great to be of use in design.

It is likely that the high *k*-values for BOD and ammonia will cause rapid reactions and that this will cause the rate-limiting factor to be the oxygen transfer rate. Therefore, this constraint must be considered in design and can in fact be its basis.

## Oxygen Transfer Rate

A second design approach is to rely on empirical observations of the apparent or implied oxygen transfer rates (IOTR) from operating systems. As discussed in Chapter 9, this procedure rests on some assumptions regarding the stoichiometry of oxygen usage. Here, we shall presume that the maximum IOTR applies to VF wetlands because of the inherently oxic environment that is intentionally created.

Platzer (1999) has measured values of oxygen transfer rate (OTR) in the range 23–64 $gO_2/m^2{\cdot}d$, Green *et al.* (1997a) report 56–60 $gO_2/m^2{\cdot}d$, and Cooper *et al.* (1999) report 50–90 $gO_2/m^2{\cdot}d$. Oxygen consumption is for nitrification and BOD/COD reduction and is estimated as

$$\text{IOTR} = 4.6 \times \Delta(\text{NH}_4\text{–N}) + \Delta(\text{BOD}) \qquad [\text{in gO/m}^2{\cdot}\text{d}] \tag{20.10}$$

This oxygen demand may be lessened by BOD settlement, ammonia volatilization, plant uptake, and denitrification, but increased to support microorganic metabolism. For instance, Cooper (1999) uses a coefficient of 4.3 instead of 4.6. The observed ranges of oxygen supply per unit area may be combined with the estimate of oxygen demand of the entire flow to determine the bed area: area = OD/IOTR = (gO/d)/(gO/m$^2$·d).

The areas so determined are relatively small. For instance, suppose the incoming flow is 1,000 m$^3$/d, $\Delta(NH_4\text{-N})$ = 20 mg/L, and $\Delta$(BOD) = 40 mg/L. Then, the oxygen demand is calculated to be 132 kgO/d. If the IOTR is taken to be 30 gO/m$^2$·d, then 4,400 m$^2$ is required. Brix *et al.* (2002a) calculate VF systems by presuming a surface aeration rate of 30 gO/m$^2$·d, and adding a 25% safety area. If the bed is 1 m deep and intermittently loaded, a nitrification efficiency of 85% is projected.

Table 9.28 in Chapter 9 is preferred for purposes of design over Table 20.10. Although Table 9.28 is based on less data, it has the advantage of including the internal ammonia loading that results from the ammonification of organic nitrogen. As is the case in other sizing calculations, it is left to the designer to select IOTR to be used in a particular design.

## Denitrification

The oxidized nitrogen that is formed by nitrification may be partially denitrified in the bed under some circumstances. There must be an anoxic zone, and there must be a carbon supply. Thus, the systems that remove BOD effectively do not denitrify effectively. Examples are given in Table 20.11.

**TABLE 20.11**
**Examples of Ammonia-N and Nitrate-N Removal/Buildup in VF Constructed Wetlands**

| Location | $NH_4$-N In (mg/L) | $NH_4$-N Out (mg/L) | $NO_3$-N In (mg/L) | $NO_3$-N Out (mg/L) |
|---|---|---|---|---|
| Dhulikhel, Nepal[a] | 27 | 0.08 | 0.4 | 27 |
| Wapserveen, Netherlands[b] | 61 | 0.23 | 0.04 | 44 |
| Lemmer, Netherlands[b] | 39 | 1 | 0.07 | 29 |
| Contrada Petrosa, Italy[c] | 97 | 0.1 | 0.1 | 80 |
| Zeutschbach, Austria[d] | 76 | 12.6 | 0.7 | 29 |
| Lappach, Austria[d] | 120 | 25 | 2.5 | 61 |

[a] Data from Laber *et al*. (2003b) *Secondary treatment of hospital wastewater with a 2-stage constructed wetland system.* Haberl (Ed.), Book of abstracts from the Conference on Achievements and Prospects of Phytoremediation in Europe, 15–18 October 2003, Vienna, Austria.
[b] Data courtesy ECOFYT/F. van Dien.
[c] Data courtesy F. Masi.
[d] Data courtesy G. Mitterer-Reichmann.

### Temperature

Nitrogen removal processes are moderately temperature sensitive. Temperature coefficients from field data in VF wetlands are in general not available and can only be presumed. Nitrification does not entirely shut down at cold temperatures. Further, VF wetlands display the ability to conduct BOD reduction simultaneously with nitrification. It is therefore not necessary to reduce BOD prior to nitrification, but there is probably an advantage in doing so.

### Phosphorus

Phosphorus is not much reduced in VF wetlands, except perhaps for a transitory period of start-up. Sorption sites become loaded, and efficiency declines. Because loadings are sometimes quite high, the removal via the biogeochemical cycle is minimal.

### Variability

Like other types of constructed wetlands, VF systems possess inherent variability in their effluent concentrations. Table 9.36, Chapter 9, provides examples that indicate that the excursion frequencies are not widely different from those for other wetlands. There are annual cosine trends and scatter about them (see Figure 9.50, Chapter 9).

### Number of Beds

The need for bed resting, coupled with the fraction of the long-term period that can be devoted to resting, determines the number of separate VF beds needed. For instance, if it is decided that beds must rest two thirds of the time to avoid clogging layer buildup, then three beds are needed, to be used in rotation one at a time. Note that the Medmenham, United Kingdom, project used four beds (Figure 20.4).

## 20.5 VF WETLAND SIZING (RECIRCULATING FILTERS)

Vegetated recirculating gravel filters have been used as a "vertical flow wetland" alternative in North America, mainly in cold-climate applications, where surface flooding or spray irrigation creates operational problems in the winter. Because surface flooding is a much simpler means of influent distribution, it is preferred where thermal considerations allow. Because these systems operate as unsaturated flow filters, the vegetation used often falls into the facultative category (see Chapter 2), so these systems can only nominally be classified as "wetlands" from the standpoint of the vegetative community, which does not play a significant role in the treatment process. At the present time, there are approximately 35 systems of this nature operating in Canada, and about 140 systems operating in the United States. The application of these systems ranges from single-family homes (Wallace and Higgins, 2001) to dilute runoff streams in excess of 1,500 $m^3$/d.

These systems operate as gravel filters, which is an old treatment technology dating back to 1889 (Grady *et al*., 1999). The degree of treatment depends on the influent mass loading, recirculation ratio, media type, and air exchange (oxygen transfer). At the present time, designs are typically based on empirical methods (Grady *et al*., 1999) presented in standard sanitary engineering textbooks.

The most common design approach is to size these systems as recirculating gravel filters and ignore the vegetation component from a treatment perspective. The overall footprint of the system is usually based on a prescriptive mass

loading scaling factor. BOD loading rates of 10 to 40 g/m$^2$·d are often used, with most designs using loading rates of <25 g/m$^2$·d (Crites and Tchobanoglous, 1998).

The hydraulic loading and recirculation must also be considered because the goal of these systems is not only to provide frequent dosing (to keep the bed media moist) but also to minimize the dose volume to promote unsaturated flow and facilitate air exchange (and oxygen tran sfer) within the wetland bed. Doses of 48–120 doses per day are common, which typically require storage tanks in the recycle loop with the capacity to retain the average daily (forward) flow (Crites and Tchobanoglous, 1998).

This technology can reliably achieve secondary treatment for BOD that has received primary treatment. Nitrification and denitrification are possible but depend on the influent mass loading rate and degree of recycle; current design tools are empirical in nature (NRC, 1946; Galler and Gotas, 1964).

## 20.6 BIOSOLIDS WETLANDS FOR SLUDGE DEWATERING

At present, information on the sizing of biosolids wetlands is based on mass loading scaling factors. These have been developed based on the cool temperate climate of Denmark. Scaling factors are <60 kg/m$^2$/yr for waste activated sludge (WAS) and <50 kg/m$^2$·yr for a combination of primary sludge and WAS (Nielsen, 2002).

This system utilizes the *ET* properties of *Phragmites* as an aid in sludge dewatering and stabilization, coupled with gravity drainage and microbial stabilization. As a result, the system is highly dependent on the loading and resting of individual beds. The overall wetland area is subdivided into beds to facilitate this mode of operation. A minimum of 8 beds are required for facilities that produce up to 500 tons of dry solids per year; ten beds are the minimum for 500–1,000 dry tons per year, and 12–14 beds are needed for >1,000 dry tons per year. (Nielsen, 2002).

The typical loading intervals last four to ten days, the remainder of the operational sequence being devoted to resting the individual bed. Shorter loading periods and longer resting intervals are the preferred operational strategy, making the number of beds an important design consideration.

## 20.7 SECONDARY CONSIDERATIONS

It is assumed that at this stage the designer has implemented an appropriate sizing method (loading charts or the *P-k-C** model) and cross-checked performance forecasts against existing performance data and biogeochemical constraints. The final stage of the sizing process is to consider other factors that may influence the overall wetland sizing. The "definitive list" of final sizing parameters is inherently project specific; here, we will consider two commonly encountered situations that may influence the final design of a HSSF wetland.

### Seasonal Impacts of the Biomass Cycle

As discussed previously in this Chapter, the annual biomass cycle can impose constraints on wetland performance. However, short-term seasonal effects of plant biomass can further influence system performance (see Figures 9.11–9.13, Chapter 9). If we take the case of a typical European HSSF wetland sized at 5 m$^2$/PE (ATV, 1998), the equivalent nitrogen loading is 2.4 g/m$^2$·d (73 g/m$^2$·month). Inspection of the biomass cycle (e.g., Figure 9.13) indicates that all of the influent nitrogen can be assimilated into plant biomass during the period of maximum growth (June and July in the northern Hemisphere). These short-term effects can be important when the system has a frequent regulatory compliance interval (e.g., weekly or monthly).

### Thermal Energy Balance Limitations

The preceding design examples have considered wetland sizing independent of thermal considerations; thus the inherent assumption of these examples is that "bigger is better," e.g., more wetland area equals a desirable outcome in terms of better treatment performance. There are obvious limitations to this assumption, which are most likely to be encountered in extreme environments; the two most common are the following:

In water-scarce environments, water quantity may be more important than water quality; hence, the effluent is too valuable to waste and is required for reuse. In these situations, a wetland designer may choose to minimize the wetland area (and associated treatment performance) to maximize the effluent volume available for reuse. In these situations, utilization of the sequential, tank-by-tank, variable flow model (see Tables 20.3, 20.6, 20.8) can greatly facilitate the design compromise between treatment efficiency and the volume of water available for reuse.

## SUMMARY

This chapter has integrated the elements of preceding chapters into a sequential approach to designing and evaluating SSF treatment wetlands. Sizing is the first step in design, and loading charts, scaling factors, and the *P-k-C** model have all been presented for comparable design examples.

The designer is cautioned that there is nothing about loading charts, scaling factors, or the *P-k-C** model that acknowledges the limitations of wetland biogeochemical cycling. Therefore, this chapter also discusses the cycling of biomass cycling and other constraints of the wetland ecosystem to ensure there are no unrealistic expectations of the nutrient, oxygen, and carbon supplies of the ecosystem. Rate coefficients must be adjusted if the presumed capacities of the ecosystem are being exceeded. Loading charts are a method of understanding where a particular design is positioned compared to the behavior of existing systems. Designs that fall outside the range of the existing data envelope will require further justification (such as pilot-scale studies) to ensure system performance.

# 21 Implementation of SSF Wetlands

The purpose of this chapter is to provide information that will help ensure successful outcomes for projects using horizontal and vertical subsurface flow (SSF) wetlands (or biosolids wetlands) by discussing the internal configuration of these systems. Common construction pitfalls and start-up errors are also discussed.

Implementation of SSF wetland systems consists of the following three stages:

1. Physical design (specification of internal components)
2. Construction of the wetland
3. Start-up (commissioning)

The sizing of the system can be done using different sizing tools, including loading charts, scaling factors, and first-order modeling. The pros and cons of these methods have been discussed in Chapter 20.

The first implementation stage, physical design, involves making decisions as to the number of cells, site grading, aspect ratio, bed depth, internal piping, media size, hydraulics, water level control, insulation, etc. The configuration of these components is then typically documented in a set of technical drawings (engineering plans) and written specifications.

Construction of the system consists of preparing bidding documents, retaining a contractor, and physically building the wetland. This is the most critical phase of the implementation because construction of SSF wetlands is essentially a "one-way street." Problems with site grading and elevations become difficult and expensive to correct once the liner (if any) is in place. Problems with the liner become next to impossible to correct once the bed media is installed. Because it is so difficult to undo construction problems, the wetland designer is often retained in some advisory capacity to observe and document the construction work.

The final stage, commissioning, includes introducing the flow and pollutant loading into the wetland, as well as managing water levels in the wetland in a manner consistent with vegetation establishment and regulatory compliance.

The overall implementation process may seem deceptively simple, but it is not. Determining the wetland size is not the same as designing the system. Mistakes made in the technical drawings and specifications will lead to serious problems down the road. Mistakes made during construction may be extremely difficult to fix, and repairs are likely to cost many times more than the original components. Failure to understand how the wetland is supposed to operate leads to start-up problems. Any of these situations is likely to lead to the intervention of regulatory authorities.

Characterization of the influent waste stream and general site assessment issues have been previously discussed in Chapter 16. Procedures for sizing the wetland (determining the necessary wetted area) have been reviewed in Chapter 20. The next stage in the implementation process is deciding how the wetland should be configured, how internal components should be placed, and how large they are. Those issues are discussed in more detail here.

## 21.1 SITING

General conditions of the potential site will have been considered during the establishment of the basis of design. There will remain a number of possibilities for the location of the system within the overall site confines. These factors have been previously discussed in a general way in Chapters 16 and 18 for FWS wetlands; these also apply to SSF wetlands. This chapter considers additional factors that often come into play when siting SSF wetlands. These systems are typically smaller than FWS wetlands and, consequently, are often constructed in or near existing infrastructure.

### Construction Access

Although the as-constructed footprint area of an SSF wetland may be small, the designer must keep in mind that these wetland cells will be constructed with earth-moving equipment such as backhoe excavators. The size of this equipment requires a clear perimeter area around each wetland cell; the track size of the equipment will define the minimum separation distance between the wetland cell, the site boundary, and additional wetland cells. Large excavators are often preferred by contractors because the long reach of these machines facilitates placement of bed media during construction. An example of a construction-imposed separation distance is shown in Figure 21.1.

Typical separation distances are in the range of 5–7 m around the perimeter of each wetland cell to allow access by construction equipment. Designers should keep in mind that devices such as cranes (Figure 21.2) may be required to set primary treatment devices such as settling tanks, so overhead vertical clearance (from obstacles such as power lines) may be another construction access issue. Excavations for tanks, pipes, wetland basins, and water level control structures raise another access issue. Clearly, there must be an adequate location on the site to temporarily store excavated soil. With deep excavations (especially trenches), the excavation may have to be braced, or the sidewalls may have to be stepped back to avoid the risk of cave-in. This greatly increases the amount of site area and access needed during



**FIGURE 21.1** Separation distance between these two vertical flow wetland cells was dictated by the need for access by construction equipment at The Preserve, Minnesota. The use of such a large excavator was desirable to place gravel media in the wetland cells. Alternate means of gravel placement, such as an extendable conveyor, would decrease this separation distance.

the construction period. This is especially important in sandy soils that have a low angle of repose and thus require large excavation widths.

It should be noted that heavily wooded sites will require clearing and grubbing, as discussed in Chapter 18, and remote sites may necessitate the construction of access roads, adding to the overall construction cost.

### Slopes

SSF wetlands require the construction of level beds. On sloping sites, this will necessitate terracing of the areas where wetland beds are to be constructed. On steeply sloping sites, this will require substantial cut (earth removal) on the uphill side and substantial fill (earth placement) on the downhill side; this additional earthwork can far exceed the volume of the wetland basin itself. Consequently, steeply sloping sites are often not preferred construction areas, unless no other alternative is available.

It should be noted that completely flat sites may also require substantial earthworks if gravity flow through the treatment process is desired. Sites with 1–3% slopes are often the easiest to work with in terms of establishing a hydraulic profile that flows by gravity.

### Existing Utilities

SSF wetlands are often used for homes and villages that have previously lacked wastewater treatment. Because the wetland (and any associated collection system component) is often the last utility in the ground, there is always the risk of hitting and damaging other underground utilities, such as water pipes, gas lines, electrical power lines, or telephone wires. Most utilities in North America have a "one-call" telephone number to dispatch an individual to locate these utilities. Obviously, sites with known utility conflicts should be avoided, or provisions to reroute the affected utilities must be incorporated into the design process.



**FIGURE 21.2** Installation of settling tanks at The Preserve, Minnesota. Note that construction access is needed for four different components: (1) the truck and trailer used to deliver the tanks, (2) the crane used to set the tanks, (3) the excavation itself, and (4) the area needed to stockpile the excavated soil. Thus, the total area needed for construction is far greater than the tanks themselves.

Biosolids wetlands are generally constructed to support sludge management at existing mechanical treatment plants and are often constructed in close proximity to these treatment works. Again, knowledge of existing utilities is a critical component of the design process.

### Floodplains

Siting of wetland systems in floodplains and floodways has already been addressed in Chapter 18. It should be noted that most flood events result in considerable sediment deposition as flood waters recede. Free water surface (FWS) wetlands are much more tolerant of this sediment load than SSF wetlands, where the additional sediment could create potential clogging issues in the wetland bed. As a result, locating SSF wetlands in floodplains requires careful consideration of the expected frequency and severity of flooding in the project area.

Biosolids wetlands will contain organic sludge with varying degrees of pathogens, depending on the age of the biosolids and level of stabilization. Because the potential to export pathogenic material exists during a flood event, a decision to locate such a facility in a floodplain would have to be carefully evaluated.

### Regulatory Issues

Regulatory requirements are often a major factor in the selection of SSF technology over FWS wetlands. For instance, in the United States, approximately 25% of homes rely on onsite (single home) septic systems (U.S. Census Bureau, 1990). There are considerable health risks associated with exposure to raw sewage (or sewage exiting the septic tank). As a result, there is a large body of prescriptive codes that require the treatment and disposal of the sewage *without surface exposure of the wastewater* (U.S. EPA, 2002c). Thus, wetlands such as FWS and surface-flood vertical flow (VF) systems cannot be used in these applications, and the regulations force the designer to use HSSF wetlands or VF wetlands with buried distribution piping so the water will not be exposed.

Another example of regulatory requirements that dictate the wetland selection process is airports. In cold climates, ice must be removed from the airplanes prior to take-off, and the associated de-icing chemicals generate stormwater runoff that is high in organic (glycol) contaminants. Constructed wetlands can remove this chemical oxygen demand (COD) load. However, because there is a potential for bird–aircraft strike hazard (BASH), open water bodies such as ponds and FWS wetlands are often discouraged or prohibited from being constructed near airport runways (FAA and U.S. Department of Transportation, 2004). For this reason, SSF wetlands (VF or HSSF) are preferred for treating de-icing runoff.

Siting of wetlands is also influenced by required separation distances from homes, water bodies, and other landscape features. These separation distances are usually prescriptive in nature and dictated by local, state, and provincial regulations. In the United States, these range from approximately 3 m to over 300 m, depending on the applicable regulations and the size of the treatment system. In situations where there are limited choices for siting the wetland, these separation distances may end up not only dictating the location but also the shape of the wetland, as illustrated in Figure 21.3.

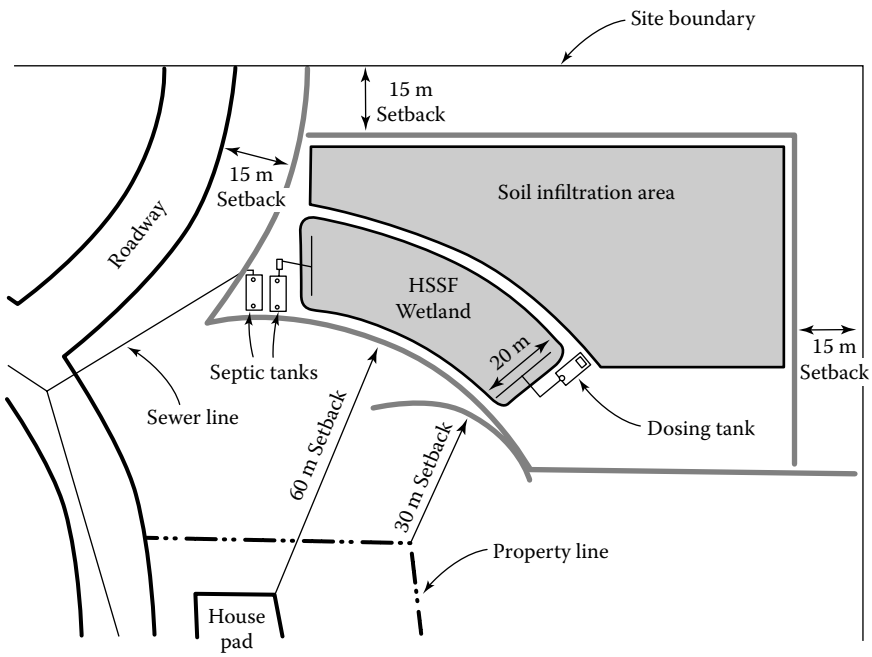



**FIGURE 21.3** The shape and location of this HSSF wetland bed in Lake Elmo, Minnesota, were dictated by regulatory setback requirements.

## 21.2 LAYOUT AND CONFIGURATION

Once the wetland area and location have been selected, the designer must then make a number of decisions regarding the layout and configuration of the wetland. These include

- The number of parallel flow paths
- The number and type of wetlands in each flow path
- Clogging dynamics
- Cell configuration (aspect ratio and bed depth)
- The hydraulic profile of the wetland system
- The internal configuration of the wetland cells (flow distribution, bed depth, media size and type, the need for insulating mulch, the water level control structures, the liner selection and type, etc.)

### Number of Flow Paths

The decision to design the wetland system with two or more parallel flow paths depends on several operational factors:

- The need to provide operational flexibility
- The need to provide treatment redundancy
- The need to operate the system on a load-and-rest basis

### Operational Flexibility

When there are multiple flow paths, one treatment train can be taken out of service for maintenance and the flow diverted to the other parallel wetland cells. This flexibility is a desirable attribute in a wetland treatment system, but it comes at the added cost associated with extra berms, liner, water level control structures, and earthwork. These added costs become especially significant for very small treatment systems. Let us consider the example of a single home horizontal subsurface flow (HSSF) wetland sized 300 $m^2$, where a 5-m perimeter is needed for construction access. The overall site area needed is approximately 35 m × 22 m, or 770 $m^2$. The results of splitting this wetland into two parallel beds is 40 m × 22 m, or 880 $m^2$. In the first case, the wetland occupies 39% of the required site area; with the two parallel beds, the wetland occupies 34% of the site area, an influent splitter structure is necessary, and the outlet control structure(s) must regulate two water levels.

Although the split-bed approach illustrated offers more operational flexibility, it comes at the cost of additional area, liner, and control structures. The added costs of these components must be weighed against the alternate methods to provide operating flexibility. For instance, many single-home septic tanks provide approximately three days of hydraulic retention time, so one means of taking the system out of service for maintenance or repairs would be to simply pump the septic tank. This would give the operator about three days to complete the necessary work, and if additional time is needed, the tank could be pumped again. This alternate approach to flexibility is less expensive than the split-bed system, and for this reason, single-home HSSF wetlands in the United States are commonly single-bed designs.

As the size of the system increases, the added cost of parallel flow paths becomes a progressively smaller component of the overall construction cost, and providing alternate means of flexibility (such as pumping of septic tanks) becomes progressively more expensive. Therefore, for larger projects, two or more parallel flow paths become the preferred design approach. For small community wastewater projects, systems with a design flow greater than 35 $m^3/d$ would typically have two parallel flow paths, and for larger projects, additional parallel flow paths may be warranted. For instance, the HSSF wetland used to treat aircraft de-icing runoff in Edmonton, Alberta, consists of six parallel flow paths, with two cells in each flow path (Figure 21.4).

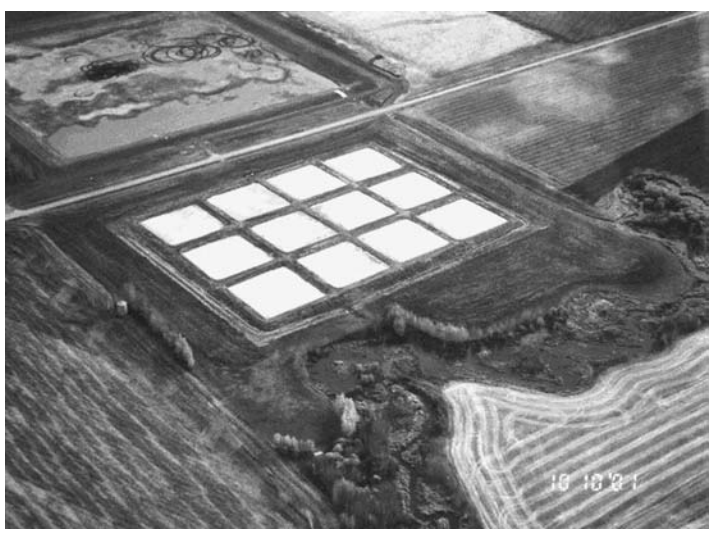


**FIGURE 21.4** HSSF wetland for treatment of aircraft de-icing runoff, Edmonton, Alberta. This particular system consists of six parallel flow paths, with two treatment cells in each flow path.

### Treatment Redundancy

Many regulatory authorities require wastewater treatment systems to be designed to treat more than the stated design flow and load. For treatment systems in the upper midwestern United States, a commonly used redundancy requirement is that the treatment system must be able to treat 75% of the influent flow and load, with one treatment train (flow path) out of service (Great Lakes UMRB, 1997). It should be noted that this redundancy requirement was developed for mechanical wastewater treatment plants treating raw municipal wastewater and may or may not be appropriate for treatment wetlands, depending on the type and intended function of the wetland.

With a redundancy requirement, two parallel flow paths are the minimum. If the designer selects two flow paths, each wetland flow path must be designed to treat 75% of the flow and load, and the wetland must be designed to treat 150% of the design ($2 \times 75\%$). In certain cases, adding a third flow path may be advantageous. With three flow paths, two treatment trains will be in operation when one is out of service, so the two trains in service must each be able to treat 37.5% (one half of 75%) of the design flow and load. So, the overall wetland is sized to treat 112.5% of the design ($3 \times 37.5\%$). The designer must then evaluate whether the savings in wetland area is sufficiently justified, given the added cost of the extra berms and control structures.

### Loading and Resting

Some treatment wetlands require a loading and resting regime to function properly. For these types of systems, the loading and resting sequence will determine the number of the parallel flow paths required.

Most VF wetlands are pulse-loaded at a rapid rate to flood the surface of the bed. In between pulses, water drains from the wetland and air is drawn into the wetland bed (see Chapter 2). This air movement reoxygenates the microbial biofilms and maintains aerobic conditions within the wetland bed. Particulate organic matter is deposited on the surface of the bed, and microbial populations are greatest in the upper regions of the bed, where the organic loading is the highest (Langergraber *et al.*, 2006b) As the loading period progresses, this leads to the development of a clogging mat on the surface of the bed (unless very low organic loadings are utilized). This clogging mat blocks the movement of air into the bed, promoting anaerobic conditions and ponding of the bed surface (Platzer and Mauch, 1997; Hyánková *et al.*, 2006).

Loading and resting are used as means to avoid clogging of the bed and associated surface ponding (EC/EWPCA Emergent Hydrophyte Treatment Systems Expert Contact Group and Water Research Centre, 1990; Platzer and Mauch, 1997; Molle *et al.*, 2004a). The need for resting intervals becomes increasingly important as the organic loading on the VF wetland is increased. For instance, in France many VF wetlands are fed with raw sewage (no primary treatment); to operate successfully, three VF beds in parallel are needed in the first stage (Molle *et al.*, 2004a). One bed is fed with raw sewage while the other two are rested (Figure 7.31). Thus, the loading and resting operating regime of these French systems dictates that the first stage of the treatment process must consist of at least three parallel flow paths.

## 21.3 NUMBER AND TYPE OF WETLANDS IN EACH FLOW PATH

Once the number of parallel flow paths has been decided, the next question facing the wetland designer is how many cells to put in each flow path. There are several scenarios that would lead the designer to consider multiple cells in series:

- There is a need for greater treatment efficiency (and hence greater hydraulic efficiency) to increase the number of *P*TIS through the use of two or more cells in series.
- There is sufficient slope to the site that breaking the flow path into multiple cells is advantageous from a civil engineering and an earthwork perspective.
- There is a desire to use more than one wetland type in the overall treatment process (e.g., vertical flow to horizontal flow, or vice-versa).
- There is a desire for staged treatment, with different treatment reactions occurring in different wetland cells.

### Greater Treatment Efficiency

The first case is a situation where the designer wishes to increase the overall treatment efficiency of the process—or to provide the same level of treatment in a smaller area. By combining two wetland cells in series, the number of tanks in series (*N*TIS) is theoretically increased. The decreasing length-to-width ratio for each physical compartment means that *N*TIS is not doubled. The relaxed parameter, *P,* accounts for both hydraulic effects (*N*) and effects of pollutant weathering; so a decision to increase *P* must be based on the pollutant under consideration and the best professional judgment of the designer. In any case, the advantage of cells in series for HSSF wetlands is not large unless the design contemplates a close approach to *C**.

As discussed in Chapter 6, compartmentalization becomes a significant aspect of the wetland sizing process when the goal is a very high level of treatment. The case of providing two wetland cells in series (instead of one cell) was previously explored for biochemical oxygen demand (BOD) removal in Tables 20.7 and 20.8. In that particular example, there was not a dramatic reduction in the required area because the design goal (94.4% reduction in mass load) allowed an effluent concentration significantly above (double) the assumed *C** value.

However, as the target effluent quality gets closer and closer to *C**, the hydraulic efficiency *N* (and hence *P*) becomes an extremely important part of the sizing process, because bypassing of untreated effluent in the TIS model has

**TABLE 21.1**
**COD Reduction for One Wetland Cell, *P*TIS = 2**

| Input Parameters | | Calculated Values | |
|---|---|---|---|
| Flow rate, $Q$ | 137.1 $m^3/d$ | Volume per tank | 547.2 $m^3$ |
| Precipitation, $P$ | 2.2 mm/d | Area per tank | 1,200 $m^2$ |
| $ET$ | 0.13 mm/d | Influent flow, $Q_i$ | 137.1 $m^3/d$ |
| Infiltration | 0.01 mm/d | Effluent flow, $Q_o$ | 142.0 $m^3/d$ |
| *P*TIS (system) | 2 | Average flow, $Q_{avg}$ | 139.6 $m^3/d$ |
| Area, $A$ | 2,400$m^2$ | Influent mass load | 16,452 g/d |
| Porosity, ε | 0.38 | Effluent mass load | 1,591 g/d |
| Bed depth | 1.2 m | Mass percentage reduction | 90.3% |
| $C_i$ | 120 mg/L | Nominal HLR | 0.057 m/d |
| $C^*$ | 10 mg/L | HLR based on $Q_{avg}$ | 0.058 m/d |
| $k$ | 356 m/yr | HLR based on *P*TIS | 0.059 m/d |
| $k$ | 0.975 m/d | Nominal HRT | 7.98 d |
| Compliance $C$ | 20 mg/L | HRT based on $Q_{avg}$ | 7.84 d |
| Variability factor | 1.78 | HRT based on *P*TIS | 7.70 d |
| Target $C$ | 11.2 mg/L | Liner required | 3,294 $m^2$ |
| | | Piping required | 1,200 m |

| Calculated Values | | System In | Exit Tank 1 | Exit Tank 2 | System Out |
|---|---|---|---|---|---|
| Net flow | $m^3/d$ | 137.1 | 139.6 | 142.0 | 142.0 |
| Precipitation | $m^3/d$ | 5.280 | 2.640 | 2.640 | — |
| $ET$ | $m^3/d$ | 0.312 | 0.156 | 0.156 | — |
| Infiltration | $m^3/d$ | 0.024 | 0.012 | 0.012 | — |
| HLR, $q$ | m/d | 0.057 | 0.116 | 0.118 | 0.059 |
| Nominal HRT | days | 7.98 | 3.92 | 3.85 | 7.70 |
| Concentration, $C$ | mg/L | 120.0 | 21.5 | 11.2 | 11.2 |

a large impact on the effluent quality. Here, we will consider a scenario where the wetland process must deliver an effluent quality very close to $C^*$, where there are very large impacts of compartmentalization on the wetland size.

Tables 21.1 and 21.2 consider the case of an aerated subsurface flow wetland (saturated flow) to remove COD from hydrocarbons. Because of the internal mixing induced by aeration ($N = 3$) and pollutant weathering, the wetland is assumed to function as 2 *P*TIS (aerated wetlands are addressed in more detail in Chapter 24). The spatially variable flow *P-k-C** model previously introduced in Chapters 17 and 20 is utilized here. The influent COD is assumed to be 120 mg/L, the regulatory compliance limit is 20 mg/L, and $C^*$ is assumed to be 10 mg/L. However, because of variability in the wetland performance, an effluent quality of 11.2 mg/L is targeted as the design objective so that the system will meet the 20 mg/L regulatory limits on a consistent basis (90% of the time). The lower targeted effluent concentration is not a safety factor and has nothing to do with treatment redundancy.

Further, it is assumed that there will be significant weathering of the COD mixture as it approaches $C^*$, so the two wetland cells in series are represented by $P = 3$ instead of $N = 6$.

As the target effluent concentration is so close to $C^*$, any bypassing of untreated water (low *N*TIS and therefore low *P*TIS) has a very large impact on treatment performance. In Table 21.1, the required wetland area is 2,400 $m^2$; in Table 21.2, the required wetland area is 1,500 $m^2$. So, in this particular example, there is a large benefit in terms of size reduction (35%) if the wetland is constructed as two cells in series, even though the "benefit" of the second cell has been reduced by pollutant weathering effects.

## Dividing Wetland Cells Based on Slope

On sloping sites, there will be earthwork cut-and-fill costs associated with terracing the site for wetland basins. However, wetland designers should be aware that dividing the wetland into multiple segments may potentially have impacts on treatment performance (as explored in Tables 20.8, 20.9, 21.1, and 21.2). If there are large topographic changes between wetland cells, this can provide the opportunity for creative design elements, such as cascade waterfall aerators (Burka and Lawrence, 1990), if these are advantageous from a process design standpoint.

## More Than One Wetland Type

Another obvious situation to divide the flow path into multiple cells is when there is a desire to use more than one wetland type along the flow path. This subdivision of the treatment process into unique process elements dates back to the original work of Seidel (Seidel, 1966), which is discussed in Chapter 15. Her work eventually led to the development of the current "hybrid" wetland systems,

**TABLE 21.2**
**COD Reduction for Two Wetland Cells in Series; *P*TIS = 3**

| Input Parameters | | Calculated Values | |
|---|---|---|---|
| Flow rate, $Q$ | 137.1 m³/d | Volume per tank | 228 m³ |
| Precipitation, $P$ | 2.2 mm/d | Area per tank | 500 m² |
| $ET$ | 0.13 mm/d | Influent flow, $Q_i$ | 137.1 m³/d |
| Infiltration | 0.01 mm/d | Effluent flow, $Q_o$ | 140.2 m³/d |
| $P$TIS (system) | 3 | Average flow, $Q_{avg}$ | 138.7 m³/d |
| Area, $A$ | 1,500 m² | Influent mass load | 16,452 g/d |
| Porosity, ε | 0.38 | Effluent mass load | 1,563 g/d |
| Bed depth | 1.2m | Mass % reduction | 90.5% |
| $C_i$ | 120 mg/L | Nominal HLR | 0.091 m/d |
| $C$* | 10 mg/L | HLR based on $Q_{avg}$ | 0.092 m/d |
| $k$ | 356 m/yr | HLR based on $P$TIS | 0.093 m/d |
| $k$ | 0.975 m/d | Nominal HRT | 4.99 d |
| Compliance $C$ | 20 mg/L | HRT based on $Q_{avg}$ | 4.93 d |
| Variability factor | 1.78 | HRT based on $P$TIS | 4.88 d |
| Target $C$ | 11.2 mg/L | Liner required | 2,107 m² |
| | | Piping required | 750 m |

| Calculated Values | | System In | Exit Tank 1 | Exit Tank 2 | Exit Tank 3 | System Out |
|---|---|---|---|---|---|---|
| Net flow | m³/d | 137.1 | 138.1 | 139.2 | 140.2 | 140.2 |
| Precipitation | m³/d | 3.300 | 1.100 | 1.100 | 1.100 | — |
| $ET$ | m³/d | 0.195 | 0.065 | 0.065 | 0.065 | — |
| Infiltration | m³/d | 0.015 | 0.005 | 0.005 | 0.05 | — |
| HLR, $q$ | m/d | 0.091 | 0.276 | 0.278 | 0.280 | 0.093 |
| Nominal HRT | days | 4.99 | 1.65 | 1.64 | 1.63 | 4.88 |
| Concentration, $C$ | mg/L | 120.0 | 34.0 | 15.3 | 11.2 | 11.2 |

which typically employ a combination of vertical–horizontal flow wetland cells (Urbanc-Bercic and Bulc, 1994; Cooper and Green, 1995; Platzer, 1996; O'Hogain, 2003; Obarska-Pempkowiak *et al.*, 2004; Vymazal, 2005), or a combination of the horizontal–vertical flows (with recirculation) (Brix, 1998; Laber *et al.*, 1999). Combination wetland systems of this nature are discussed in more detail in Chapter 24.

### Staged Treatment

Another situation where a designer may elect to use multiple wetland cells along the flow path is when each wetland cell is intended to provide a different treatment function. In many situations, these wetland cells will be different types of wetlands, as discussed previously and in Chapter 24.

However, in certain applications, the same type of wetlands may be linked together solely based on treatment function. A good example of this approach is the two-stage VF wetland systems in France (Molle *et al.*, 2005a). The first stage of the treatment process is often three pulse-fed VF beds that are alternately loaded and rested. The primary function of these beds is BOD and total suspended solids (TSS) removal. The second stage of the treatment process is typically two parallel pulse-fed VF beds that are also alternately loaded and rested. These second-stage cells are intended primarily for BOD polishing and nitrification.

## 21.4 CLOGGING DYNAMICS

Both horizontal and vertical flow SSF wetlands rely on the movement of water through a porous media for proper functioning. Both systems are susceptible to bed clogging, which restricts the flow of water through the bed media. In some instances, this restriction in flow will create flooding and will be considered to be a "failure" of the wetland treatment process. However, the definition of *failure* is qualitative and is often based on the perception and expectations of local regulators, government officials, system operators, and nearby residents.

### Defining Failure in SSF Wetlands

Because *failure* is a subjective term in the treatment wetland field, it is useful to define what the expectations of the regulators, owners, and users of the system might be. Several cases concerning hydraulic failure help to illustrate this point.

- In Denmark, soil-based HSSF wetland technology was adopted early and spread rapidly throughout the country. Although the design intent was subsurface flow, these systems did not have sufficient hydraulic conductivity, and overland flow occurred instead (Brix, 1998). However, the treatment performance of these systems was satisfactory and the overland flow mode was tolerated.
- In the United States, regulations for single-home septic systems prohibit the surface exposure of

wastewater because of public health concerns (U.S. EPA, 2002c). HSSF wetlands that have clogged inlet zones and overland flow (even for a small percentage of the wetland bed) are thus deemed "failed," and considerable effort and expense have gone into maintaining these wetlands to eliminate overland flow, even when treatment performance was satisfactory. A similar situation exists in the United Kingdom (Cooper *et al.*, 2006a).

- VF wetland systems in France are highly loaded with BOD and TSS (raw sewage) in the initial treatment stage (Molle *et al.*, 2004a). A large component of the initial treatment is physical filtration and deposition of organic matter on the surface of the beds. In other words, the beds are designed to clog. However, these high loads can only be tolerated for a short time; so, a loading and resting regime (at least three parallel beds in the first stage) is utilized so that there is sufficient resting interval to recover from the clogged condition via drying and cracking of the cake (Figure 7.31). Therefore, in these systems, clogging does not equate to failure because there is a routine operational program to address the clogging issue.
- However, other VF wetlands have clogged and failed. This occurs when there are insufficient beds to allow adequate rest intervals. When these beds clog, water ponds on the surface and air movement into the bed ceases, resulting in poor treatment (Platzer and Mauch, 1997; Cooper *et al.*, 1997). In extreme cases, ponded water may essentially fill the entire basin, requiring the entire wetland to be bypassed.

Thus, it can be seen that the definition of *hydraulic failure* depends on the regulatory requirements, degree of treatment flexibility, operational regime, and local expectations. These definitions of failure may or may not involve treatment efficiency issues. For design purposes in this book, the following conditions are assumed to constitute a failure mode:

- An inability to meet treatment standards on a consistent basis based on the regulatory compliance interval.
- For HSSF wetlands, the flow should be kept within the wetland bed. When clogging occurs in the inlet region, any overland flow path should be kept to a required minimum. This may be zero exposed water, or some minimal length that may be considered an inlet distribution zone. This defines failure based on length of overland flow, not the area of the bed that is clogged.
- For VF wetlands, the inability of the system to hydraulically pass the design flow. This will require sufficient beds to allow adequate resting periods to resist clogging.
- For biosolids wetlands, not having sufficient beds to allow adequate resting periods to stabilize the applied sludge.

## Clogging in HSSF Wetland Beds

The aspect ratio (L:W) of each wetland cell is an important design decision for HSSF wetlands. Longer and narrower beds increase the organic loading applied to the cross-sectional area of the bed. High cross-sectional loadings will result in a greater length of the bed operating as an overland flow system due to the clogging mechanisms described in Figure 7.25, Chapter 7. This is likely to be unacceptable from a regulatory standpoint in the United States, and if the overland flow channelizes, treatment performance will be compromised.

At the present time, there is considerable interest in determining whether or not a sustainable balance between solids deposition and solids decomposition/resuspension can be reached in the inlet zone of HSSF wetlands. This issue is of profound design importance. If a steady-state criterion can be identified, then HSSF wetlands can be designed with a sufficiently conservative inlet loading criteria so as to preclude the need for bed-media maintenance or replacement. If bed clogging is inevitable (and merely delayed by lower inlet loading rates), then the need for scheduled bed cleaning/replacement in the inlet zone becomes an integral part of HSSF wetland technology operation and lifecycle costs.

To achieve a steady state in the inlet region, the deposited inlet-suspended solids and the microbial biomat generated by particulate/soluble organic matter must be in equilibrium with the combined rate of organic solids decomposition and the rate of particulate matter resuspension. This implies that the rate of accumulation, $A$, in the inlet zone is zero. This condition can be represented (using terms previously presented in Equation 7.16, Chapter 7) as indicated in Equation 21.1:

$$G + S = D + R \tag{21.1}$$

where

$G$ = generation rate, g/m$^2$·d
$S$ = settling rate, g/m$^2$·d
$D$ = decomposition rate of transportable solids, g/m$^2$·d
$R$ = resuspension rate, g/m$^2$·d

Reduction in the hydraulic conductivity of the inlet zone can be attributed to a variety of mechanisms, including

1. Accumulation of mineral (biologically inert) sediments associated with influent TSS
2. Accumulation of particulate organic matter (subject to biodegradation) within the inlet zone
3. Formation of chemical precipitates within the wetland bed
4. Generation of microbial biofilms in response to the combined load of the particulate and soluble organic matter

These materials combine to form a mixture that is here termed *biosolids*, and the degree of biosolid accumulation depends on all four mechanisms. Because a significant portion of the incoming organic load in domestic wastewaters is present as

decomposable particulate organic matter, depositions of this particulate organic matter may result in organic loadings tenfold higher in the inlet region as opposed to the rest of the wetland bed (Puigagut *et al.*, 2006). This has profound effects in biosolid formation in the inlet zone. Reductions in the hydraulic conductivity from biosolid accumulations are a function of the media size, as discussed in Figure 7.24, Chapter 7.

### Short-Term HSSF Inlet Zone Clogging

At the present time, there is insufficient knowledge to quantify the generation $G$, decomposition $D$, and resuspension $R$ rates within HSSF wetland beds. This is largely due to the difficulties encountered when doing *in situ* sampling of HSSF systems, as previously discussed in Chapter 7. Without the ability to quantify the terms $G$, $D$, and $R$, the alternative is to extract design information from the performance of existing HSSF wetlands, as shown in Figure 21.5.

Data illustrated in Figure 21.5 suggests that HSSF wetland beds should be configured such that the cross-sectional area (orthogonal to the flow) results in a BOD loading of <250 g/m$^2$·d for bed medias with a $d_{10} > 4$ mm. BOD has been selected as the influent loading parameter in this case because (1) the majority of degradable organic matter in municipal effluents is present as particulates, (2) soluble organic matter also contributes to inlet zone biosolids formation, and (3) chemical precipitates (as represented by the difference between COD and BOD) occur more uniformly in the HSSF bed and are likely not restricted to the inlet zone.

There are drawbacks to limiting performance information to BOD, as illustrated in Figure 21.5. First of all, a HSSF wetland with a high inlet flow of nonorganic (mineral) TSS may plug at organic loads much lower than those indicated in Figure 21.5. Secondly, the nature of the organic material must be considered. Some waste streams contain organic matter that is much more degradable than others. HSSF wetlands treating soluble, rapidly degradable organic material can reasonably be expected to have a shorter clogged inlet zone.

### Long-Term HSSF Inlet Zone Clogging

Systems illustrated in Figure 21.5 generally have an operational performance history less than ten years. Although reducing the cross-sectional organic loading may have short-term benefits in reducing bed clogging, the long-term viability of HSSF hydraulic conductivity remains uncertain. Two factors argue against the ability to stabilize long-term hydraulic conductivity of HSSF beds. These include

- Not all organic matter deposited in the inlet zone is biodegradable. The refractory (nondegradable) component represents a long-term accumulation of organic TSS.
- A certain component (variable between HSSF systems) of influent TSS consists of nondegradable (mineral) material. This inert material is efficiently removed from the water column through the particulate settling/filtration/interception mechanisms discussed in Chapter 7. Because these removal mechanisms are very efficient in HSSF beds, most of the TSS reduction occurs in the early stages of the wetland bed (see, for example, Figure 7.26). Long-term loading of inert TSS results in an ongoing reduction of the hydraulic conductivity in the inlet zone.

The current understanding of HSSF wetland design does not allow a quantitative determination of the region of inlet

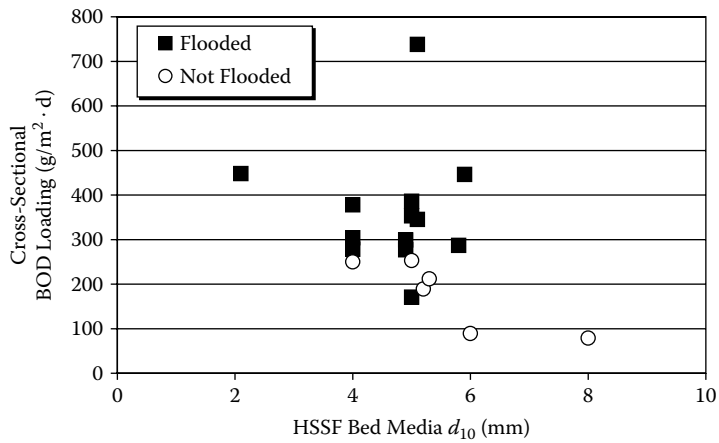



**FIGURE 21.5** Flooding status of HSSF wetlands as a function of cross-sectional organic loading and bed media size. (Data from Steinmann *et al.* (2003) *Water Research* 37: 2035–2042; Wallace and Knight (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements*. Final Report, Project 01-CTS-5, Water Environment Research Foundation (WERF): Alexandria, Virginia; Puigagut *et al.* (2006) *Size distribution and biodegradability properties of the organic matter in horizontal subsurface flow constructed wetlands*. Kröpfelová (Ed.), Paper presented at the 6th International Workshop on Nutrient Cycling and Retention in Natural and Constructed Wetlands, 31 May–4 June 2006; Trebon, Czech Republic.)

clogging (biosolids penetration distance), although this is believed to be related to the size of the bed media, as indicated in Figure 21.5. The $d_{10}$ of the media is often used to approximate the pore volume, as only 10% of the bed media is smaller than this size.

Studies on sand clogging indicate that the clogging distance $D_c$, can be related to the $d_{10}$ of the media: $D_c \approx 150\ d_{10}$ within the range of $5 \times 10^{-3}$ mm $< d_{10} < 3{\times}10^{-2}$ mm (Blazejewski *et al.*, 1994). Because this $d_{10}$ is much smaller than the media sizes used in most HSSF wetlands, the usefulness of this relationship is limited to soil-based HSSF systems. Blazejewski *et al.* (1994) suggested a clogging thickness of 3 cm for fine-grained HSSF beds.

Bavor *et al.* (1989) noted that clogging within a series of long, narrow HSSF trenches (L:W ratio of 25:1) was remedied by excavating the first 5 m of the bed and replacing with coarse rock. Kadlec and Watson (1993) provided evidence from the Benton, Kentucky, system that the farthest biomat-penetration distance stretched 100 m into a 300-m bed. But this type of information does not help in design unless couched in terms of organic loadings and decomposition rates.

Based on the information currently available, it appears that bed clogging (as evidenced by overland flow length) is reduced, or at least delayed, by minimizing the cross-sectional loading, and using coarser bed materials in the inlet zone.

However, based on the current state of the art, there is no available evidence that argues against the phenomenon of long-term bed clogging ($A > 0$), even if short-term clogging pitfalls are avoided. Long-term performance information from the United Kingdom (Cooper *et al.*, 2006b) indicates that HSSF systems will require some degree of inlet zone media replacement or cleaning during the course of their operational lives. Whereas the frequency of this maintenance is dependent on the bed clogging factors described in this book, it appears that even well-designed, well-operated HSSF wetlands will require inlet zone maintenance approximately every ten years to avoid the development of large regions of overland flow. At the present time, media replacement is the most common (and most expensive) method of maintenance (Wallace and Knight, 2006) although trials using hydrogen peroxide as a cleaning agent have shown promising results (Hanson, 2002b; Behrends *et al.*, 2006).

### Clogging in VF Wetland Beds

VF wetlands are also subject to solids accumulation, which can fill the pore volume within the upper region of the wetland bed, potentially leading to ponding and hydraulic failure. Equations describing clogging of VF wetland beds have previously been presented in Equations 2.58–2.61 in Chapter 2.

The organic loading applied to the VF wetland appears to play a large role in the clogging process. For a soil/sand VF wetland at Merzdorf, Germany, Platzer and Mauch (1997) observed a log-linear decrease of hydraulic conductivity as a function of the cumulative applied organic load. With resting intervals, the hydraulic conductivity of the bed was restored to the original values.

At the present time, there are two operating regimes to address the clogging of VF wetland beds:

1. Systems that are continuously operated without resting. These VF wetlands are loaded at relatively low organic loads to minimize biosolids accumulation and associated clogging of the bed.
2. Systems that employ multiple beds in a load-and-rest regime. These VF wetlands are designed to clog during the loading phase; hydraulic conductivity is restored during the resting phase.

Because clogging and organic loading appear to be intimately related, it is not possible to specify an organic loading rate for a VF wetland system without also specifying the resting sequence associated with that loading, as previously discussed in Chapter 20.

Approximately 180 VF wetlands in North America have been designed as recirculating systems, which are functionally very similar to recirculating gravel filters (Crites and Tchobanoglous, 1998). Recirculation ratios vary depending on design goals, but ratios of 3 to 12 times the inlet flow rate are common.

An array of distribution pipes results in a nonuniform application of influent TSS and organic matter. This loading is concentrated at the discharge orifices of the distribution pipes. Biosolids then preferentially form at each discharge orifice in a zone about 30 cm in diameter. In extreme conditions, these biosolids will plug the upper layer of the wetland bed because recirculating wetlands operate at hydraulic loadings that are much higher than the inlet flow reductions in bed porosity and loss of hydraulic conductivity can rapidly lead to flooding of the wetland bed. An example is shown in Figure 21.6.

Operational experience in the United States indicates that the organic loading (as represented by BOD) should be less than 15 g/d per orifice for buried pipes to avoid biosolids clogging with bed media greater than 4 mm. The equivalent BOD loading in the biosolids clogging region at the orifice is approximately 200 g/m$^2$·d.

The uniformity of application is increased and the organic loading is reduced if there is a secondary redistribution of the influent by splashing or spraying. Figure 21.7 shows a cold-climate method of achieving this. Perforated pipes are placed in infiltration chambers that are then buried in the upper portion of the wetland. Water sprays out of the distribution orifices up into the infiltration chamber, where it splashes around and undergoes secondary distribution. Orifice shields (resembling plastic pipe caps) are commercially available for this purpose.

## 21.5 CELL CONFIGURATION

As discussed in the previous section, clogging dynamics will often determine the number of wetland treatment cells and, for HSSF wetlands, the aspect ratio (L:W) will be a function of the cross-sectional loading if delaying or minimizing the



**FIGURE 21.6** Example of biosolids clogging in a small recirculating VF wetland. This system in central Minnesota experienced rapid hydraulic failure in response to extremely excessive BOD loadings from illegal drug manufacturing.

overland flow is a design goal (see Figure 21.5). Even if the cell dimensions are governed by clogging considerations, there are still a number of design decisions. Several factors of SSF wetland bed design warrant further consideration at this stage; these include

- The length-to-width (L:W) ratio of the wetland cells
- The depth of the wetland cells
- The types and sizes of the bed media that will be placed in the wetland cells
- The overall size of the wetland cells

### Length-to-Width Ratio

For VF and biosolids wetlands, the aspect ratio (L:W) is relatively unimportant, because these wetland beds are top loaded. For HSSF wetlands, the aspect ratio is critically important, because this determines the cross-sectional loading and flow velocity and, to some extent, the *N*TIS value.

When working with HSSF wetlands, the designer is faced with two mutually contradictory design goals:

- The desire to spread the influent over a very wide planar area. (This delays the time to clogging.)
- The desire to spread the influent as uniformly as possible over the vertical cross section. (This improves flow distribution.)

Spreading the influent over wide areas presents a number of hydraulic challenges, especially for small systems, where the flow rate is close to zero. One approach, which has been implemented in Denmark, is to divide the inlet chamber into two or more segments that can alternately be loaded to gain

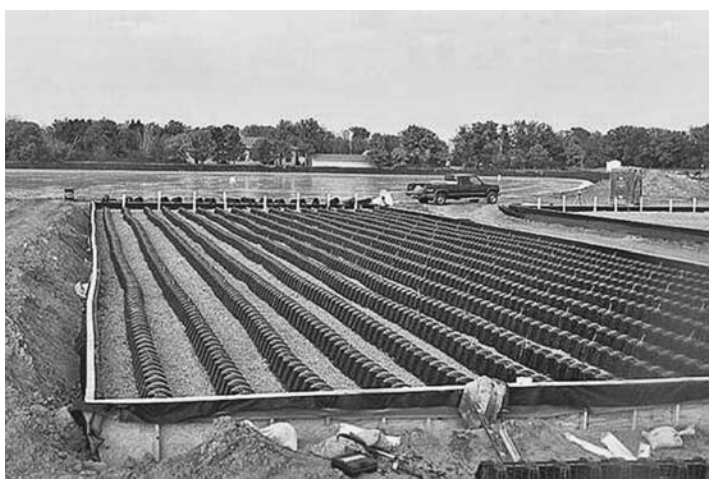


**FIGURE 21.7** Recirculating VF wetland under construction at The Ponds, Minnesota. The black infiltration chambers are used as a means of secondary flow distribution to reduce the organic loading at the distribution orifice. (This photo was taken at an intermediate stage of construction; additional gravel was later placed to bury the influent distribution chambers.)

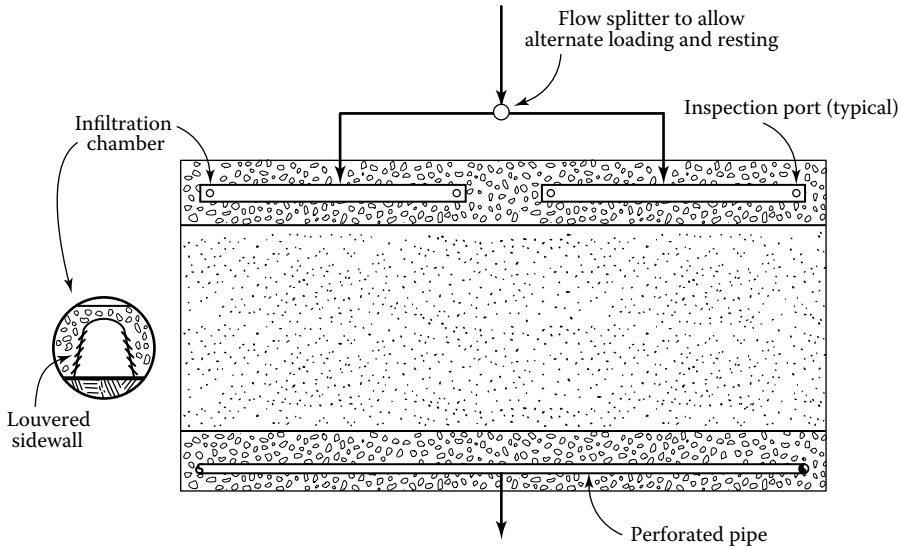



**FIGURE 21.8** Layout and design factors for a HSSF wetland utilizing an influent flow split to enhance flow distribution. (From Wallace and Knight (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements*. Final Report, Project 01-CTS-5, Water Environment Research Foundation (WERF): Alexandria, Virginia. Reprinted with permission.)

better control of the flow distribution (Brix, 1998). A schematic of this is shown in Figure 21.8.

For small-scale wetlands, flow-splitting approaches use weirs, stoplogs, or rotating elbows if flow balancing is an operations goal. All such flow-splitting devices require periodic maintenance to remove accumulated solids that may interfere with the flow split, especially for small-scale systems with low flows.

An alternate approach is to split the flow two ways using a central effluent collection pipe with influent distribution channels on either side of the wetland. An example of this design strategy is presented in Figure 15.5, Chapter 15.

## Wetland Cell Depth

HSSF wetlands were originally designed with a depth of 60 cm based on the belief that this was the maximum depth that *Phragmites* root systems would penetrate (Kickuth, 1977). However, analysis of many HSSF wetlands has indicated that the root systems of emergent wetland plants grow preferentially in the upper region of the bed; this concept is illustrated in Figure 3.4 in Chapter 3, with the zone of root penetration typically being only about 30 cm. Thus, deeper beds often have a lower region without plant roots, and water flows preferentially along the bottom of the wetland bed. This lower region is typically associated with more reducing conditions and less efficient treatment.

For HSSF wetlands, the designer is faced with choosing the bed depth. Some observations include the following:

1. Deeper beds promote vertical stratification of flow, as discussed in Chapter 2. This allows bypassing of the flow below the plant root zone (Fisher, 1990; Pilgrim *et al*., 1992; Marsteiner *et al*., 1996; Garcia *et al*., 2004b), with adverse impacts on treatment performance.
2. Shallower beds often result in better treatment (see Tables 8.14 and 9.33 for BOD and ammonia, respectively).
3. Deeper beds provide a greater cross-sectional area, which reduces cross-sectional loading (see Figure 21.5). This theoretically reduces the flow velocity (in the absence of vertical stratification), resulting in less head loss through the wetland bed, which makes maintaining a uniform water surface profile easier.
4. Deeper beds speculatively provide additional volume for storage of refractory and nondegradable (mineral) sediments. This added storage volume would presumably decrease the time intervals between clogging and associated bed maintenance.
5. A deep bed speculatively allows for some ice formation at the top while still passing the water underneath the ice within the bed.

From a treatment standpoint, the additional cost of the extra bed depth (e.g., 60 cm versus 30 cm) appears to add zero or negative value to a fixed area of wetland, but there is no quantification across multiple systems. Control of head loss is better done through aspect ratio and media size. If sediment storage is a design consideration, more bed depth may be warranted. Ice formation can be adequately controlled via a variety of other insulation techniques. To date, virtually all HSSF wetlands have been designed with beds between 30 and 60 cm deep. But, controlled field-scale research studies indicate that shallower depths provide better treatment.

VF and biosolids wetlands are typically constructed to a prespecified bed depth, 50–80 cm for VF wetlands and 50–60 cm for biosolids wetlands.

### Type and Size of Bed Media

Whereas HSSF beds almost always use coarse stone in the inlet and outlet regions, a variety of materials are used for the main bed media.

The original HSSF beds utilized sand in the Max Planck Institute Process (MPIP) system (Seidel, 1966) and soil in the root zone method beds (Kickuth, 1977). Many more of the root zone method systems were constructed, despite significant problems with the hydraulic conductivity and associated problems of overland flow (Brix, 1998).

The response to the overland flow problem was to design HSSF beds with coarser materials so that the water could flow through the wetland bed. Different countries took various routes in the development of HSSF wetland technology, and there are regional differences in the size and type of the bed media. Perusal of Table 10.14, Chapter 10, will provide an indication of the wide variety of bed medias that have been evaluated. In certain cases, the use of media with specific chemical properties, such as phosphorus sorption or a high cation exchange capacity (for ammonia retention), may be desirable. Examples of bed media and size gradations that have been used in HSSF wetlands are summarized in Table 21.3.

Finer materials have a greater surface area for microbial biofilms and may result in better treatment, as indicated in Table 9.33, Chapter 9, for ammonia. Finer materials are more prone to clogging and associated hydraulic problems, as illustrated in Figure 7.23, Chapter 7. The use of coarser material in the first part of the HSSF wetland bed and finer material closer to the outlet has been proposed as a compromise between these two competing design objectives (Wallace and Knight, 2006). The type of wastewater may also be a factor, because biosolids production and/or distribution appears to be a function of the organic loading. With lower organic loadings, finer bed medias could conceivably be used without clogging for the same time period as coarser materials with higher loadings, as proposed in the Austrian design guidelines (ÖNORM B 2505, 2005) and summarized in Table 21.3.

Economics, availability, and local design practices typically dictate what material will be preferred for the wetland bed. To date, medias in the size range of 8 to 16 mm have been most commonly used in HSSF wetlands (IWA Specialist Group on Use of Macrophytes in Water Pollution Control, 2000).

VF wetlands typically use a layered approach to the bed, with fine material (typically sand) at the top of the wetland bed and progressively coarser materials lower in the bed—down to the drainage layer. It has long been recognized that the composition of the sand layer is critical; the material must be fine enough to trap TSS on the surface of the bed but coarse enough to allow water movement without clogging (Cooper, 2003). Depending on the level of pretreatment, finer bed media may be warranted; the French systems use coarser bed material in the first VF stage (which treats raw sewage) and finer bed material in the second VF stage, which provides COD polishing and nitrification (Molle *et al.*, 2004a).

Biosolids dewatering wetlands typically have a fine layer (sandy loam or sand) with an underlying drainage layer (Nielsen, 2002). However, it has been noted that it is important to maintain continuity of capillary suction to facilitate flow through the various drainage layers; otherwise, the result will be "hanging water" as evidenced by a lack of sludge dewatering (Nielsen, 2004). So, large differences in the size range between adjacent drainage layers should be avoided.

Different bed media and layering for VF wetlands are summarized in Table 21.4.

**TABLE 21.3**
**Examples of Bed Materials Used in HSSF Wetlands**

| | Main Bed Media | | | Inlet/Outlet Regions |
|---|---|---|---|---|
| **Source** | **Media Size** | **Material** | **Hydraulic Conductivity** | **Media Size** |
| EC/EWPCA (1990) | 3–6; or 5–10 mm | Gravel | 86 m/d | 50–200 mm stone |
| TVA (1993) | 3–6 mm | Gravel | 2,600 m/d (secondary effluent)<br>260 m/d (primary effluent) | 50–100 mm stone |
| ATV (1998) | 0.2–1.0 mm | Soil/sand | $k = \frac{(d_{10})^2}{100}$<br>where $k$ is in m/s and $d_{10}$ is in mm | |
| U.S. EPA (2000a) | 20–30 mm<br>5–20 mm top layer | Gravel | 100,000 m/d<br>10,000 m/d inlet region | 40–80 mm stone |
| IWA (2000) | 8–16 mm | Gravel | $k = 12{,}600\ D_P^{1.9}$<br>where $k$ is in m/d and $d_{10}$ is in cm<br>use 10% of this value in the inlet region | 75–100 mm stone |
| ÖNORM B2505 (2005) | 0–4 mm (gray water)<br>1–4 mm (secondary influent)<br>4–8 mm (primary influent) | Sand<br>Gravel<br>Gravel | Use the saturated hydraulic conductivity; head loss is sized on 1/10 of the daily flow being delivered in 1 hour | 16–32 mm gravel or gradated bands of 8–16 and 4–8 mm gravel |
| Wallace and Knight (2006) | >4 mm | Gravel | 260 m/d | 25–50 mm gravel |

**TABLE 21.4**
**Summary of Bed Materials Used in VF Wetlands**

| Source | Wetland Type | Layer | Thickness | Media Size | Material |
|---|---|---|---|---|---|
| EC/EWPCA (1990) | Pulse-fed | Surface layer | 8 cm | — | Sharp sand |
| | | Second layer | 15 cm | 6 mm | Pea gravel |
| | | Third layer | 10 cm | 12 mm | Round gravel |
| | | Drainage layer | 15 cm | 30–60 mm | Round gravel |
| ATV (1998) | Pulse-fed | Main layer | >80 cm | 0.2–1.0 mm | Soil/sand |
| Molle *et al.* (2004a) | Pulse-fed (First stage) | Surface layer | >30 cm | 2–8 mm | Fine gravel |
| | | Second layer | 10–20 cm | 5–20 mm | Gravel |
| | | Drainage layer | 10–20 cm | 20–40 mm | Coarse gravel |
| Molle *et al.* (2004a) | Pulse-fed (Second stage) | Surface layer | >30 cm | $0.25 < d_{10} < 0.4$ mm | Sand |
| | | Second layer | 10–20 cm | 3–10 mm | Gravel |
| | | Drainage layer | 10–20 cm | 20–40 mm | Coarse gravel |
| ÖNORM B2505 (2005) | Pulse-fed | Surface layer | 10 cm | 8–16 mm | Gravel |
| | | Second layer | >50 cm | 0–4 mm | Sand/gravel |
| | | Third layer | 5–10 cm | 4–8 mm | Gravel |
| | | Fourth layer | Separation fabric | — | 2.5-mm mesh |
| | | Drainage layer | 20 cm | 16–32 mm | Gravel |
| NAWE (2006) | Recirculating | Main layer | 75 cm | 12–15 mm | Gravel |
| | | Drainage layer | 15 cm | 16–40 mm | Coarse gravel |
| Nielsen (2002) | Biosolids dewatering[a] | Surface layer | 5–10 cm | — | Sandy loam |
| | | Second layer | 15 cm | | Sand |
| | | Third layer | Separation fabric | | — |
| | | Drainage layer | 30–45 cm | | Pebble gravel |

[a] Sludge dewatering effectiveness is dependent on maintaining continuity of the capillary suction under unsaturated flow. Therefore, large transitions in the size range of adjacent drainage layers should be avoided, especially between the overlying biosolids material and the top layer of the filter (Nielsen, 2006).

### SSF Wetland Cell Size

Another factor enters into the design process when determining the footprint area of individual wetland cells, specifically the construction equipment that will be used to place the bed media. Even large excavators have a maximum reach of approximately 10 m (see Figure 21.1). Therefore, SSF wetland cells greater than about 20 m wide cannot be filled with standard excavators, and the remaining construction options consist of

- Allowing equipment to operate inside the wetland basin (typically, skid-steer loaders for small basins, or bulldozers for large basins)
- Using a specialized piece of equipment, such as an extendable conveyor, to place the bed media

The pros and cons of these different approaches are discussed under the SSF wetland construction section of this chapter. However, the designer should give thought to construction methods and the type of construction equipment locally available when determining the dimensions of individual wetland cells.

## 21.6 SSF WETLAND HYDRAULICS

Once an overall size has been determined for the wetland system, and decisions about the number of flow paths and wetland cells have been made, the next stage in the design process is to determine the hydraulic profile of the system.

There are a number of hydraulic aspects that must be considered when designing SSF wetlands. Generally, these will include the following elements:

- The overall hydraulic profile for the treatment process, including frictional losses within the wetland bed
- Distribution of the influent flow
- Collection of the effluent flow
- Water level control

Gravity flow through the treatment process is the preferred situation from an operations and reliability standpoint. Pumping of the flow is less desirable because of the reliability, maintenance, and power consumption issues associated with wastewater pumps. On very flat sites, grading the site to achieve the "stair-step" configuration needed for gravity

flow must be weighed against the construction and operations costs of pumping. For countries with a high level of infrastructure support, such as the United States, pumping is perceived as less expensive than extensive earthwork during construction. For developing countries, the inverse may be true.

The hydraulic profile for a wetland system results from a number of different calculations. Major components of the hydraulic profile for SSF wetlands typically include:

- The driving head needed to distribute the influent flow
- Frictional losses (head loss) in conveyance piping
- Elevation differences between different stages of the treatment process
- Frictional losses in the wetland bed (HSSF wetlands)
- If the bed is intended to completely drain, the depth of the bed
- The elevation and hydraulic characteristics of water level control devices such as weirs or standpipes

Separate calculations are made for each unit process, resulting in a stair-step diagram describing the relative elevations of the water surface profile (and for pumped systems, the hydraulic grade line). Typically, calculations are made at different flow rates, such as the average daily flow, the minimum flow rate, and the peak daily or hourly flow rates. The peak flows should include the capture of water from rainfall events; often a 10-year or 25-year storm event is used for this purpose, depending on the regulatory jurisdiction.

For very small wetland systems, the low flow at night will be zero due to the diurnal fluctuation in water use patterns. This can create a number of design challenges in terms of flow distribution and, in very cold climates, freezing of pipes. Specific design challenges for small-scale wetland systems have been addressed in the literature (Wallace and Knight, 2006).

Generally speaking, the larger the system, the more important detailed hydraulic calculations become. For SSF wetlands, these calculations must also address anticipated phenomena such as bed clogging (see Chapter 7). The preferred design practice for SSF wetlands is to make each bed completely drainable and also to have a discrete elevation drop between cells (often, a minimum drop of 0.3 m is desirable, unless the site cannot support this elevation change or hydraulic calculations indicate that this differential is insufficient).

## Hydraulics of HSSF Wetlands

For HSSF wetlands, the water surface profile within the wetland bed is assumed to be governed by frictional losses within the wetland bed. If flow is within the laminar range, a simple relationship such as Darcy's law can be used to compute the frictional loss (Equation 2.40, Chapter 2), restated here as Equation 21.2.

$$u = -k\frac{dH}{dx} \tag{21.2}$$

where
$H$ = elevation of the water surface, m
$k$ = hydraulic conductivity, m/d

This is the one-dimensional version of Darcy's law. It is restricted to the laminar flow regime. At higher flow velocities, there will be a turbulent component to the flow, which must also be considered (see Equations 2.44, 2.46, 2.47, Chapter 2). The use of these friction-loss equations are only the start of the hydraulic design process, because what is of most use to the wetland designer is the longitudinal depth profile. This is typically a nonuniform profile due to changes in the flow rate (gains or losses from precipitation, infiltration, and ET) and hydraulic conductivity of the wetland bed. Calculation procedures have been previously established in Equations 2.48–2.54, and the results of such calculations are illustrated in Figure 2.26.

These calculations are typically performed by dividing the wetland bed into subunits (each with its own hydraulic conductivity) and calculating the head loss across each subunit. The standard procedure is to start at the effluent end of the bed (the tailwater condition) and work backwards to the inlet.

Here, we explore the effects of design parameters on the water surface and its relation to the media surface. The goals of hydraulic design for HSSF wetlands are to convey the design flow within the wetland bed without surfacing while providing an acceptable hydrologic environment for the survival of emergent wetland plants. Two philosophies may be used to achieve these goals:

1. A management-intensive scheme, in which operators compensate for hydraulic upsets through operational adjustments
2. A design-intensive scheme, in which the design is made sufficiently conservative so as to eliminate the need for most operational adjustments

In either case, the overall wetland footprint area is presumed to be set from the treatment requirements as previously discussed in Chapter 20.

Normally, the requirement for treatment performance will provide a specification of either the hydraulic loading rate or detention time. In turn, these combine with the required volumetric flow rate to determine either the volume of water in the bed or area of the bed. However, it must be recognized that many wastewater sources create flows that vary considerably on both diurnal and seasonal time scales. Neither the minimum nor the maximum anticipated flow should cause hydraulic failure.

The requirements for stable water depth and controllable water flow for creating proper vegetation conditions serve to further restrict the geometry of the bed and the size of the media.

These requirements can be summarized as

1. Anticipated design flows must pass through the bed without overland flow or flooding.
2. Anticipated design flows must pass through the bed without stranding the plants above water; i.e., there must not be protracted, excessive headspace that results in desiccation and die-off of wetland plants.
3. Operation should remain acceptable in the inevitable event of changing hydraulic conductivity, especially in the inlet zone. As the bed clogs with roots and microbial biofilms it should not flood.
4. The bed should be drainable.
5. Water levels within the system should be fully controllable through the use of inlet and outlet structures.
6. The configuration must fit within the available site area in terms of project boundaries and hydraulic profiles.

There are often serial and parallel arrangements of individual wetland cells within a system. In the present discussion, attention is restricted to individual cells and to simple rectangular geometries. The variables that may be chosen to satisfy these considerations are: length ($L$), bottom slope ($S_b$), and the media ($D$ or $k$). Bed depth ($\delta$) is usually in the narrow range of 30–60 cm and is therefore set by the desire for plant root penetration within the HSSF bed rather than hydraulics. *The width of the system is scaled by the need to carry the full design flow and by clogging concerns.*

There is no theoretical need for a slope to the top of the bed. If water level control is to include the ability to totally inundate the bed for vegetation management, then a top slope is detrimental.

Because the media is likely to undergo significant changes in hydraulic conductivity, its frictional resistance cannot be relied upon for control of the water depth. This implies that the conductivity of the media must be large enough, or the water travel distance short enough, to sustain a nearly level pool condition controlled by the elevation of the wetland outlet control structure.

In certain cases, it may be desirable to be able to drain the HSSF wetland bed. In this case, the bottom slope should be set to provide for complete bed drainage. Normally, a few centimeters of elevation differential allows for this requirement. The wetland bottom slope cannot be considered as the design driving force for water movement. The reason is that designs based on bed slope are excessively sensitive to changing conditions of flow and hydraulic conductivity. Dry-out or flooding are virtually certain to occur with such designs.

### *Headspace Considerations*

Under conditions of very low flow, the water surface will be a horizontal level pool controlled by the elevation of the outlet structure. It is necessary that this condition should not create a dry zone of excessive depth (headspace, $f$) (Figure 21.9), which would occur due to a tilt of the upper bed surface. If the top and bottom of the HSSF wetland are both sloped, it creates a condition very conducive to overland flow within the system, as illustrated in Figure 21.10.

However, many old HSSF systems have been built to specifications that require uniform bed depth and a significant bed bottom slope. Indeed, early design guidance documents recommended high bottom slopes: 0.5–2.0% (TVA, 1993); 1–5% (Cooper *et al.*, 1996). Another early guideline was established for Danish systems, namely, restricting the bed surface slope to no more than a 30-cm drop, thereby ensuring that plant roots could reach water even under level pool conditions (Johansen, 1994).

The effect of bed slope is seen in Figure 2.26, Chapter 2, for the Benton, Kentucky, system. There, a 0.1% bed slope over 334 m ($\Delta B$ = 33 cm) results in a drained bed water elevation that is not compatible with the hydroperiod needs of emergent wetland plants at low flow conditions. The result was unsuitable water conditions for the target wetland plant species (*Scirpus validus*).

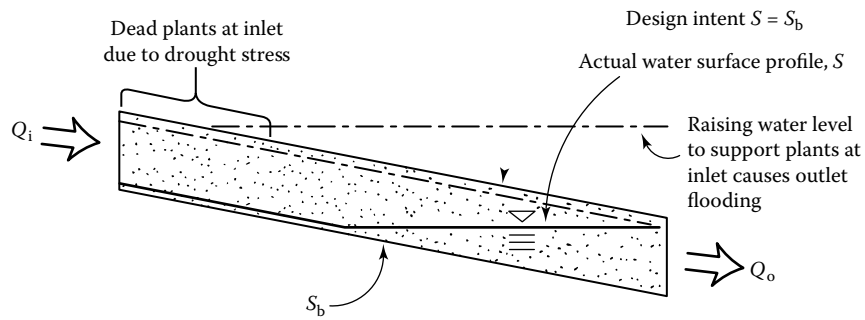



**FIGURE 21.9** Water level problems occurring in sloped HSSF wetland beds under conditions of low flow. (From Wallace and Knight (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements*. Final Report, Project 01-CTS-5, Water Environment Research Foundation (WERF): Alexandria, Virginia. Reprinted with permission.)

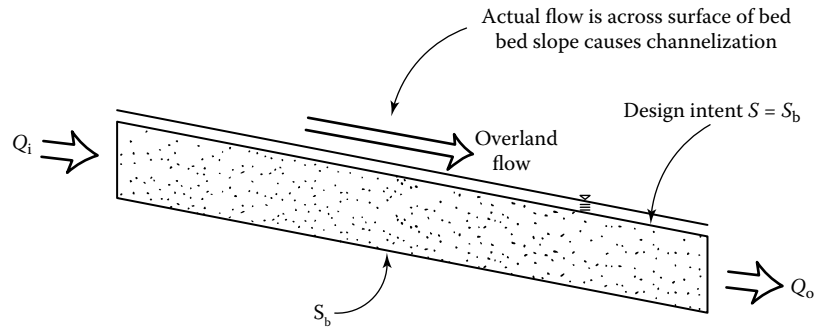



**FIGURE 21.10** Overland flow problems resulting from HSSF beds with high flow and/or low hydraulic conductivity. (From Wallace and Knight (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements*. Final Report, Project 01-CTS-5, Water Environment Research Foundation (WERF): Alexandria, Virginia. Reprinted with permission.)

Under conditions of high flow, the headspace concern moves to the outlet end of the system. When the outlet water level is lowered to prevent inlet zone flooding, the result may be excessive freeboard at the outlet. Emergent wetland plants cannot tolerate this hydrologic regime, and the vegetation will become terrestrial (Figure 21.11).

If the drop in the water level is extreme enough, desert-like landscapes sometimes result at the wetland outlet, because the target wetland plants may not tolerate this hydroperiod (Figure 21.12). Neither for the low flow or high flow scenario, the maximum allowable headspace should exceed plant tolerance for the depth to the water surface. As a conservative (safe) rule of thumb, headspace should be limited to about a third the maximum rooting depth, or about 10 cm:

$$f < 10 \text{ cm} \tag{21.3}$$

If there is little or no slope to the bed surface, using a media with a sufficiently high hydraulic conductivity will prevent these design pitfalls.

### *Hydraulic Loading Considerations*

Equations 2.47–2.52, Chapter 2, provide the framework for determination of the allowable flow rate per unit width that may be passed through a given media of specified length, without creating excessive headloss. Based on prior considerations, the case of interest is a level bottom and a level top to the bed. The datum may then be chosen as the bed bottom, and therefore $h = H$. Hence, the simplification of Equation 2.54 results, which, on substituting a mean depth, describes a quadratic relation between depth and flow:

$$h_i^2 = h_o^2 + 2\frac{L}{W}\frac{Q}{k} \tag{21.4}$$

A nonlinear relation of this type has been reported by many investigators (Kadlec *et al*., 1981; Hobson, 1989; Ranieri, 2003). Headloss is exacerbated by high length-to-width (aspect) ratios and high flows, or low conductivity. Equation 2.56 may be rearranged for use in conceptual design to determine the maximum allowable flow to a specified media and geometry:

$$Q < \frac{k}{2}\left(\frac{h_i^2 - h_o^2}{L/W}\right) \tag{21.5}$$

The allowable headloss ($h_i - h_o$) would typically be restricted to about 5 cm. The outlet depth would be set about 10 cm below the bed surface to allow for headloss and moderate rain events without flooding. As an example, for a 60-cm bed depth, this gives $h_o$ = 50 cm and $h_i$ = 55 cm. The two

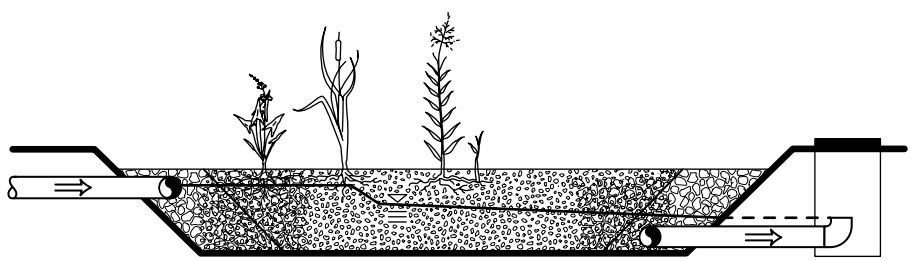


**FIGURE 21.11** Lowering of the outlet control to prevent inlet zone flooding. In extreme cases, this can exceed the hydrologic tolerances of the wetland vegetation (see Figure 21.12). (From Wallace and Knight (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements*. Final Report, Project 01-CTS-5, Water Environment Research Foundation (WERF): Alexandria, Virginia. Reprinted with permission.)



**FIGURE 21.12** A desiccated zone of the Columbus, Mississippi, HSSF treatment wetland.

remaining design choices are the media conductivity (size) and the aspect ratio (*L*:*W*).

The literature abounds with advice on aspect ratio. TVA (1993) suggests that cells should be made as long and narrow as possible, and gives examples with $4 < L/:W < 28$. Reed *et al.*, (1995) pronounce an aspect ratio of 2:1 if site conditions permit. U.S. EPA (2000a) provides methods that generally result in $0.5 < L/:W < 1.0$. Treatment effectiveness may not depend strongly on aspect ratio (Bounds *et al.*, 1998), but hydraulics definitely do. Selection of a bed configuration that does not take into account subsurface flow hydraulics, or does not factor in the effects of bed clogging, will lead to flooding problems.

The literature also abounds with advice on hydraulic conductivity. TVA (1993) prescribes 3- to 6-mm washed gravel, which is presumed to have a conductivity of 260 m/d. Measured whole-bed conductivities were typically in the range of 4,000 to 8,000 m/d (Watson and Choate, 2001), so a sizeable safety factor is associated with the TVA approach. U.S. EPA (2000a) concludes that size does not matter much for treatment and suggests 20–30 mm smooth stone, for which a bare stone conductivity of 100,000 m/d is assumed. However, a safety factor of 100 is used to account for clogging in the inlet zone, resulting in a design $k = 1{,}000$ m/d.

As an example, consider the EPA (2000a) recipe of $k = 1{,}000$ m/d and $L{:}W = 1.0$. The maximum allowable flow would be

$$Q < \frac{1{,}000\ \text{m}^3/\text{d}}{2}\left(\frac{(0.55\ \text{m})^2 - (0.50\ \text{m})^2}{1.0}\right) = 26.25\ \text{m}^3/\text{d}$$

The water surface profile associated with this limiting condition is nearly a straight line. It is noteworthy that this constraint applies without regard to the size (area) of the wetland. If the maximum design flow is more than this limit, the wetland would need to be replicated at $L{:}W = 1.0$.

The foregoing illustrative analysis is based on a uniform bed, with no spatial variation from inlet to outlet. But this is clearly not often the case owing to preferential inlet zone clogging. It is excessively conservative to assign a worst-case (low) conductivity to the entire bed. A two-zone approach is preferred, as has been suggested by U.S. EPA (2000a), with a low conductivity in the inlet zone and a higher value in the outlet zone (see Figure 2.22, Chapter 2). Because there is a factor difference of ten or more in these, the headloss in the outlet zone is considerably less than in the inlet zone. It becomes useful to consider a two-piece headloss formula:

$$h_i^2 = \left(h_m^2 + 2\frac{L_i}{W}\frac{Q}{k_i}\right) = \left(h_o^2 + 2\frac{L_o}{W}\frac{Q}{k_o} + 2\frac{L_i}{W}\frac{Q}{k_i}\right) \tag{21.6}$$

where

$k_i$ = inlet zone hydraulic conductivity, m/d
$k_o$ = outlet zone hydraulic conductivity, m/d
$L_i$ = inlet zone length, m
$L_o$ = outlet zone length, m
$h_m$ = depth at end of inlet zone, m

There are now two conductivities and an inlet length to be chosen. To illustrate, the approximate Richmond, New South Wales, Australia, observation was $Q = 24$ m$^3$/d, $W = 4$ m, $L_i = 20$ m, $L_o = 80$ m, $k_i = 2{,}000$ m/d, and $k_o = 30{,}000$ m/d. For $h_o = 0.45$ m, this results in $h_m = 0.48$ m and $h_i = 0.56$ m. Thus, most of the headloss occurs in the inlet zone.

Only a small amount of quantitative information is available for the two-zone model. The Fisher (1990) and Watson and Choate (2001) data suggest a ten-fold decrease in conductivity in the inlet zone (see Figure 2.27, Chapter 2). The size of the inlet zone is approximately 20–30% of the bed length.

There are steps that may be taken to alleviate inlet zone clogging, including limiting the TSS and BOD loading to the

inlet zone and distributing such loads over the entire inlet area (see Figure 21.5). In this design approach, the cross section of the wetland bed is determined by the organic loading and is often overdesigned hydraulically. Wallace and Knight (2006) suggest limiting the cross-sectional BOD loading at the wetland inlet to less than 250 g/m$^2$·d.

### Hydraulics of VF and Biosolids Wetlands

As discussed in Chapter 2, water movement in VF wetlands is a complex mixture of saturated and unsaturated flow processes. However, in terms of laying out the hydraulic profile, a more simplified procedure can be used. Here, we will consider the case of pulse-fed, VF wetlands as commonly employed in Europe. The hydraulic profile consideration then must consider two major points:

- Is there enough driving head to uniformly distribute the flow?
- Is the bed drainable?

If we assume that the VF wetland will be fed via a siphon (one of the simplest methods to gravitationally release large volumes of water in a short time) and that 1 m of driving head is needed to activate the siphon (and overcome frictional losses in the distribution piping), we have established the driving head criteria. If the VF wetland is 1 m deep and must completely drain, that is an additional 1 m of elevation change. So, in this example, if the designer assumes that he/she must reserve 2 m of elevation drop (1 for the siphon and 1 for draining the bed), then the constraints of the hydraulic profile should be met.

As the VF wetland is loaded, the hydraulic conductivity may change over time as a result of the cumulative organic load (Platzer and Mauch, 1997). This effect is dealt with either by using low organic loadings or a load-and-rest operational strategy, as discussed in Chapter 20.

Biosolids wetlands are typically governed by the rate of organic matter application (kg/m$^2$·yr), not the rate of hydraulic application. Underdrain piping is usually overdesigned from a hydraulic standpoint, because one of the major functions of these pipes is to aerate the wetland bed. As a result, most hydraulic issues with biosolids wetlands are likely to be related to pumping sludge to the wetland beds. As the solids content of the biosolids increase, so does the viscosity of the fluid (in a very nonlinear manner). Pumping of viscous fluids is outside of the scope of this book, and designers are encouraged to consult standard wastewater engineering textbooks and the guidance of equipment manufacturers in this regard.

## 21.7 FLOW DISTRIBUTION AND MANAGEMENT

Each type of SSF wetland presents different challenges in terms of flow distribution. Major design issues regarding flow distribution are discussed in this section. The fundamental design goal in all cases is to distribute the flow (and associated organic loading) as uniformly as possible across the wetland.

### HSSF Wetlands

The design goal for influent flow distribution in HSSF wetlands is the ability to distribute the flow as uniformly as possible across the cross section of the wetland bed. This was done initially using castellated or V-notch weirs in the United Kingdom; these proved to be maintenance-intensive devices due to solids accumulations, which interfere with the flow distribution, and are no longer recommended (EC/EWPCA Emergent Hydrophyte Treatment Systems Expert Contact Group and Water Research Centre, 1990).

In North America, the initial means of influent distribution was perforated pipes buried within the inlet region (TVA, 1993). Experience with perforated pipes led to a realization that the organic loading gets concentrated at the distribution orifice and the zone immediately adjacent to the orifice is very susceptible to clogging. This has led to a switch to infiltration chambers (see Figure 21.27) as a means of influent distribution, as the louvered sidewalls of these chambers allow a more uniform distribution of the organic load than perforated pipes (Campbell and Ogden, 1999; Wallace and Knight, 2006).

Pipes with rotating tees or elbows have also been used to distribute flow. The advantage of this method is that the elevation of each discharge point can be individually adjusted to achieve uniform flow distribution. The disadvantage of this method is that the primary effluent is exposed during the treatment process, which would lead to regulatory problems in the United States and northern Europe. As well, such exposed distribution systems can freeze in cold climates.

As knowledge of the certainty of clogging of the influent zone within HSSF wetlands has advanced, there has been a realization that the influent distribution works must have a multi-functional purpose—not only to distribute the influent flow, but to distribute organic load throughout the wetland bed, to delay flooding as long as possible, and also to provide access for bed maintenance activities and influent sampling. One approach has been to arrange the influent distribution into a "T" arrangement to reduce the organic loading at the influent zone, as indicated in Figure 21.13. This header arrangement using infiltration chambers would speculatively be advantageous from a load distribution standpoint, but hydraulic studies (at least in the startup phase) indicate that flow distribution is unequal (Muñoz *et al.*, 2006).

In any case, designers should consider the need for operator access to influent distribution works, in order to visually inspect the influent flow, collect samples, or conduct bed maintenance activities (Figure 21.14).

### VF Wetlands

Most VF wetlands are pulse-fed. The design goal of pulse feeding is to rapidly apply the influent flow such that the bed is rapidly flooded and that uniformity of application is ensured (see Figure 2.33, Chapter 2). Typically, pulse feeding

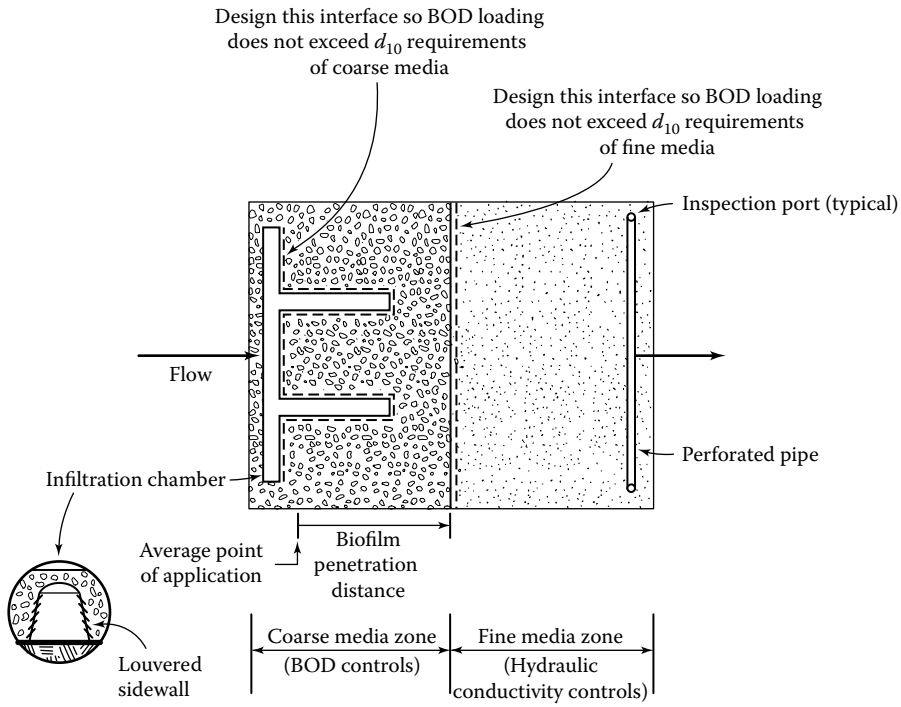



**FIGURE 21.13** HSSF influent header using a "T" layout to reduce the organic cross-sectional loading. (From Wallace and Knight (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements*. Final Report, Project 01-CTS-5, Water Environment Research Foundation (WERF): Alexandria, Virginia. Reprinted with permission.)

is by storing influent water and then rapidly dosing it onto the surface of the bed, either through the use of pumps or, if there is an adequate drop across the site, by gravity drains or siphons (Molle *et al.*, 2004a), as indicated in Figure 21.15.

Siphon dosing can be accomplished by sink and float "swimmer" siphons, as illustrated in Figure 21.15, or by air-bell siphons. Regardless of the method of pulse feeding, the design goal is to rapidly flood the bed in less than 15 min (ÖNORM B 2505, 2005), and the shorter dose intervals in the 5–10 min range are more desirable. Water then infiltrates through the VF wetland bed through the mechanisms of air entrapment and unsaturated flow, as described in Chapter 2.

A variety of piping methods have been used to distribute the influent flow. One of the most common methods is a series of perforated influent distribution pipes placed on the surface of the wetland bed (Cooper, 2003); an example of this approach for a small-scale VF wetland is shown in Figure 21.16.

Other distribution arrangements have been used, such as spray irrigation (similar to Figure 18.23, Chapter 18), but this creates problems under freezing conditions (Figure 21.17). Perforated distribution pipes laid on top of the bed is one of the most common distribution methods. In France, suspended self-draining distribution pipes are common (Molle



**FIGURE 21.14** Influent distribution channel for a tertiary treatment HSSF wetland. This open channel is a preferred design approach of Severn Trent Water in the United Kingdom because it allows easy operator access for cleaning. (Photo courtesy P. Cooper.)



**FIGURE 21.15** VF wetland dosing siphon, Rousillion, France. This is a "swimmer siphon," which alternately sinks and floats during the dosing cycle. At the time of this photo, the dosing chamber was in the "fill" phase, and the siphon assembly was floating.



**FIGURE 21.16** Perforated distribution piping for a pilot-scale VF wetland in Israel. The photo was taken during the feeding phase of the cycle to illustrate the desired effect of temporary bed flooding. Pipe spacing is approximately one meter.



**FIGURE 21.17** Ice formation around an influent spray distributor, Connell, Washington.

*et al.*, 2004a). In addition, very simple point-source distribution methods have also been used in the United States (see Figure 2.33, Chapter 2).

At the present time, there is not a rational design method for the number of square meters per distribution point. If the distribution rate is fast enough (e.g., well in excess of the hydraulic conductivity of the bed media), temporary flooding of the bed will occur, ensuring a good redistribution of the flow and load across the bed surface.

Similarly, there is not a rational design method for the volume per dose. In the original Max Planck Institute Process (MPIP) system of Seidel, the design criterion was a hydraulic loading rate of 1 m/d, and hourly dosing (EC/EWPCA Emergent Hydrophyte Treatment Systems Expert Contact Group and Water Research Centre, 1990), which equates to a flooding depth of approximately 4 cm/dose.

Recirculating VF wetlands have sometimes been employed in North America (Figure 21.7) (Wallace and Hallahan, 2005; García-Pérez *et al.*, 2006). In cold climates, these system often have a distribution system below the surface of the bed to combat freezing (Lemon *et al.*, 1996).

Pumping of the flow is required for flow recirculation. These systems typically employ perforated pipes to distribute the flow, which is much higher than the influent flow rate, in order to achieve the flow/pressure demands for even flow distribution across the VF wetland bed. One hydraulic design procedure is to calculate hydraulics based on the terminal end of the distribution lateral with an assumed residual head (typically 0.3 m). Orifice size, spacing, and pipe diameters are adjusted with the design goal so that there is less than a 10% differential in the orifice discharge rate between the beginning and end of the perforated distribution lateral. These calculations are best performed on a computer spreadsheet.

Water must be stored in a tank or sump for the recirculation pump to draw from. The volume of this storage tank is typically sized to provide between 12 and 36 hours of hydraulic retention time (based on forward flow), with 24 hours being a common design value (Crites and Tchobanoglous, 1998).

### Biosolids Wetlands

The inherent design challenge with biosolids dewatering wetlands is not necessarily distribution of *flow*, but rather, distribution of *solids load*. Because sludge is a viscous material, it will not flow across the bed and distribute itself in the same way as water would. Early sludge bed designs utilized piping along the sides of the wetland cells in conjunction with tees and splash plates to apply sludge to the basins. This approach has largely been abandoned. Modern design practice tends to utilize standpipes (with splash plates) within the wetland bed to more uniformly distribute the influent load. Figure 1.20, Chapter 1, illustrates a biosolid wetland cell being dosed by the standpipe method.

Both the rate of application and the area per distributor are important in order to ensure uniform application of sludge across the wetland cell. The general guidelines are that the sludge application system (typically pumps) must be able to apply 15 cm of liquid over one hour and the sludge distribution standpipes should cover no more than 250 $m^2$ per distributor (Nielsen, 2002). Facilities that do not comply with these design guidelines will result in preferential deposition of sludge near the distributor pipes, resulting in localized overloading and nonuniform application of the sludge layer (Nielsen, 2003).

Organic loadings, feed intervals, and the recommended number of beds for these systems are discussed in Chapter 20.

### Effluent Flow Collection

Collection of the effluent flow in SSF wetlands depends on the process objectives for each type of wetland. For HSSF wetlands, flow collection may be as simple as a perforated pipe with minimal concerns about clogging. For VF and biosolids wetlands, the situation is more complicated, because a second primary purpose of the effluent piping is aeration in addition to flow collection.

For HSSF wetlands, flow collection is relatively straightforward, and can be as simple as a perforated pipe extended across the width of the wetland cell, as indicated in Figure 21.8.

The situation is more complicated for VF and biosolids wetlands, as the collection system is typically overdesigned to allow passive aeration of the wetland bed. For biosolids wetlands, the effluent collection system is typically overdimensioned, with 10-cm perforated pipes spaced every 3 m (maximum) in the wetland bed (Nielsen, 2002). The primary purpose of these pipes is aeration, not drainage, as indicated in Figure 1.7 in Chapter 1.

The section of the drainage/ventilation pipes that passes through the filter layers of the bed should not be perforated to avoid short-circuiting of flow. The elevation of the ventilation cap should be above the maximum flooding level of the bed for the same reason.

### Water Level Control

The operator has control over water levels in HSSF wetlands. The general water level control objectives in HSSF wetlands are the following:

- Maintain the water level below the surface of the wetland bed.
- Manage the water level such that it remains within the root zone of the emergent wetland plants.

This is most easily done with a control structure that has a wide range of adjustability, as indicated in Figure 21.18. For small-scale systems, flexible hoses, as indicated in Figure 23.11, Chapter 23, or rotating elbows are commonly used.

Water level control is essential to the establishment of plants in HSSF wetlands. The minimum-cost alternative to Figure 23.11, Chapter 23, for small-scale wetlands is a fixed standpipe with removable extensions, as implied in Figure 21.18. For large-scale systems, adjustable weir

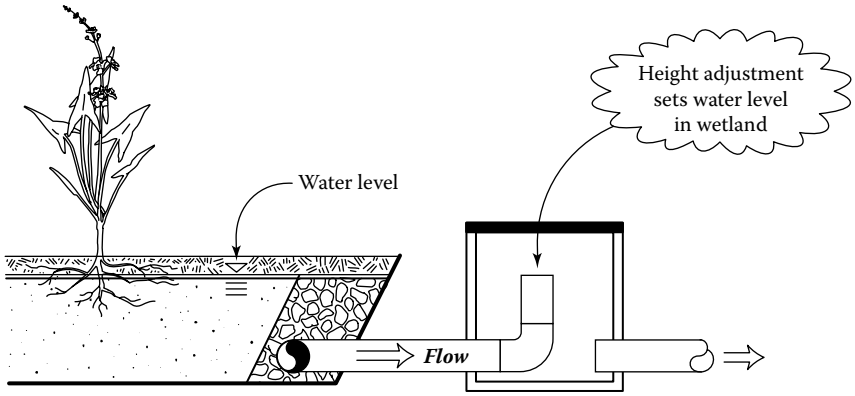



**FIGURE 21.18** Water level control structure for a HSSF wetland. (From Wallace and Knight (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements*. Final Report, Project 01-CTS-5, Water Environment Research Foundation (WERF): Alexandria, Virginia. Reprinted with permission.)

gates are preferred. In the intermediate range, stacking-plate designs such as the one illustrated in Figure 18.27, Chapter 18, are often cost-effective choices, provided that the plates achieve a leak-free seal, which almost invariably means that premanufactured units must be used.

VF and biosolids wetlands are most commonly designed as free-draining systems; water level control may be desirable, but is not mandatory for these systems.

## 21.8 LINER SYSTEMS

Various synthetic and clay liner systems have already been discussed in Chapter 18 for FWS wetlands; the same considerations apply also to SSF wetlands. In North America, one-piece factory-seamed PVC liners (0.76 mm) are commonly used on small wetland basins (<1,000 $m^2$). Larger projects typically use HDPE liners in the 1.1–1.5 mm thickness range; these liners are shipped to the project site in rolls. Individual liner panels are fusion-welded together after placement to create a water-tight liner system. In some regulatory jurisdictions, leak testing of the liner may be required (see Figure 2.7, Chapter 2).

If some infiltration out of the wetland basin is acceptable, then the expense of a synthetic liner system is likely not warranted. If the local soil has sufficient clay content, compaction of this material may adequately seal the basin; amendments such as bentonite or lime can also be used to enhance the sealing properties of the soil.

The choice of a liner system often is a function of the local economy and the materials that are commonly available. For instance, ethylene propylene diene monomer (EPDM) roofing membranes (1.5 mm) and geosynthetic blankets (bentonite enclosed in geotextile) have been successfully used in SSF wetland projects.

### Thermal Considerations

The most obvious example is in cold climates; small and on-site HSSF wetland systems must be designed to operate in subfreezing environments, because winter influent storage is usually not an option.

For HSSF wetlands, cold climate design typically involves burying all the distribution piping and conducting energy balance calculations to determine if an insulating layer of mulch is warranted to prevent freezing of the bed; procedures for this type of thermal analysis are discussed in Chapter 4.

In VF wetlands, ice formation on top of the bed may be an operational concern, and distribution piping must be designed to self-drain between dosing events. In extreme cold climates, the pulse-fed, surface flood method of distribution common in Europe may have to be replaced with designs using buried distribution piping (see Figure 21.7 for example).

For biosolids systems, freezing is generally seen as beneficial because it lyses the cell walls within the sludge material, which aids in dewatering (Reed *et al*., 1988). Some modification of the distribution piping may be needed in very cold climates; for instance, compressed air may be required to purge the dosing standpipes to prevent freezing damage.

## 21.9 SSF WETLAND CONSTRUCTION

Construction of SSF wetlands includes preparing bid documents (if necessary), hiring the services of a construction contractor, and physically building the wetland. This is a critical phase of the implementation process. Because of the nature of SSF wetlands, errors made in the early stages of construction become almost impossible and very expensive to correct after the bed media is installed. As a result, a clear understanding and correct execution of the design documents are essential.

In most areas, there are not a sufficient number of treatment wetland construction projects to allow a contractor to specialize in this type of construction. The construction contractor will likely be unfamiliar with treatment wetlands and often will not have a good understanding of how the system is supposed to function. Contractors on treatment wetland projects will come from backgrounds such as single-home (onsite) septic systems, general civil construction, or

mechanical wastewater treatment plants. Each of these disciplines bring their own history and bias to wetland construction. For instance, septic contractors may understand the purpose of the system as wastewater treatment, but they may be unfamiliar with the level of workmanship and detail necessary to implement a treatment wetland. Civil contractors will likely understand the earthwork and elevation aspects of the project very well, but are unlikely to be familiar with wastewater treatment. Mechanical contractors will be skilled at installing pipes and equipment such as pumps, but are unlikely to understand how a treatment wetland is supposed to "work." In addition to these three disciplines, a specialty subcontractor is often involved to install the liner and do any seaming or associated work. Generally speaking, the construction process is most successful when the wetland designer acts as the bridge to integrate these different construction backgrounds via construction and start-up supervision.

### Elevations and Grading

The basic earthwork process for excavation and grading of SSF wetlands is essentially the same as for FWS basins, which has been previously discussed in Chapter 18. Basic tools, such as Equation 18.1, can be used to estimate earthwork volumes, but for larger projects, 3-dimensional computer modeling is often employed.

The basic objective is to excavate a level basin (or a flat basin with a constant bottom slope). On small projects, this may be done with an excavator (similar to the unit shown in Figure 21.1). Larger projects often use bulldozers or skid-steer loaders to facilitate the basin leveling process.

The setting of the correct elevation of each basin, pipe, and control structure is one of the most fundamental and critical aspects of the construction process, because mistakes made in this regard become almost impossible to fix later on. This usually involves the establishment of a local benchmark of known elevation on the project site. The distance and elevation of the different process components are then measured against this relative benchmark. Generally, these types of measurements are made for two purposes:

1. To confirm that the individual process component is at the correct elevation.
2. To confirm that the earthwork meets the specified degree of flatness (tolerance of at least ±5cm).

Laser levels are increasingly used as a quick and easy means to check elevations, although basic hand levels or water levels can provide acceptable results if used properly. For large projects, the services of a surveyor with specialized equipment are often employed, and the location and elevation of project components are measured using a total station. As-built confirmation of the rough grading is recommended.

### Liners

Various liner systems have been previously discussed in Chapter 18 for FWS wetlands, and those same considerations apply to SSF wetlands. The primary disadvantage of PVC is that seaming material in the field requires the use of adhesive mastic or tape; consequently, most PVC liners are factory-seamed and delivered to the job site. This generally limits their use to small wetland basins (<1,000 $m^2$). Even factory-seamed liners need to be checked in the field, as errors in factory seaming are not uncommon, and pipe penetrations through the liner still need to be sealed, usually with mastic adhesive.

Larger projects will require field seaming; the liner is delivered to the job site in rolls. Having an organized plan for liner placement greatly facilitates field seaming. Figure 21.19 shows one method of liner placement using a tracked excavator and manual labor.

On project sites with stones or other material that may damage the synthetic liner, a protective layer of sand is often



**FIGURE 21.19** Use of a chain and hoist assembly to facilitate liner placement in a HSSF wetland basin, Delft, Minnesota. Sandbags are used to hold the liner pieces in place prior to seaming.

placed on the wetland basin prior to liner placement. After the liner is installed, factor seams must be inspected, and on larger projects, field seaming will commence. Liner seaming is typically done by subcontractors who specialize in this work.

## Berms

The basic elements of berm design have been discussed in Chapter 18 for FWS wetlands. Berm design is generally less complicated for SSF wetlands, because SSF wetlands do not have to contend with wave action or damage by wildlife and require less freeboard. One critical aspect of berm construction is the installation of the liner, with an effective and easy way to anchor the liner edge.

### Earthen Berms

Edge treatment in SSF wetlands is often the earthen berm; the basic cross section for this construction system is shown in Figure 18.18, Chapter 18. Berm construction appears deceptively simple; in reality, the berm/liner interface is a common source of construction mistakes. The design and construction of berm systems must take into account a variety of factors, including

- The angle of repose of the native soils at the project site
- The anchor trench to secure the edge of the liner
- Grading of the outside edges of the berm to prevent stormwater runoff from entering the basin
- The surface treatment of the inside berm face

Wetland designers often assume that a prescriptive side slope (3:1, for example) can be constructed; however, this is dependent on the angle of repose of the soil. Designers should verify that the specified side slope is actually constructable at the project site.

Anchoring the liner edge is often a source of construction problems. Liner placement is greatly facilitated if the berm face is smooth, uniform, and level. Construction of the anchor trench is best done with a chain trencher. Figure 18.24, Chapter 18, is an example of good grading and liner installation. This level of detail in site grading requires a skilled heavy-equipment operator and cohesive soils. If the berm is not uniform, or poorly graded, liner installation is much more complicated and prone to construction errors. Figure 21.20 shows a poor job of berm grading that has resulted in an uneven top elevation and numerous bulges in the liner. The exterior grading outside the berm is insufficient to protect the wetland from the intrusion of stormwater runoff and associated sediment.

In some instances, it may be acceptable to leave the liner exposed on the inside of the berm face, or the liner may be covered with a protective fabric, as illustrated in Figure 21.21. In other applications, it may be desirable to provide more protection to the liner. This is commonly done through the use of rip-rap, as indicated in Figure 21.22.



**FIGURE 21.20** Example of poor berm finishing, liner installation, and site grading. Stormwater runoff is carrying sediment into the HSSF wetland basin.



**FIGURE 21.21** VF wetland in St. Thomé, France. Inside face of the berm consists of liner covered with a protective fabric. The pipes against the berm face are drainage pipes with ventilation caps.

If a decision is made to leave the liner exposed, textured liner material should be used on the berm face. Standard liner materials are very slick when wet (Figure 21.23) and can present a considerable safety hazard; it is not possible to climb out of a wetland basin under these conditions if the berm slope is at 3:1 or steeper. That includes animals, such as dogs, cats, etc.

### Vertical Sidewalls

An alternative to earthen berms is the use of vertical sidewalls. Sidewalls require additional material to construct and are quite expensive, but offer the advantages of greater space efficiency and much easier liner installation. Vertical sidewalls can be constructed out of plywood, concrete block, or cast-in-place concrete. Figure 21.24 shows an example of plywood construction. This type of wall system is also shown in Figures 21.1, 21.26, and 21.28. The plywood wall system is not structurally free standing; both the outside and inside of the wall must be filled in equal lifts to hold the wall in place. It is anticipated that over time the wood will decay, leaving the liner in place.

In many areas of the world, the concrete block is a common and relatively inexpensive construction material when vertical sidewalls are advantageous. The interior cavities of the concrete block are typically poured with concrete, and reinforcing steel is added for structural strength; Figure 21.25 is an example of this type of sidewall construction. The third alternative is cast-in-place concrete walls (see Figure 1.15, Chapter 1). This is the most expensive method of construction, but offers



**FIGURE 21.22** HSSF wetland divider berm armored with rip-rap, Albuquerque, New Mexico.



**FIGURE 21.23** HSSF wetland berm in southern Israel. A light rain has made this surface extremely slippery because smooth high-density polyethylene (HDPE) was used on the berm face.



**FIGURE 21.24** Example of vertical sidewall construction using plywood, Prinsburg, Minnesota. The inside wall face has a 5-cm-thick extruded polystyrene (XPS) board for thermal insulation and also to minimize wear on the liner.



**FIGURE 21.25** HSSF wetland with concrete block sidewalls, Akumal, Mexico.

the greatest structural strength. This method of construction is sometimes used for biosolids wetlands, because, when the basin is unloaded, removal vehicles such as skid-steer loaders can push directly against the concrete wall.

### Influent and Effluent Piping

After the liner installation, the next major step in the construction process is the installation of the influent distribution piping and effluent collection piping. For VF wetlands, the drainage pipes are installed prior to placement of the various layers of bed media (Figure 21.26) and the influent distribution piping is typically installed on top of the bed (see Figure 1.14, Chapter 1, for an example), although biosolids wetlands that use the riser pipe approach shown in Figure 1.20, Chapter 1, may have the influent distribution pipes installed prior to placement of the bed media. HSSF wetlands have the influent and effluent piping installed at either end of the bed prior to placement of the bed media (Figure 21.27).

### Bed Media Placement

Placement of the bed media is one of the most critical aspects of SSF wetland construction. This is especially true for VF and biosolids systems, which rely on stacked layers of filter material that often has to meet a fairly tight grain-size distribution. One of the major decisions during SSF wetland construction is how the bed material will be placed into the wetland cell. From the perspective of the construction contractor, there are three major choices, each of which has pros and cons:

- Utilization of a general construction machine, such as a tracked excavator
- Allowing equipment within the wetland cell to spread bed material (typically skid-steer loaders or small bulldozers)
- Use of a specialty device, such as an extendable conveyor, to spread the bed material

The lowest-cost alternative for most construction contractors will be to use a general purpose machine such as an excavator (see Figure 21.1). This type of equipment can (1) excavate the basin, (2) facilitate liner placement, and (3) place the bed material once the liner is installed. The drawback to this approach is that most excavators have a reach less than 10 m, which implies that 20 m is about the maximum width the wetland cell that can be constructed with this method. Also, the use of such large excavators for media placement will likely impose greater construction-access separation issues on the overall wetland project.

If direct placement of the bed media by an external piece of equipment is not feasible, the next lowest-cost alternative is to allow tracked vehicles within the wetland cell. In HSSF wetland cells, this is somewhat easier, as the placed bed media can be "pushed" forward in a manner that the construction equipment does not have to travel directly on the liner.



**FIGURE 21.26** VF wetland perforated drainage pipes and nonperforated ventilation risers prior to bed media placement, Lake Elmo, Minnesota. Apparatus in the rear corner of the wetland cell is for leak testing of the liner (see Figure 2.7).



**FIGURE 21.27** HSSF wetland influent distribution chamber, Cambridge-Isanti School District, Minnesota. The white PVC pipe inside the chamber is perforated to distribute the flow from a dosing pump. The white PVC pipe to the right is an aeration header.

The situation is more complicated for VF and biosolids wetlands; if construction vehicles are allowed within the basin for these systems, they must travel directly over the effluent collection piping (Figure 21.28). Even if the ground-bearing pressure of the tracked vehicle is acceptable, wetland designers should keep in mind that the shear forces associated with turning tracked vehicles can easily break the underdrain piping, and considerable care must be taken in placing bed media in this manner.

There are situations where the wetland basin will be too large (or too small) to allow cost-effective access by construction vehicles such as those illustrated in Figure 21.28. One solution is an extendable conveyor system that can minimize the need for secondary releveling.

### Installation of Control Structures

For small SSF systems, control structures are likely to be simple standpipe units (see Figure 23.11, Chapter 23, for example) or stacking-plate designs (see Figure 18.27, Chapter 18).

The primary design challenge is that the control structures must be designed in a way that allows for efficient and safe operator access.



**FIGURE 21.28** Example of drainage material being placed over effluent collection piping in a VF wetland in Lake Elmo, Minnesota. Gravel is being placed by the excavator outside the basin; secondary re-leveling is done by the skid-steer loader inside the basin.

In the United States, entry into confined spaces is subject to considerable federal regulatory requirements, which, at a minimum, requires the testing of the ambient atmosphere (to check for breathable or explosive atmospheres). If wetland designers can create "open access" controls, this is strongly encouraged because it will greatly simplify operator use of the facility.

## 21.10 COMMISSIONING AND START-UP

One of the most important aspects of the start-up and commissioning of SSF wetland systems is vegetation establishment. Plant establishment is a process of planting the wetland vegetation combined with appropriate water level control. Various methods of plant establishment are discussed in this section.

### Plant Selection

The selection of plant species in SSF wetlands is often a product of cultural and regulatory constraints. SSF wetlands tend to be heavily loaded compared to "agronomic" FWS wetlands, so the influence of the plant geochemical cycle often is not as important. Also, the aboveground plant canopy does not interact with the water column in the same way. In FWS wetlands, the emergent plant detritus falls into the water column, providing surface area for microbial biofilms and a degradable carbon source for treatment processes such as denitrification (see Chapter 9). At the present time, there is not sufficient knowledge to recommend a single species as being able to conclusively improve treatment over a mixed plant community.

In Europe, common reed (*Phragmites* spp.) is used almost exclusively as the plant of choice and often forms monocultures in SSF treatment wetlands; HSSF systems in the Czech Republic often use a combination of *Phragmites* and reed canary grass (*Phalaris arundinacea*) (Vymazal, 1998). Both species are considered invasive in the United States, and their use is often discouraged by natural-resource managers.

In North America, the primary emphasis in HSSF wetlands has been on *Typha* spp., *Schoenoplectus* spp., and *Carex* spp. There is an emerging body of information showing that the root zone oxygen transfer of these species is different (Stein *et al*., 2006b), especially in the dormant season, but to date there is insufficient knowledge to translate this into design guidance.

For single-home SSF systems, the aesthetic benefits of the plant community is often an important design consideration. Plants such as *Asclepias incarnata*, *Solidago rigida*, *Canna* spp., *Iris* spp., and *Cyperus alterniflorus* have been used successfully in North America (Wallace and Knight, 2006) depending on the climate, although the aesthetic quality of the plant community typically requires some "gardening" from the system owner to maintain the best visual presentation.

In tropical countries, locally available species of *Phragmites* and *Typha* have been the most common choice to date.

Cold climates offer a different set of selection choices. If the wetland is insulated with a mulch layer, this creates a permanent unsaturated zone that shifts the competitive advantage away from the obligate wetland plants towards facultative plants (Chapter 2). *Schoenoplectus* (*Scirpus*) species such as *Scirpus atrovirens* and *Scirpus fluviatilis* have proven to be good choices in the cold climates of North America (Wallace *et al*., 2001).

The selection of a plant species (or suite of plant species) is generally a function of two factors:

1. The degree of rhizome spread and, hence, the ability to achieve plant canopy and crowd out unwanted invasive species
2. The development of more belowground root biomass and depth of root penetration, which benefits wastewater treatment

Qualitative observations in Minnesota indicate that *Phalaris* is very effective in Factor #1, but poor in #2; hence, project specifications often include a one- to two-year maintenance contract to promote the development of a *Schoenoplectus* plant community over the initial colonization pressure of *Phalaris*. Contact herbicides are routinely used in this regard.

The cost of the initial plantings of SSF wetland systems is discussed in Chapter 23.

### Vegetation Establishment

#### Planting Wetland Vegetation

Planting in SSF wetland treatment systems is done by hand. Rhizomes and similar rootstock are planted directly into the SSF bed media. The water level is typically lowered temporarily to facilitate planting. After planting, the water level is immediately raised so that the rhizomes do not dry out.

For mulch-insulated wetlands, the procedure is slightly different. After planting, the water level is typically raised above the top of the SSF bed so as to subirrigate the mulch layer. If potted or bare-root plants are used, the mulch is firmly packed around the plant for stability, protection, and to ensure good water contact with the roots, as shown in Figure 21.29.

For soil-based SSF systems, extreme care must be taken to avoid compaction of the soil; in these applications, workers typically use pieces of plywood to distribute their weight and minimize impact to the media (Kickuth, 2002). Owing to the repetitive nature of the work, laborers quickly adapt tools to the task. One of the most commonly used tools for planting is a military entrenching tool.

#### Successful Plant Establishment

For plant establishment in a treatment wetland to be successful, the plant material (seed, tuber, rhizome, pot, etc.) must

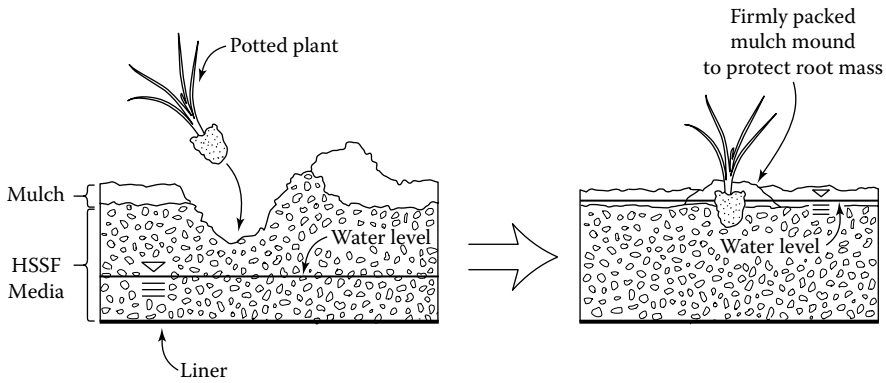



**FIGURE 21.29** Potted-plant installation procedure for a mulch-insulated HSSF wetland. (From Wallace and Knight (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements*. Final Report, Project 01-CTS-5, Water Environment Research Foundation (WERF): Alexandria, Virginia. Reprinted with permission.)

be viable at the time of planting. This may require provisions such as the shading and watering of the planting stock at the project site so that it does not dry out and die before it can be planted. After planting, water level management during the start-up phase must be compatible with the needs of the plants.

Wetland plants are adapted to a hydric environment and have very little drought tolerance. If the plants are allowed to dry out during the establishment phase (where there is very little root network available from which the plants can draw water), widespread mortality can occur.

For SSF wetlands, the water level should be raised up to the level of the bed for nonmulched systems and about 1 cm above the gravel bed (so that the mulch is subirrigated) for mulched systems. The goal is to provide a continuous flooding of the plant root zone during establishment. Figure 21.30 shows a HSSF wetland with *Solidago rigida* during the plant establishment phase and the resulting established plant community at the end of one growing season (Cross, 2001).

Design specifications for vegetation establishment in constructed wetlands should clearly delegate responsibility for plant maintenance from the time of initial planting through plant establishment. Successful establishment requires periodic inspections to document water level, plant survival, and plant growth. The frequency of these inspections is project-specific, but must be often enough to deal proactively with potential problems (see Figure 18.36, Chapter 18).

HSSF wetland beds are often filled with clean water before planting. As this clean water is displaced by the influent flow, pollutant loads gradually increase. This is a preferred



(a) (b)

**FIGURE 21.30** Newly planted (a) and established (b) mulch-insulated HSSF wetland in Anamosa, Iowa. (Photos courtesy C. Cross.)

approach, as it does not expose workers to potentially harmful substances in the wastewater during planting and provides a time window for the vegetation to adapt to the pollutant loadings.

VF wetlands are designed to operate as free draining systems and, as a result, may require a supplemental water supply during start-up to assist in vegetation establishment. The dosing interval must be sufficient to maintain moisture in the root zone to keep the plants alive, typically, once or twice per day, at a minimum. The use of portable sprinklers to spray irrigate water during the establishment phase has been used in recirculating VF systems because newly installed plant material typically cannot reach the buried distribution piping.

For biosolids systems, it is desirable to apply a thin layer (5 cm) of sludge across the surface of the basins (Nielsen, 2003).

Planting densities vary based on the plant spreading rate, desired time to canopy coverage, and growing season. For cool temperate climates, planting densities between 2 and 6 plants per square meter are recommended (ATV, 1998; Nielsen, 2003); this will typically achieve complete plant coverage in approximately two growing seasons, depending on the climate. Expansion of the plant coverage is primarily through lateral runners (see Figure 3.12 for example).

## SUMMARY

The physical design of SSF wetlands requires numerous decisions concerning siting and layout. The compartmentalization and number of flowpaths as well as the aspect ratios of the cells must be determined. Because of competing objectives, designers must often compromise between the number of cells, flow paths, aspect ratio, bed depth, and the size of the bed media.

VF and biosolids wetland systems will require the selected loading rate that can be matched with a load-and-rest regime that allows the system to function without short-term clogging problems.

Based on the information that is currently available, it appears that clogging of HSSF wetlands can be delayed by minimizing cross-sectional loading. However, there is no information to indicate that clogging of HSSF wetlands will not occur over the long term, and it is likely that HSSF wetland beds will require periodic maintenance about every ten years to maintain hydraulic conductivity, especially in the inlet zone. Although this chapter explores the design and construction of SSF treatment wetlands, the guidelines presented herein are by no means a definitive checklist for implementation of treatment wetland projects, as the physical, climatic, and political landscape of each project will be unique.

# 22 Management, Operations, and Maintenance

As discussed in the preceding chapters, most treatment wetlands are designed and constructed to require infrequent operational control or maintenance. Through conservative design and simple, low-maintenance mechanical controls, many free-water surface (FWS) wetland treatment systems will experience minimal ecological changes and will continue to meet final effluent limits for a long period of time. The situation is more problematic for horizontal subsurface flow (HSSF) wetlands, which will require periodic bed maintenance throughout the lifetime of the system, and for vertical flow (VF) wetlands, which are dependent on loading-and-resting regimes to maintain hydraulic conductivity, unless lightly loaded.

Monitoring and adjustment of flows, water levels, water quality, and biological parameters are the principal day-to-day activities required to achieve successful performance of these low-tech treatment wetlands. Other operations and maintenance activities in treatment wetlands, such as repair of pumps, levees, and water control structures; vegetation management; pest control; and removal of accumulated mineral solids, typically must be attended to at much less frequent intervals.

This chapter provides guidance on the appropriate level of operational monitoring necessary for the control of wetland treatment systems. Typically, these are natural systems with low hydraulic and constituent loadings. Ecological integrity therefore requires minimal attention, unless a particular preconceived plant mix has been mandated. Hydraulic balance and hydropattern control may require infrequent attention. Compliance monitoring is driven by the permit conditions, but often a modest amount of extra monitoring is useful in optimizing operations. Emphasis should be placed on the avoidance of unnecessary operational and maintenance activities, including the management of nuisance species when this achieves no measurable compliance goals.

## 22.1 START-UP

It should be fairly obvious that a newly built wetland is not in a condition to perform the functions that will ultimately prevail in the system. Yet, there have been many misunderstandings resulting from start-up conditions that do not meet long-term expectations. Start-up transients are not necessarily brief, because it can take considerable time for the wetland biology to develop. Several processes occur during start-up, importantly including:

- Plants increase in density and areal coverage.
- Senesced plant parts form a litter layer.
- Newly placed soils either release constituents if initially loaded or absorb constituents until they are fully loaded.

Because plants form part of the "biomachine" that accretes phosphorus and other chemicals, and because plant detritus forms the sites for periphyton and bacterial assemblages, it is expected that removal will be enhanced by grow-in. One pattern of start-up is typified by the rapid vegetation of a bare soil wetland after an initial planting. If that planting is sparse, the start-up period will be characterized by the time for plant fill-in plus the time for litter development. That can be relatively brief, especially in warm climates. At the Orlando Easterly Wetland in Florida, the start-up duration was approximately 24 months, during which considerable change in the ecosystem took place. Figure 10.21 shows two of the principal variables: vegetation density and phosphorus rate constant. Note the close parallel between the amount of vegetation and the rate constant during the first two years. After that period, the vegetation density levels off, and the rate constant drops to its long-term average value of 13.5 m/yr. The phosphorus requirement for building the new standing crop of biomass leads to a peak uptake rate constant, that is, roughly double the long-term average value.

An interesting feature of the unvegetated wetlands is the absence of a carbon source to fuel the conventional heterotrophic denitrification. That process would conventionally require a supply of carbon of approximately equal mass to that of nitrogen lost (Ingersoll and Baker, 1998). A newly constructed wetland does not have the carbon source created by litter decomposition, unless the system is mulched initially with carbonaceous material.

Algae are an opportunistic first-colonizer in new wetlands, because the canopy is not yet extensive, and sunlight is available, perhaps accompanied by copious nutrient supplies (see Figure 3.1). Although this initial vegetative cover develops rapidly, it utilizes only a small fraction of the nutrients in the system and temporarily creates a pond environment.

### Antecedent Conditions

The initial soils and sediments may contain extra nutrients or contaminants, or they may possess an initially high sorption capacity. The antecedent load of pollutants should be assessed to determine if there is treatment liability that must be accommodated before the incoming water can be treated. The same processes that will eventually provide treatment

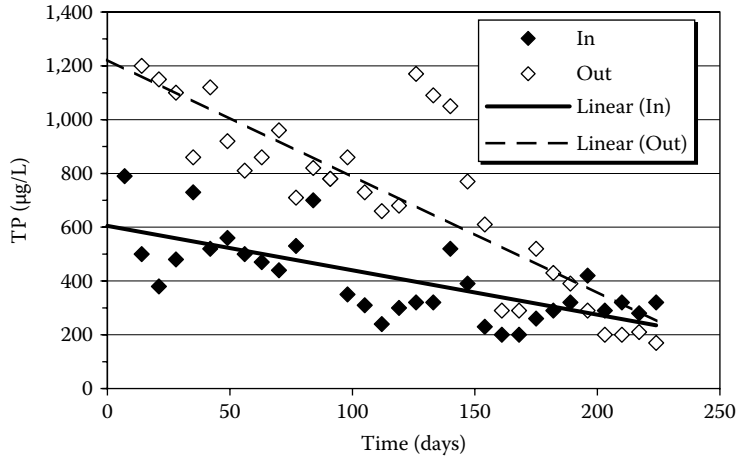



**FIGURE 22.1** Start-up of a FWS wetland near Orlando, Florida. The antecedent soils were laden with labile phosphorus, which was released to overlying water during start-up. This initial load was absorbed and flushed over the first 200 days of operation.

for incoming water can also remove and immobilize such antecedent pollutants. However, if the water flow is immediately commenced, then the wetland can temporarily serve as a "source" of contaminants. If unanticipated, such releases can create regulatory problems. For instance, a wetland treatment system in Orlando, Florida, released phosphorus and nitrogen from antecedent land uses and conditions, during a start-up in the flow through mode (Figure 22.1). Despite the improvements to performance that ultimately resulted in phosphorus removal, the regulatory expectation was that removal should occur "from day one." Very large fines were levied just as the wetland came out of start-up, causing closure of the facility.

The keys to avoiding such start-up problems are an understanding of potential treatment liability and a plan to avoid discharges until the wetland has "settled down" to a post-start-up condition. However, it is difficult to forecast the direction or magnitude of potential start-up releases. Accordingly, a regulatory framework has been developed in South Florida for start-up of FWS wetland systems. Three periods in the (early) life of a treatment wetland are identified:

- *Start-up phase.* The permittee shall manage water depths in the treatment cells to facilitate the recruitment of marsh vegetation in accordance with an Operations Plan, which may include recirculating waters within the system. The start-up test for an individual flow-way begins when the previously mentioned samples demonstrate, over a four-week period, a net reduction in the target pollutant (phosphorus for the STAs) occurs. Discharge/flow through operations, from an individual flow-way that has passed the start-up test, may commence once documentation and all supporting data and analyses are submitted. The intent of this phase is to prevent flow through operation until it can be demonstrated that the project is producing net benefit.
- *Stabilization phase.* Once flow through discharges from a flow-way begin, the permittee shall initiate water quality monitoring for that flow-way consistent with the monitoring program. Performance during this period is not expected to be optimal because the wetland is not yet fully developed. The stabilization test is met when the long-term criteria for the facility are met, i.e., the concentration limits are being met. If, after the two years of full flow through operation, the facility has not met this stabilization test, the permittee is to submit a report evaluating reasons for not meeting the stabilization test.
- *Routine operations phase.* During the routine operations phase, the wetland is deemed in compliance if the permit effluent limitation is being achieved.

In general, the South Florida projects have achieved start-up and stabilization in periods ranging from a few weeks to more than two years, which is typical of warm climates. In cold climates, a grow-in period of approximately two growing seasons may be anticipated, depending of the planting density and rate of vegetation propagation (see Figure 3.12).

It is generally a bad idea to initiate the development of a wetland with large flows of high-strength wastewater. Newly planted vegetation is susceptible to destruction if exposed to high-oxygen stress or high concentrations of some chemicals such as sulfur compounds. Therefore, a "ramp up" from fairly good quality water to the strong influent is often used. For instance, the Saginaw, Michigan landfill leachate treatment wetland anticipated source water of several hundred mg/L of ammonia nitrogen. The wetland was developed, including litter and plants, using rainwater and a potable water supplements. The system was placed on large recycle and the introduction of leachate commenced. The raw leachate was thus diluted with treated water. This start-up strategy was successful. As a counterexample, the potato water

treatment wetland at Connell, Washington, was started-up with an abrupt transition from clean irrigation water to full-strength process water (anaerobic). Large populations of purple sulfur bacteria developed, and the wetland developed sulfides in excess of 40 mg/L (Burgoon *et al*., 1999). Very strong odors were generated and inlet vegetation suffered. These were fortunately only start-up problems, which have not continued beyond the commission phase.

## Vegetation Start-Up

Design specifications for vegetation establishment in constructed wetlands should clearly delegate responsibility for plant maintenance from the time of planting until system start-up. This task may be the responsibility of the contractor or planting subcontractor, the engineer, the owner, or some combination of these parties and should be carefully described in the planting specifications (see Table 22.1). The key requirements for healthy wetland plant propagules to succeed are water, soil, nutrients, and light. The first two components must be controlled to some extent by the engineer and the contractor, and nature generally provides the other requirements for plant growth. The establishment of vegetation is usually a task of a planting contractor and not the facility operator.

Water level control is of primary importance during the plant establishment phase. Therefore, clear lines of responsibility need to be spelled out: who is required to supply start-up irrigation water, and who is responsible for watering the new plants? Failure to supply water in appropriate amounts can result in planting failure (see Figure 18.35). Generally, there are three phases to water level adjustment during this phase:

- *Phase 1:* Decreasing the water level to expose the rooting media (mud flat) for the initial planting or for replanting.
- *Phase 2:* A gradual increase of the water depth as emergent plants become established. The emergent shoots should be above the water surface at all times to give the plants access to sunlight and oxygen. Water level adjustments should be less than 2 cm/d during this time. Typically, the higher the oxygen stress on the plants (high organic and/or nutrient loads), the smaller the incremental adjustment should be.
- *Phase 3:* Setting the water level at the design depth after plants are established.

Basic procedures for establishment of wetland plants for SSF wetlands have already been provided in Chapter 21.

If there is significant failure during the initial planting, these steps may need to be repeated. Initial plant inspection should examine the viability of the planted propagules, whether seeds, seedlings, or field-harvested mature plants. If plants were planted in rows on specified centers, the surviving plants in a subset of these rows can be counted to determine the survival rate. When counting surviving plants, each of the original seedlings or plant clumps should only be counted once, and any daughter plants that may have arisen from rhizomes should not be counted. If seeds were used, then random plots can be used to estimate germination success. Planting success can be compared to planting specifications based on these plant counts. In some cases, a contractor may have to fill in some areas with new plants to comply with the requirements of the planting specifications.

Inexperienced contractors and inexperienced engineers writing specifications and providing construction supervision have often killed their wetland plants through insufficient soil moisture, excessive water depths, inadequate soil preparation, damaged plant material, inadequate plant spacing, inappropriate planting methods, and bad timing. Success is highly probable as long as a contractor understands the growth requirements of the desired wetland plants and uses information available to most horticulturists.

Once emergent plants are well established, replanting is usually not necessary unless the system is overwhelmed by

**TABLE 22.1**
**Example of Contractual Responsibilities for Plant Establishment**

**3.2 Maintenance of Stand**

**A. Maintenance**

The planting contractor shall make arrangements and bear all costs of providing adequate water for initial planting and plant maintenance. Treatment effluent from the existing facultative lagoon can be used, if desired by the contractor, as long as off-site runoff is controlled. The planting contractor shall be responsible for maintaining the plants to ensure optimum growing conditions until accepted by the Owner. Initial water depth in the wetlands shall be increased gradually from 0 inches to maximum depth in 2-inch increments over a period of two months following completion of planting. All planting shall be done in moist soil without standing water. Maintenance may include watering or dewatering of the wetlands cells to ensure optimum growing conditions.

**B. Inspection**

Sixty days after the completion of the planting, the Owner or Engineer will make an inspection to determine if satisfactory stands of cattail and bulrush, have been produced. A satisfactory stand is defined as one in which: (1) spacing between planted seedlings averages 3 feet or less; (2) the seedling survival averages at least 80 percent; and (3) no areas of greater than 100 contiguous square feet with a seedling survival rate of less than 50 percent. If satisfactory stands have not been established, another inspection will be made after the planting contractor has corrected any deficiencies and provided the Owner or Engineer with written notice that the stands are ready for inspection.

rodents or waterfowl, which can destroy the emergent plant community if left unchecked. In cold climates, grazing pressure in the winter and early spring is a significant operational concern during the plant establishment phase, and wetland designers should anticipate replanting and/or exclusion activities (Wallace *et al.*, 2001).

## 22.2 MONITORING

All wetland treatment systems should be monitored for at least inflow and outflow water quality, water levels, and indicators of biological condition. Following proper design and construction, monitoring is the next most important factor in successful operation of treatment wetlands. Monitoring information is essential to the ability to anticipate the need for operational changes and should be collected accurately and consistently, and frequently reviewed by a knowledgeable operator. Changing the character or function of the wetland ecosystem takes considerable time and operator investment. The appearance of the wetland is not necessarily representative of what it is doing for water quality, because many wetland functions do not tie strongly to the abovewater canopy. Consequently, it is water quality and quantity data that are the primary indicators of wetland operational status. There are few controls for Type *A* wetlands (the ones that are passively designed) but more for Type *B* systems (the ones with added controllable features). Water level control is possible for all types. Added possibilities may include flow control (fill and drain), recycle, bypass, added streams with supplemental constituents, and aeration, as discussed in Chapter 24. Alteration of the ecosystem is also possible for all types, but usually is a drastic step, to be avoided if possible. Dredging of sediments or soils, bed media replacement, new plantings, or removal of plants by mechanical or chemical means can all be accomplished but at considerable expenditure of time, money, and loss of treatment potential. In all cases, appropriate diagnostic information is a prerequisite to initiating maintenance or control steps.

### Water Quality

Continuous flow wetlands will require compliance monitoring of a specified list of chemical constituents and inflow rate, because most operate under some kind of discharge permit. The sampling locations can vary from project to project, and usually include some or all of the wetland inflow and outflow points. However, some projects may require internal monitoring points in the treatment wetland, and in some cases, monitoring of the receiving water body. The locations, types, and frequency of compliance sampling are spelled out in the individual permits, as are the averaging periods and limits for the various regulated constituents. Laboratory protocols for chemical analysis are also officially prescribed and include a variety of established methods, as defined in Standard Methods (APHA, 2005) or by U.S. EPA, for example.

However, the methods for acquiring wetland water samples are neither well defined nor standard. There is no apparent difficulty with obtaining samples at inlet and outlet structures, yet there can be serious anomalies related to sampling methodology. For instance, either autosamplers or grab sampling may be employed. The autosampler can be set to flow-proportion, and thus (hopefully) produce flow-weighted concentrations for use in mass balances. It is instructive to see what happens when both procedures are used over a long period of record (Figure 22.2). Not only do the sampling methods yield much different results, there is an apparent bias of the autosamplers to give higher results. Speculatively, the reason is that autosampler intake tubing may "suck mud" if the intake strays from its optimal position.

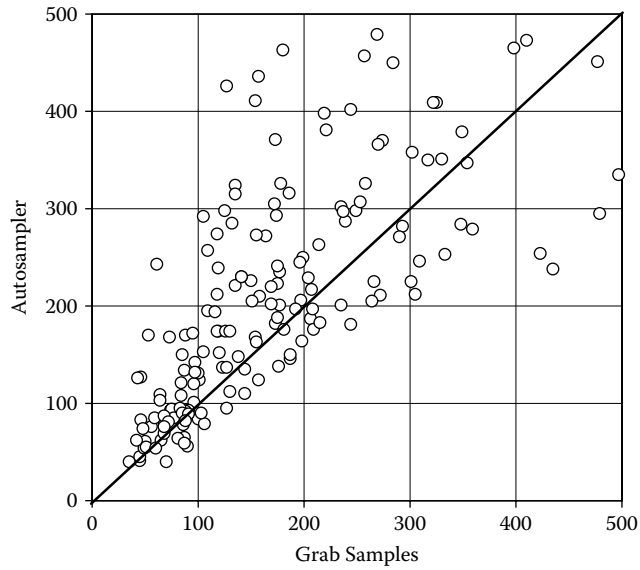



**FIGURE 22.2** Autosampler versus grab samples for total phosphorus for one of the inlet structures of STA2 of the Everglades Protection project. Data span eight years of operation. Note that the autosampler results are generally higher (mean = 223 μgP/L) than grab sample results (mean = 149 μgP/L).

Acquiring representative internal wetland water samples in a FWS environment is very difficult. Floating materials, such as *Lemna*, can often be excluded by careful sampling. Water depths may be quite shallow, and there can be large amounts of readily suspended materials in the vicinity (see Figure 7.9). The inclusion of particulates in a dipped sample will not be representative of the particulates that move with water in the undisturbed environment. The local velocities created by the sampling activity resuspend an abnormally high amount of total suspended solids (TSS). That solid material can bias the lab analyses for total amounts of contaminants such as total phosphorus or total Kjeldahl nitrogen (TKN). The intense vertical gradients in water quality near the wetland bottom can also create anomalous results. In very shallow water, the sample can contain some or all of pore water, which usually has much different (higher) concentrations from surface waters (see Figure 10.5, for example). All of these factors create a need to avoid disturbance in the act of sampling and to avoid very shallow depths that may involve pore water. In the various Everglades sampling



**FIGURE 22.3** Effluent grab sampling using a portable peristaltic pump at a HSSF wetland in Scandia, Minnesota. Considerable care must be taken with the sample tubing to obtain representative samples.

activities, depths less than 20 cm are not bottle-dip sampled for these reasons. However, carefully aspirated samples may be acquired at smaller depths.

Similar problems exist in sampling SSF wetlands, especially because dedicated sample ports must be used to extract *in situ* samples, which may or may not representative of the flow path. Figure 7.1 illustrates a typical sample port assembly for SSF wetland systems.

Peristaltic pumps are often a preferred sampling device for SSF systems, as shown in Figure 22.3. Considerable care must be taken in the placement of the sampler tubing to obtain a representative sample; poor tubing placement will disturb biofilms and/or suck sludge from the bottom of control structures or sample ports. Dedicated sample tubing that is permanently installed is often used at facilities where routine sampling is conducted.

### Sampling and Analyses in Support of Mass Balances

Compliance monitoring may not be enough to gain an understanding of wetland function, because it is focused upon measuring the variables that may affect the receiving waters, and is commonly restricted to outflows. A fuller understanding of wetland performance rests upon constituent mass balances, which in turn depend upon the water mass balance for the wetland.

### Water Flows

The water budget has been described in detail in Chapter 2. The principal inflow is usually the water to be treated, which is augmented by rainfall, evapotranspiration, and seepage before forming the wetland outflow. Rainfall records may be kept at the site or procured from a nearby weather station. Evapotranspiration may be derived from weather bureau pan evaporation or from governmental irrigation services, as discussed in Chapter 4. Therefore, the primary monitoring requirements are usually the wetland inflow rate and outflow rate. Both are required if there is significant seepage, so that the seepage may be estimated by difference.

Flow measurement can be accomplished using any one of several techniques. For pumped inflows, either pump-hours plus the pump-discharge curve may be used or a turbine meter can be installed. For small flows, a calibrated weir, possibly with continuously recording head, is a common choice (Figure 22.4). Most large inlet and outlet devices that operate with a measurable head loss can be calibrated to determine flow from headwater and tailwater elevations. For large inflows without a calibratable structure, devices such as Doppler flow meters or ultrasonic velocity meters have been employed. It is noteworthy that virtually every type of



**FIGURE 22.4** A V-notch weir, with recorded head, is a user-friendly method of measuring small inflows and outflows. Photo from the Duck Spring wetland system near Alcoa, Tennessee.



**FIGURE 22.5** This small HS flume is used in conjunction with an ultrasonic head to record influent flows for a HSSF wetland in Delft, Minnesota.

flow measurement relies upon clean conditions and periodic recalibration. Any of the measurements are prone to being somewhat in error with field accuracies on the order of ±15%. For any of the devices that include sensors and data logging, frequent manual cross-checks are recommended.

Low flows in small systems will be zero (no flow). This has several design implications. Most notably, there will be an insufficient flow to create scouring velocities in household plumbing and gravity sewer pipes. During intermittent flow, solids will "hop" down the pipe, moving only when the flow and depth exceed certain limits. Solid particles accumulate into dams, partially obstructing the invert of the pipe. When flow pulses occur, water accumulates behind the dam and pushes the solids along the pipe (Littlewood and Butler, 2003). This "sliding dam" mechanism of movement is the dominant form of solids transport in small treatment systems.

Conventional weirs or upflow-splitting devices are generally not used as primary flow control elements for small wetland systems unless the solid material has been removed in a primary treatment process. Due to the low flow, toilet paper and other solids will inevitably get caught on one of the overflow weirs because there is insufficient velocity to remove the material. All flow will go to the other outlet, until it, too, clogs with solid material. Flow will switch back and forth between the outlets in an unpredictable manner and create problems for the operations staff.

Accurate flow measurement for a small collection network is difficult to obtain for many reasons. Use of an overflow weir as the primary control element (combined with an ultrasonic head or other sensing device) can present an ongoing problem for maintenance personnel due to solids accumulation. Magnetic flow meters have a minimum flow velocity threshold of about 6 cm/s for accurate flow determination. For small-scale applications, this generally requires "necking down" the piping to 25 or 50 mm to get the necessary low-flow accuracy. This is generally not a viable option unless solids have been removed from the wastewater in a settling tank.

Small flumes, such as H or HS types (as shown in Figure 22.5), offer acceptable accuracy at low-flow rates (Grant and Dawson, 1997) but may not achieve scour velocity, leaving the flumes susceptible to solids accumulation. If a flume is used after a solids removal device (such as a settling tank), the velocity at low flow will allow the growth of a biofilm on the flume surface. For this reason, bubbler units are not recommended as a sensing device because biofilm growth will rapidly plug the air tube. Noncontact devices, such as ultrasonic heads, are recommended for use with flumes as shown in Figure 22.5. Tipping buckets offer very accurate low-flow measurements, but the corrosive atmosphere present in sewage systems precludes the use of mechanical counters. Also, tipping buckets can become "pinned" under high flows, so it is important to determine the suitability of the tipping bucket assembly at peak flow.

Due to the reasons cited in the preceding text, elapsed time meters are often used as a means of flow measurement on systems that receive a pumped influent, even though this may result in lower accuracy flow determinations.

In pumped systems, the flow will be stagnant (zero velocity) in pressure mains most of the time. To prevent freezing problems in cold climates, pressure mains must be installed below the frost line. If this is not possible, the pipe should be sloped to drain and provisions for a drainage outlet should be made at the pump end as well as at the discharge point.

In the long term, water storage changes in the wetland are almost always negligible in comparison to other flows. However, water depth is a very useful parameter for other purposes than the water budget, and hence water stage in each cell is a desirable data set. Staff gages should be installed in each cell of the wetland and surveyed for vertical control at the time of start-up (Figure 22.6). When combined with a topographic survey of the wetland, stage measurements provide a quantitative tool for assessing the average, maximum, and minimum water depths in the wetland and the frequency with which these depths occur. These data are essential for



**FIGURE 22.6** A staff gage provides valuable information on the water depth and volume in the wetland. Photo from the Lake Nebagamon, Wisconsin, treatment wetland system, in the exit deep zone. The screen box covers the intake for the effluent siphon.

interpreting tracer measurements of hydraulic residence time and for assessing any detrimental hydroperiod effects on biota. A variety of water level measuring instruments (e.g., analog chart recorders, ultrasonic, pressure-sensor, and capacitance-sensor units) are available for continuously recording water levels at various key points within the wetland.

### Constituent Concentrations

Table 22.2 summarizes an example of monitoring program in support of operation of a municipal wastewater-polishing wetland treatment system. This list includes measurements of the water quality of all major inflows and outflows associated with the treatment wetland. Inflows include the sources of pretreated wastewater entering the wetland as well as natural inflow streams that may have a significant effect on the water quality or the hydrologic budget of the natural wetland treatment system. As described in Table 22.2, the parameter list to be tested at all major inflows and outflows at least monthly includes all regulated pollutants and integrative measures such as $BOD_5$, TSS, pH, dissolved oxygen, water temperature, conductivity, $NO_2$-N + $NO_3$-N, ammonia, nitrogen, TKN, total phosphorus, chloride, and sulfate. The frequency of operational monitoring for system control is dictated by the size and capacity of the system, the sophistication of the owner's staff and sampling equipment, and site-specific factors related to influent quality variability and climatic factors.

Inflow and outflow stations may also be monitored less frequently for selected heavy metals or organics that might be present in the wastewater and for whole effluent acute and chronic toxicity. If water quality characteristics are highly variable for any of the inflow or outflow locations, or if there are weekly or monthly permit limits, sampling should be more frequent than monthly or quarterly.

These water quality data as well as water-budget parameters should be organized and recorded in computerized spreadsheets for visual analysis of variability and trends. The constituent loadings may be compared to design criteria to ascertain if the system is being operated according to design. The constituent mass balances are easily constructed from system data on such a spreadsheet and may be used to determine if design conditions are being achieved, and whether the system is performing as expected.

Table 22.2 reflects the example of municipal wastewater treatment, but constructed wetland systems are used for a wide

**TABLE 22.2**
**Example of Monitoring Requirements for Operation of a Large FWS Wetland Treatment System**

| Monitored Parameters | Sample Locations | Sampling Frequency |
|---|---|---|
| Flow | Inflows and outflows | Daily |
| Rainfall, pan evaporation, and air temperature | Adjacent to wetland | Daily |
| Water stage | Within wetland | Daily |
| **In the Field** | | |
| Water temperature, dissolved oxygen, pH, and electrical conductivity | Inflows and outflows | Weekly |
| Plant cover (dominant species) | Near inflow, near outflow | Annually |
| **In the Laboratory** | | |
| $BOD_5$, TSS, $Cl^-$, $SO_4^{2-}$, $NO_2$+$NO_3$–N, $NH_4$–N, TKN, TP, and pathogens | Inflows and outflows | Monthly |
| Metals, organics, and toxicity | Inflows and outflows | Quarterly |

*Note:* Some of these are needed for permit reasons, others for mass balancing.

variety of other purposes, including animal waste treatment, groundwater remediation, and urban and agricultural stormwater. Each of these applications generates a similar requirement for a strategy to promote understanding. All are based upon knowledge of the water budget, but the suite of chemicals of interest may change, depending upon the application.

### Biological Monitoring

Biological monitoring within a wetland treatment system provides information, concerning the structural integrity (health) of the vegetation and fauna. Protection of this biological integrity is important from an environmental habitat perspective and because of the biota's control of wetland operational performance. It has been suggested that every component of the wetland ecosystem should be monitored, including sediments, water, vegetation, benthic invertebrates, fish, amphibians, reptiles, birds, and mammals (Ecological Services Group, 1996; Wren *et al*., 1997). This concept originates from the fact that some of these ecological components have been routinely found to be above pristine background for some contaminants, as expected for a treatment system.

However, there have been no ecological disasters associated with the thousands of treatment wetlands over the thirty-plus years of the technology, and thus the question involves (possibly) impaired wetland ecology, not acute problems for wildlife. Therefore, biological studies fall in two categories: monitoring needed to foster the design water quality functions, and research needed to more fully define impacts of less-than-perfect water on various organisms that may inhabit the treatment wetland. There is usually an important distinction between constructed wetlands and natural wetlands as recipients of wastewater: the former are usually not subject to intensive biological monitoring but the latter may be in some states (e.g., Florida).

A compromise is typically put in place with some facet of biological monitoring or research being conducted in addition to the requisite compliance and mass balance monitoring. Typical categories are bird, mammal, and fish studies. Broad-scale potential for ecological problems is commonly assessed by examining or forecasting the concentrations of specific chemicals or toxins. For instance, great care is used in the arid regions of the United States because of the known acute effects of selenium that has impacted wildlife wetlands (e.g., Kesterson, California).

Wetlands designed for ancillary benefits can have nontreatment goals and requirements. Trends in use of the treatment wetlands by plant and wildlife species may indicate needed changes in system management to support those nontreatment goals. Surveys for rare or threatened species may also be conducted when appropriate.

### Monitoring Vegetation

Cover estimates and observations concerning plant health should be a routine part of operational monitoring in a constructed wetland treatment system. Because plants grow slowly and are important for maintaining the performance of wetland treatment systems used for water quality treatment, problems must be anticipated and prevented before they are serious or have progressed too far. Reestablishing a healthy plant community in a constructed wetland is a slow process when the plants have been harmed because of operator neglect.

Plant growth is monitored by estimating percent cover and average plant height. These nondestructive techniques are used to ascertain the status of plant development before and during wetland treatment system operation. Plant cover is an estimate of the percentage of the total ground area covered by stems and leaves. This parameter can be estimated by walking through or next to a plant stand and visually determining a cover category for the plants. Typically, seven cover categories are sufficient for this visual estimation method: (1) less than 1%, (2) 1–5%, (3) 6–10%, (4) 11–25%, (5) 26–50%, (6) 51–75%, and (7) 76–100%.

Cover estimates can be made on a finer scale by using a frame to delineate specific areas. Cover estimates should be made at enough locations in the constructed wetland to provide reasonable statistical averages for comparison between cells and between dates.

Ground-level photography may be used to provide archival information about vegetation. Photos at identified specific locations give a visual time series of potential succession in the system. For instance, a rodent eatout may be documented (see Figure 19.6 for example).

Aerial photography is an excellent supporting tool that can provide estimates of the type and percent cover of vegetative communities in treatment wetlands. Figure 22.7 illustrates the vegetative patterns for one of the constructed treatment wetlands at the Des Plaines River site near Wadsworth, Illinois. The various areas of different vegetation are clearly visible, but ground-truthing is required to establish which plants are growing in the zones.

## 22.3 WATER LEVEL AND FLOW MANAGEMENT

There are not many controls on the operation of a treatment wetland, especially Type *A* systems that mimic passive, natural wetlands. Water depth and water flow rate are the two principal controllable features.

### Flow Management

For many continuous flow FWS treatment wetlands, there is not a choice of the amount of water that reaches the wetland on a daily basis. However, there are choices for some stormwater systems and for lagoon-wetland combinations. For larger continuous flow systems, there are multiple flow paths and hence decisions on how much water to allocate to these parallel paths. Any system may incorporate recycle, which will be under the control of the operator.

### Flow Balancing

In the case of multiple parallel paths, the operator has the choice of how much of the incoming flow to apportion to each flow path. Often, the adjustment of the proportion is

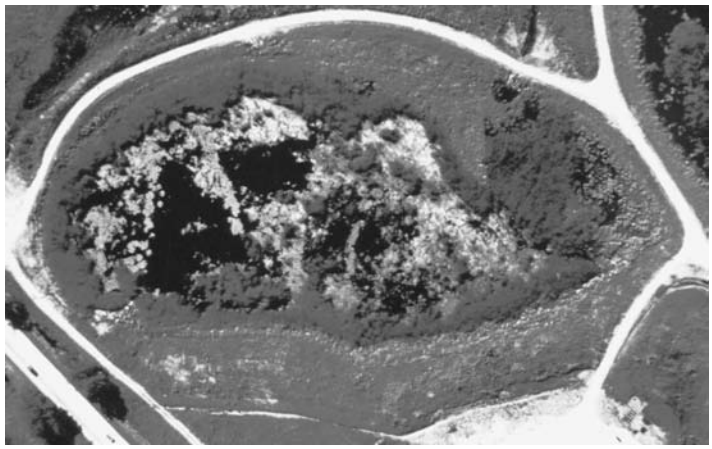


**FIGURE 22.7 (A color version of this figure follows page 550)** Wetland EW3 at the Des Plaines River site near Wadsworth, Illinois. The fringe zone is vegetated by cattails and bulrushes, whereas the interior has submerged aquatic vegetation and floating-leaved plants. Open water shows as the dark areas on this false color infra-red photo.

not physically easy but rather involves opening and closing of multiple valves or structures. The performance of each flow path depends upon the amount of water it receives, with the optimum occurring when the hydraulic loadings to the various flow paths are equal, so that no one flow path is underworked or overworked. It is informative to examine the sensitivity of the over all system performance to the flow split or potential imbalanced loading. To do so, consider a hypothetical example of a system with two flow paths of equal area in parallel, both of which perform according to the *P-k-C** model.

For the conditions indicated in Figure 22.8, if the equal fractions of the flow are sent to each of the two paths, the inlet concentration is reduced by about 80%, from 100 to 19.3 mg/L. As more water is diverted to path 2, its performance starts to decline, until the extreme when all the water goes to path 2, doubling its hydraulic loading compared to design. At that extreme, the outlet concentration is about double the design value at 39.3 mg/L (about 60% reduction). Of course, path 1 becomes underloaded, and its performance increases. The combined outflow from the system shows that imbalance as well. But interestingly, a modest amount of imbalance is not particularly harmful. For the hypothetical example, a 60–40 split is forecasted to raise the outlet concentration to 20.1 mg/L, about 5% higher than design.

The extreme of shutting down one flow path may be necessitated by a need for unusual maintenance activities. For instance, flow paths in STA1W of the Everglades protection project have suffered hurricane damage, requiring shut down of a parallel path for vegetation reestablishment. To the extent that such events may be anticipated, allowance can be made in design. Water may be diverted elsewhere for

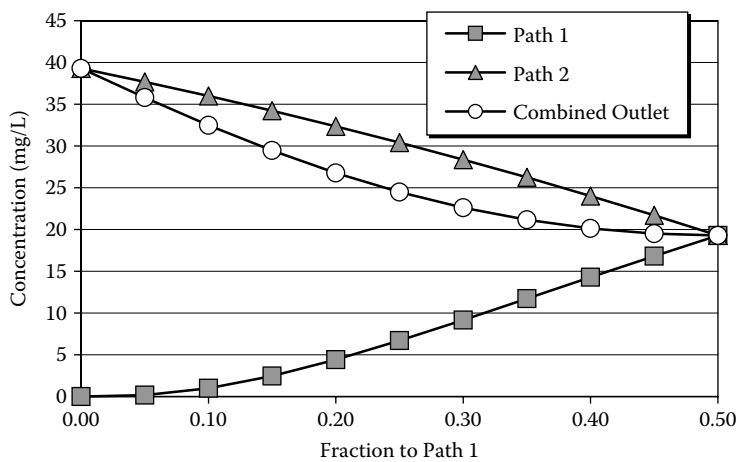



**FIGURE 22.8** A hypothetical example of flow imbalance between two parallel paths in a treatment wetland, for the conditions shown in the corresponding table. Path 1 is overloaded at the expense of Path 2.

| | |
|---|---|
| Total Flow $Q$ = 5,000 $m^3/d$ | $C_i$ = 100 mg/L |
| Path 1 Area = 100,000 $m^2$ | $C^*$ = 0 mg/L |
| Path 2 Area = 100,000 $m^2$ | $k$ = 20 m/yr |
| Wetland Area = 200,000 $m^2$ | $P$ = 3 TIS |

treatment if such alternatives exist, or extra area and paths may be included in anticipation of path shutdowns.

### Systems with Storage

Flow management is an important part of those systems that involve seasonal or event storage because the stored water may be discharged according to any of several schedules (see Chapter 17). In all of these systems, the overriding goal is to bleed down the stored water in a timely fashion to make room for the next event or for the next storage season.

In small-scale systems with winter storage, cost considerations will usually lead to storage that does not much exceed the volume of inflow over the cold season. In spring, the operator is faced with selecting the program of withdrawals from storage to be sent to the treatment wetland. Such withdrawals may be limited by pump capacities, which would also be constrained by cost considerations, or if gravity discharge is possible, the withdrawal rates may be much less constrained. For instance, the Houghton Lake, Michigan, system has a wetland discharge season extending from May 1 to October 15 (170 days), but the transfer pump is capable of transferring the water over about 100 days. Therefore, the operator has discretion over which days there is inflow to the wetland, just so long as the storage is at its minimum level at the end of the wetland-operating season. Strategies ranging from paced uniform discharge to fast emptying followed by minimum pond level maintenance have been used (see Chapter 17).

The discharge of municipal wastewater treatment lagoons is in some circumstances constrained by the ability of receiving waters to maintain acceptable water quality. Such cases occur when the volume of the discharge is significant compared to a receiving stream flow. In that situation, the stream may become dominated by the effluent, especially at times of low recipient flow. It may then be desirable to vary the discharge of treated effluent to avoid unacceptable water quality in the combined flow downstream of the discharge. The ability to vary discharges is in turn conditioned on the ability to store water during periods of low acceptable discharge. The flows in the recipient dictate the magnitude of allowable discharge, and hence this strategy is termed *hydrograph controlled release* (HCR; Kadlec and Pries, 2004).

Stormwater systems may be preceded by flow-equalization reservoirs to prevent the low degree of treatment that accompanies large event flows. Examples of this are currently under design and construction in South Florida, where vast (5–10,000 ha) reservoirs will collect stormwater runoff before passing it through the comparably vast stormwater treatment areas (constructed wetlands), many of which are already built. These are likely to be HCR systems, serving the water-quantity needs of the downstream-receiving systems, which are the Florida Everglades. Operating protocols are currently under development.

### Bypassing

It is often not feasible, for hydraulic reasons, to treat large event flows in stormwater wetlands. If the hydraulics allow, treating all the water would generally result in the greater load removal, but will pass higher concentrations to receiving waters for the large events. Hydraulic integrity often requires that extreme flows be avoided or else damage to the wetland may result. Therefore, the design of such systems typically allows for bypass. However, this is usually accomplished by the design of control structures rather than by any overt operator action. Exceptions are the STAs of South Florida, most of which have operator-controlled bypass structures.

### Depth Management

It has been shown that depth is not a crucial factor in the areal removal rates of many wastewater parameters (see Chapters 9 and 10, for example). The extra detention time of deeper water does not foster a higher removal, and in many cases, shallower water shows better removals. It is much more important to manage water depth for the health and vigor of the plant community, which is typically sensitive to water depth.

#### Depth Controllability

In very large FWS wetlands, the hydraulic resistance of the vegetation creates a dilemma for operations. The water depth is not controllable by changes in outlet structure settings, except for a small region near the outlet (see Chapters 2 and 18). Over most of the system, the water flow rate and the vegetation resistance will control the water depth. Therefore, it is important to distinguish between wetlands that can have depth control via outlet structures and those that cannot. In the latter case, depth control is unavoidably linked to flow control and vegetation management, and not to any modification of structure settings.

As an illustration, consider the two hypothetical FWS wetlands for which different level scenarios are calculated in Figures 22.9 and 22.10. Both have the same hydraulic-loading rate of 5 cm/d, and both are presumed to have the same (high) vegetation resistance to flow. First consider a modest size 1-ha wetland with $L{:}W = 4$. If operation is at an exit weir setting of 0.5 m, then there is essentially a zero gradient in the water surface (level pool). If the exit weir is then lowered to 0.15 m, the water level goes down everywhere, with only a few centimeters of head loss forecast (Figure 22.9; 18.9 cm down to 15.0 cm). The average depth is lowered to 17.1 cm. Next consider the long, narrow wetland with $L{:}W = 25$. When the outlet weir elevation is 0.5 m, the inlet elevation is 3.9 cm higher and the mean depth is 0.52 m. For this system, when the outlet elevation is lowered to 0.15 m, there is not a drawdown of the entire wetland (Figure 22.10). The inlet depth drops only to 0.48 m, and the mean depth is reduced only to 36.5 cm. For the specified flow, the vegetation is holding back the water and lowering the outlet level cannot change that.

Design should be such that water depth control is at the outlet structure or else the operator has no option for depth control except to lower the flow rate. For HSSF wetlands, the hydraulic profile will be governed by both Darcy's Law and the outlet weir setting. Again, outlet control is the preferred method of regulating the water level (see Figure 21.11). This will require a conservative bed design with a high hydraulic conductivity.

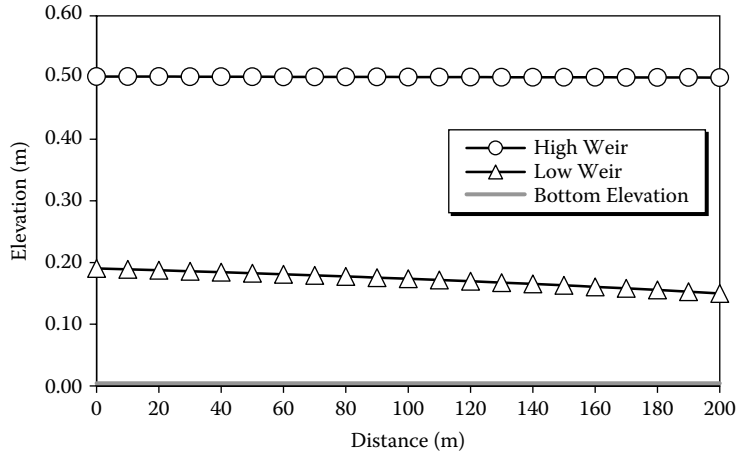



**FIGURE 22.9** Effect of lowering the outlet weir setting on the water depth in a small, moderate aspect ratio wetland.

| | | |
|---|---|---|
| $L$ = 200 m | $A$ = 10,000 m$^2$ | |
| $W$ = 50 m | Aspect = 4.0 | |
| $Q$ = 500 m$^3$/d | $Q$ = 0.006 m$^3$/s | |
| $q$ = 0.050 m/d | HLR = 5.00 cm/d | High Weir Inlet Depth = 0.500 m |
| $a$ = 1.00E +07 m$^{(1-b)}$/d | $\Delta x$ = 10 m | High Weir Outlet Depth = 0.500 m |
| $b+1$ = 3 | $\Delta hB$ = –0.002 m | Low Weir Inlet Depth = 0.189 m |
| $c$ = 1 | d$B$/d$x$ = 0.000010 m/m | Low Weir Outlet Depth = 0.150 m |

## Water Depth, Plants, and Nutrient Loadings

Water depth is an important variable in the survival of wetland vegetation. In mitigation and habitat wetlands, it is coupled with a hydropattern to provide the desired growth conditions for any particular species. The depth that may be tolerated for continuous inundation is less than that which can be withstood for brief periods of time (Table 22.3). Some of the more robust plants, such as *Typha*, can withstand prolong periods of drought, lasting many months, and can temporarily be completely submerged. At continuous flooding water depths of 1 m or more, emergent plants in treatment wetlands can no longer survive.

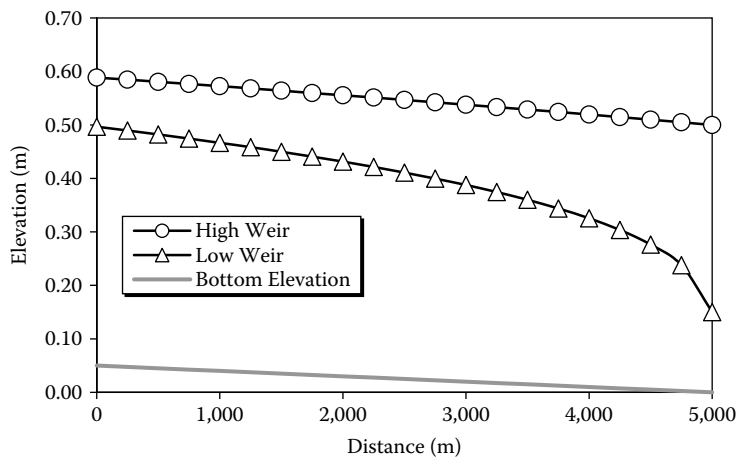



**FIGURE 22.10** Effect of lowering the outlet weir setting on the water depth in a large, high-aspect ratio wetland.

| | | |
|---|---|---|
| $L$ = 5,000 m | $A$ = 1,000,000 m$^2$ | |
| $W$ = 200 m | Aspect = 25.0 | |
| $Q$ = 5,000 m$^3$/d | $Q$ = 0.058 m$^3$/s | |
| $q$ = 0.005 m/d | HLR = 0.50 cm/d | High Weir Inlet Depth = 0.539 m |
| $a$ = 1.00E +07 m$^{(1-b)}$/d | $\Delta x$ = 250 m | High Weir Outlet Depth = 0.500 m |
| $b+1$ = 3 | $\Delta hB$ = –0.050 m | Low Weir Inlet Depth = 0.447 m |
| $c$ = 1 | d$B$/d$x$ = 0.000010 m/m | Low Weir Outlet Depth = 0.150 m |

**TABLE 22.3**
**Water Depths for Common Treatment Wetland Plants**

| Common Name | Scientific Name | Best Depth (cm) | Flood Depth (cm) |
|---|---|---|---|
| Bulrush | *Scirpus californicus* | 30–60 | 120 |
| Soft-stem bulrush | *Scirpus validus* | 15–30 | 60 |
| Broadleaf cattail | *Typha latifolia* | 30 | 60 |
| Narrowleaf cattail | *Typha angustifolia* | 30 | 60 |
| Common reed | *Phragmites australis* | 30–60 | 90 |
| Arrowhead | *Sagittaria* spp. | 3–8 | 30 |
| Yellow flag | *Iris pseudacorus* | 3–8 | 45 |
| Pickerel weed | *Pontederia cordata* | 8–15 | 30 |
| Soft rush | *Juncus effusus* | 8–15 | 25 |
| Woolgrass | *Scirpus cyperinus* | 3–15 | 30 |

*Source:* Adapted from Pineywoods RC&D (2001) *Constructed Wetlands in East Texas* (4 volumes). Pineywoods Resource Conservation and Development Council, Nagodoches, Texas.

For SSF wetlands, the situation is different because there is an unsaturated region within the wetland bed. For HSSF wetlands, this unsaturated region is a result of the water level control regime and the desire to keep flows within the wetland bed. If insulating materials such as mulch are used, this will increase the thickness of the unsaturated rooting zone. In VF wetlands, unsaturated conditions will occur throughout the wetland bed on a periodic basis due to pulse-feeding and the load-and-rest operating regimes typically used in these systems.

The net result of such unsaturated conditions is to shift the competitive advantage away from plants that require submerged root zones. SSF wetlands are more likely to be dominated by plants that enjoy wet but not submerged root zones. Plants in the genera of *Phragmites*, *Schoenoplectus*, and *Phalaris* are commonly used in such systems (Vymazal *et al*., 1998; Wallace and Knight, 2006; Stein *et al*., 2006a).

There are many sources of recommendations for water depths and associated hydroperiods for the common wetland plants (for example, Hotchkiss, 1972; Niering, 1985; Dressler *et al*., 1987; Boon and Groe, 1990; Thurnhorst, 1993; Eggers and Reed, 1997; Pineywoods RC&D, 2001; Shaw and Schmidt, 2003). Hydrologic tolerances of wetland plants are further discussed in Chapter 3. The treatment wetland operator should understand the allowable depth and hydroperiod ranges for the plants at a particular site.

As noted in Chapter 3, floating mats are prone to form under excess depth conditions in FWS wetlands. For instance, if a good growth of cattails (*Typha* spp.) develops under shallow water conditions, it will likely have a shallow, interlocked rhizosphere. This root mat has an inherent buoyancy created by various gas-generating processes (such as methanogenesis). If water depths are then raised, the entire mat may tear loose from the rooting media and form a floating mat. If the mat covers the entire wetland cell, then it can function effectively in treatment. But if floating islands develop, they are subject to wind-driven motion and severe disruption of the wetland. This happened in Cell 2 of the Everglades Nutrient Removal Project (ENRP), with disastrous impairment of treatment. Removal of the plants was necessary.

### Seasonal Depth Adjustments

Although shallow water fosters plant growth and is effective in promoting treatment in FWS wetlands, it can be problematic in winter due to ice formation. In densely vegetated wetlands, the plant stems hold the ice layer in place. If water levels subsequently drop, the ice layer generally drops with it, although an air gap can form under the ice at some localities within the wetland if the ice sheet is held up by plant stems. However, this anchoring of the ice layer can create severe damage if water levels are subsequently raised. The ice layer floats and tears out embedded vegetation as it moves upward. If the ice layer becomes thick enough, under-ice flow is impeded, which can lead to over-ice flow (Kadlec, 1987). Water flows upward through holes or cracks, emerges, and flows downgradient over the ice and under the snow. For example, this mode of flow is experienced each winter at the Frank Lake, Alberta, treatment wetland.

It is possible to provide more space for flow under the ice by raising water levels prior to ice formation. However, care must be taken that such increased water depths do not entirely submerge the plants, because the oxygen transfer necessary for their survival is primarily obtained via the standing dead stalks and their airways (see Chapter 5). Indeed, an effective method of destroying cattails is to cut them and overflood the stubble. This procedure of water level increase was used effectively at the Listowel, Ontario, FWS wetlands (Herskowitz, 1986) and is currently employed at the Brighton, Ontario, wetland.

For HSSF wetlands, there is a treatment penalty with deeper beds (see, for example, Table 8.13). If shallower beds (0.3 m) are selected, there is the potential that some bed volume will be lost due to ice formation in the winter. For very cold situations, mulch insulation of the SSF wetland may be required, as discussed in Chapter 4.

## 22.4 CONTROL OF NUISANCE ANIMALS

Experience has shown that a large variety of creatures may interfere, in some way, with the desired operational conditions treatment wetlands. The list of animals that have caused some degree of concern includes deer, elk, cattle, wild hogs, gophers, monkeys, coyotes, feral cats, manatees, and turtles. However, most problematic are some birds, rodents, some fish, and mosquitoes.

### Birds

Blackbirds are attracted to treatment wetlands, both as a breeding location and as a place of refuge. Redwing blackbirds (*Agelaius phoeniceus*) love to nest in cattails and do so in great numbers in treatment wetlands of temperate North America. Yellowheaded blackbirds (*Xanthocephalus*



**FIGURE 22.11** Waterfowl at the Commerce Township, Michigan, municipal wastewater-polishing wetland.

*xanthocephalus*) also nest and find nighttime refuge in treatment wetlands (see Figure 19.5). Although their large numbers may contribute to pathogens in the wetland, no management techniques are typically used for blackbirds.

Waterfowl may be more problematic because of their large size and numbers during migration. Canada geese (*Branta canadensis*), ducks, and other waterfowl are known to cause significant depredation through grazing and uprooting of plant materials in FWS wetlands. There are a number of threatened and endangered birds that make use of wetlands in general and may be attracted to treatment wetlands. According to federal regulations in the United States, their presence requires that the area in question be managed on their behalf.

Wetlands in an urban setting are very attractive to geese, which congregate in large numbers at such isolated pockets of habitat. Even if their grazing does not seriously harm the wetland vegetation, they can deposit large quantities of fecal matter on walking paths and grassy margins of the wetlands (Figure 22.11). It is herbivory that can cause considerable grief for operators, especially during the vegetation establishment phase (see Figure 18.37). Young plants and tubers are preferred food for geese, swans, and some ducks. If the wetland planting happens to coincide with either the spring or autumn migration, the wetland can be essentially denuded in a day or two.

Under the Endangered Species Act, the U.S. Fish and Wildlife Service (USFWS) is mandated to identify and list plant and animal species that are threatened or endangered by extinction, or that are considered likely to be threatened in the future. Threatened and endangered species cannot be harmed, killed, or otherwise negatively affected by human activities. The USFWS reviews potential impacts to federally listed species and can veto a discharge permit by issuing a "jeopardy" opinion. The potential for occurrence of threatened or endangered species inhabiting a project site should be considered during project planning.

Experience indicates that the presence of federally protected species does not always necessitate substantial changes to wetland treatment project siting, design, or operation. In a number of cases, the discharge of treated wastewater effluents to constructed wetlands has enhanced the population of threatened or endangered wetland-dependent species by creating additional habitat and food resources. For example, populations of bald eagles (*Haliaeetus leucocephalus*), wood storks (*Mycteria americana*), and snail kites (*Rostrhamus sociabilis*), all federally protected bird species, have increased in and around the 485-ha Orlando Easterly constructed wetland treatment system in Florida. Conversely, the presence of protected species may require modifications of operations. For instance, the STAs of South Florida are attractive to burrowing owls and stilts, which nest in or on the levees. Therefore, flooding depths that endanger their nests are precluded. The treatment wetland operator should be aware that operations may have to be modified if threatened or endangered species begin using the wetland.

### Fish

Fish may be problematic in treatment wetlands, particularly those which feed or nest on the bottom. For instance, at the Des Plaines constructed wetlands in Illinois, carp (*Cyprinus carpio*) were unintentionally introduced into the wetland from the pumped source water (Des Plaines River), presumably as very small individuals. Over the course of two years, the fish grew in size to about 15 cm. These larger fish were observed to stir bottom sediments and impair suspended solids removal. TSS in the wetland outflow was at about 15–20 mg/L with fish present and was reduced to less than 10 mg/L after they were removed (Figure 22.12). The carp were removed from the wetland through a winter drawdown and freeze (Figure 22.13).

A similar phenomenon occurred at the Tarrant County constructed wetlands in Texas, where floodwaters from the source river brought carp (*Cyprinus carpio*), shad (*Dorosoma cepedianum*), and bullheads (*Ictalurus* spp.) into the system. The activity of the fish caused severe bioturbation of the sediments during feeding and spawning activities (APAI, 1994).

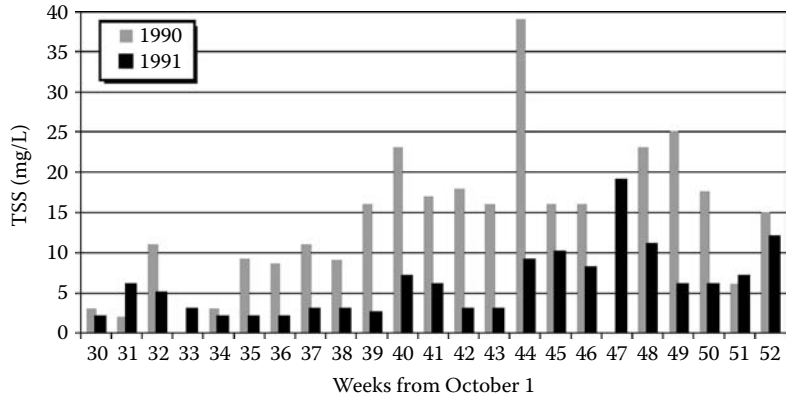



**FIGURE 22.12** Effect of carp on the TSS level in the Des Plaines, Illinois, treatment wetland. Carp were present in 1990 but had been removed in 1991.

The cells were dewatered, and egrets and herons effectively removed the stranded fish.

## Rodents

There are over 2,000 species in the order Rodentia, but it is primarily the water-loving types that are of concern in treatment wetlands. For HSSF wetlands, minor problems are often incurred by small burrowing rodents; in VF systems, such activities can cause a disruption of the filtering sand layer, leading to a loss of treatment efficiency (Weedon, 2002). There are design techniques that can help in controlling muskrats, nutria, and beavers in FWS treatment wetlands, but often these are not completely successful, and the operator should be prepared to exercise appropriate control measures.



**FIGURE 22.13** A drawdown followed by a freeze killed the carp in the Des Plaines, Illinois, treatment wetland.

### Muskrats (*Ondatra zibethicus*)

Muskrat grazing can change FWS treatment wetland areas from a densely vegetated environment to a patchwork of open and emergent areas. Cattails are particularly susceptible to damage due to herbivory and mound building. Additionally, muskrats create open water travel paths, connecting open water areas such as deep zones. These channels are hydraulic short-circuits, which impair the effectiveness of treatment (Figure 22.14).

Muskrat invasions of treatment wetlands are sometimes quite impressive in terms of numbers. Knowlton *et al.* (2002) note that muskrats have been problematic at the 37-ha Columbia, Missouri, project with removal of over 1,300 muskrats in 2.5 years.

Destruction of the wetland vegetative infrastructure may create an attendant loss of some water quality functions but may not harm others (Kadlec *et al.*, 2007). Some wetland functions may not be significantly impacted or even improved. Wetland operators at some sites, such as Brighton, Ontario, have reported that the water quality as it relates to permit limits has been relatively unaffected by herbivory. At the Olentangy, Ohio, site, reductions in oxidized nitrogen continued, with slight improvement, after the eatout. At Nanticoke, Ontario, the wetland continued to meet the effluent objectives with the exception of *E. coli*, after a muskrat eatout. Possibly, waterfowl and muskrat activity produced an abundance of fecal material around the entire perimeter of the wetland and within the wetland. Therefore, it is prudent to examine water quality performance before embarking upon an expensive set of control measures because of muskrat activity.

Muskrat control is given scant attention in existing treatment literature. Kadlec and Knight (1996) provide no information on potential muskrat problems or on the means to eliminate such difficulties. U.S. EPA (2000a) provides brief



**FIGURE 22.14** Muskrat travel channel, connecting two deep zones in the Hillsdale, Michigan, treatment wetland.

advice on the use of hazing and exclusion to protect wildlife from adverse conditions in the wetland and goes on to note that the excessive use of wetlands by wildlife may also cause problems. Controls include trapping, shooting, poisoning, hazing, and exclusion to protect the wetland from excessive vegetation destruction by these rodents (Kadlec *et al*., 2007).

Mink, the primary muskrat predator, is ubiquitous across the range of muskrats. These small animals can and will enter treatment wetlands in search of prey and are not deterred by chain-link fencing. Another option is the use of predator urines to scare off muskrats. When muskrats smell the urine of a predator they will typically avoid the area. Predator urine is commercially available from several sources, and for muskrat deterrent, bobcat urine is recommended. The efficacy of such scent deterrents is not known.

Trapping is an effective method of control and includes live, leg-hold, and colony traps. Live trapping involves the capture of live muskrats, which can be released outside of the treatment wetland environment. Lethal foothold traps capture muskrats around their feet and, typically, the animals are killed after capture to harvest their pelts. Such traps are nonspecific, and a drawback is that they also capture muskrat predators, especially mink. Colony traps can capture multiple muskrats at once and can be used as lethal or nonlethal devices. For muskrat captures, a colony trap is commonly placed underwater to quickly drown muskrats that enter it. Shooting muskrats is another control option. Concentrated efforts should be made at dusk or in the early morning hours for effective control. It is more selective than other methods as it ensures that only target animals are killed. Commercial, licensed hunting can be very expensive; one quotation was $100/h plus $25 per animal ($USD). Additionally, it may be difficult to control large muskrat populations with this method alone. Before this approach is used, local firearms regulations and permitting must be reviewed and abided by.

### Beavers (*Castor canadensis*)

There are two species of beavers: *Castor canadensis* in North America, with recent introduction in southern South America, and *Castor fiber* in Europe and Asia (Müller-Schwarze and Sun, 2003). In North America, beaver is a very destructive wildlife species, causing millions of dollars in damage to roads, dikes, and dams. Beaver damage management is, therefore, of great concern throughout the United States, with many agencies cooperating to minimize damage (U.S. Department of Agriculture, 2000). Many experts believe that the cost of beaver damage is greater than that caused by any other wildlife species in the United States. (U.S. Department of Agriculture, 2004). Their range is from the Gulf of Mexico to the Canadian subarctic, and northern Europe, Scandinavia, and the former Soviet Union. In the absence of habitat constraints or excessive mortality, populations grow rapidly with a typical pair producing three or four kits per year (Long, 2000).

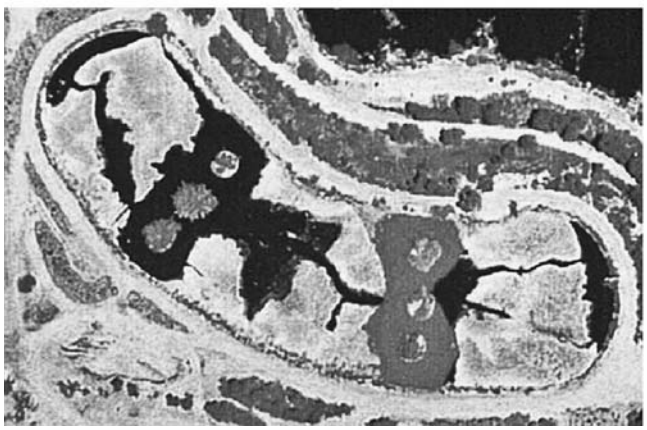
#### *Structure Clogging*

Beavers have several activities that can cause difficulties for treatment wetland operations. The first, and far more serious, is their proclivity for damming sources of running water. This leads to the plugging of treatment wetland structures (see Figure 19.7). At a small wetland site, it is common for the animals to completely clog a structure in one night. Beavers are resourceful in finding and using local materials for damming and clogging structures. Mud, sticks, and small logs are commonly used, but they will also use stones up to several kilograms in weight. Vegetation research quadrat stakes and research sediment traps are also considered as useable dam materials, based on observations from the Des Plaines River wetland site in northern Illinois.

Because of a long history of interactions between beavers and man-made water structures, principally culverts, there has been a great deal of consideration given to methods of



(a)



(b)

**FIGURE 22.15** Beaver travel channel at the Tres Rios Hayfield treatment wetland near Phoenix, Arizona. This short-circuit became evident during this water drawdown event (a). The bottom panel shows that such channels connect the two interior deep zones (each with three islands), and the inlet and outlet deep zones (b).

prevention. There is a long list of patented devices designed to prevent beavers from damming culverts (Goedderz, 1987; Lambert, 1987; Carroll, 1991; Thurber, 1991; Fleury, 2002). These are typically grills, grates, and small mesh devices often installed to protect the upstream openings of culverts; however, they usually require frequent cleaning because they are easily dammed by beavers. Two of the lower maintenance alternatives are the "beaver deceiver" and the "beaver baffler" (U.S. Army Corps of Engineers, 2001).

A beaver deceiver is a fence that discourages damming due to its unnatural shape and large perimeter. Deceivers are usually trapezoidal but may be pentagonal or form long rectangles at narrow flowages. Fences are constructed of braced cedar posts and 6-gauge, wide mesh fencing. A beaver baffler is a cylindrical flow device constructed from multiple layers of wide mesh wire. This system excludes beavers from a culvert opening or dam breach while allowing free flow of water. The likelihood of beaver clogging is reduced by spreading the water intake over a large area. A baffler consists of two 5-m long, concentric cylinders of concrete reinforcing wire, including an inner cylinder that fits inside the culvert and an outer cylinder that is 30 cm larger in diameter.

### *Channelization*

Because beavers are primarily aquatic animals that are adept swimmers and capable of long periods of underwater activity, they prefer to travel in water. This affords protection from predators. Travel canals are often constructed so that the animals can proceed under water. This can pose a problem in treatment wetlands, particularly those with deep zones. Beavers may construct travel channels to move from one deep zone to another, thus constructing short-circuits (Figure 22.15).

### *Herbivory*

Beavers are herbivorous but can utilize an extremely varied diet. Various parts of trees are the preferred food at times of the year, including the inner bark (cambium) and roots, shoots, leaves, and buds. Favorites are poplars and aspen, willows, birches, and alder. However, they will also make use of water plants including reeds, rushes, and sedges (Long, 2000). Many treatment wetlands are built with a view toward integrating the system into the surrounding landscape by surrounding the system with appropriately sized clumps or bands of terrestrial vegetation. Trees are often an important component of these auxiliary plantings, which strive for habitat diversity. The combination of a wetland, particularly one with deep zones, and an adjacent tree plantation succeeds in providing ideal habitat for beavers. Once established, the animals go about their lifestyle, which includes felling trees, usually within a few tens of meters of the water. North American beavers prefer the genus *Populus* and will cut numerous trees in a hybrid poplar plantation (Figure 22.16).

### *Burrowing*

Beavers utilize both constructed lodges and bank dens as refugia and a wintering domicile in which to raise a family. The banks along lakes and streams are places for dens, which relieves the animals of the effort necessary to build elaborate in-water lodges (Müller-Schwarze and Sun, 2003). Less than half of Eurasian beavers build lodges, but slightly more than half of North American animals do so. The berms and levees of treatment wetlands apparently qualify as acceptable burrowing sites. The animals are not small (their weight ranges upward to 25 kg), and bank burrows are, therefore, not small either. Considerable amounts of wetland water leakage or transfer may occur as a result of such burrows. More seriously, the integrity of a containment levee may be compromised, ultimately resulting in failure (Figure 22.17).

### *Control*

A treatment wetland operator should be prepared to deal with beaver problems, despite proper preventative efforts of design and construction. There are various ways to manage beavers, so population goals can be met. In some circumstances, treatment wetland owners who have minor damage problems, such as a beaver chewing trees, may elect to do



**FIGURE 22.16** A hybrid poplar plantation adjacent to the Roblin, Manitoba, treatment wetland proved to be a favorite food source for beavers. Note that trees in the foreground have been clipped, but most enjoy some regrowth.

nothing. It is typically legal to shoot or trap beaver causing damage or a nuisance on a property, and trapping is effective. For example, in the state of Wisconsin, the annual harvest is on the order of 40,000. An average pelt price is about $20 USD, and therefore commercial trapping may be an option (Wisconsin Department of Natural Resources, 2005). In most cases, live-trapping and relocating beaver is not a good idea. Relocation may be less humane than killing beaver if they do not have time to store food or build a dam before winter or if the new location's habitat cannot support them.

Heavy wire mesh, heavy gauge hardware cloth, or tar paper will discourage beaver from cutting and gnawing trees along berms and in adjacent terrestrial plantings. About 1 m of height is wrapped. Mesh size should be less than 2 cm to be effective. Area fencing may also be used. Getting rid of attractive food items may discourage them from settling. As noted earlier, beaver prefer trees like poplar, willow, birch, and alders. Planting spruce and balsam trees are good alternatives to discourage beaver settling. Removal of the beaver dam, the lodge, and the beaver is generally the last option because of the difficulty and expense involved and because new beaver may recolonize.

### Nutria (*Myocastor coypus*)

Nutria is a nonnative species in North America. They are South American rodents that were introduced in North America in 1899 by California fur ranchers, but there are now feral populations in 22 states across the United States. Introduction of a few animals in Louisiana in 1938 led to a population of 20 million animals by the 1950s (Bounds, 2000). They were also introduced into Britain, where they caused serious damage to crops, drainage systems (through



**FIGURE 22.17** Beaver excavation on the containment berm of the Tres Rios Hayfield treatment wetland near Phoenix, Arizona.

burrowing), and native plant communities in East Anglian wetlands (Gosling *et al*., 1988).

Nutria are water-loving rodents, intermediate in size between muskrats and beavers, with a typical maximum body weight of 8 kg. They eat about 25% of their body weight each day. They consume the basal portions of plants, including roots, rhizomes, and tubers, and typically excavate the entire plant leaving little chance for regrowth. A vegetated marsh is converted into an open water. In Maryland, their diet (unfortunately for treatment wetlands) includes bulrushes (*Scirpus* spp.), spikerushes (*Eleocharis* spp.), cattails (*Typha* spp.), pickerel weed (*Pontederia* spp.), and arrowhead (*Sagittaria* spp.) (Bounds, 2000). Also, burrowing activities can damage dikes, levees, and other water control structures. Nutria can also outcompete and displace native muskrat populations (May, 2004).

As noted in Chapter 19, the Victoria, Texas, wetlands suffered significantly because of herbivory by the nutria, necessitating hunting and trapping. A contract trapper was employed at the Mt. Angel treatment wetland in Oregon. The treatment wetlands at Halsey, Oregon, also encountered nutria eatouts. Treatment wetlands in the southern tier of states, plus the east and west coastal states, could potentially encounter nutria problems, and operators should be prepared to implement control measures as necessary.

### Insects

The wetland environment offers a wide variety of niches for various insects. Aquatic insects comprise a diverse assemblage of 11 orders in freshwater systems and fill a wide spectrum of ecological niches, especially in FWS wetlands. Common are the nymphal and/or adult stages of the *Odonata* (dragonflies and damselflies), *Hemiptera* (true bugs such as back swimmers, creeping water bugs, and water scorpions), *Coleoptera* (aquatic beetles), and *Diptera* (mosquitoes, horse flies, deer flies, and midges).

Some stages of certain insects can cause extensive herbivory in treatment wetlands (Snoddy *et al*., 1989; Wallace *et al*., 2001). However, it is often observed that performance is not materially affected by such insect destruction, and a robust wetland ecosystem can recover quickly from these temporary events. Therefore, insect herbivory can generally be tolerated with no maintenance activity, despite the outward appearance of a vegetative disaster. It should be recalled that many of the treatment functions do not rely upon the green parts of plants. Further, insects typically attack one species of vegetation, and the diversity in the wetland plant community will lessen the impact of any one species being temporarily impaired. If the attack does not destroy the belowground plant parts, regrowth is probable. The general advice to the operator is do not panic, but watch the treatment performance.

Aquatic diptera are the most important macroinvertebrates in FWS wetlands in terms of number of species and population size. Mosquitoes (Family Culicidae) and midges (Families Chironomidae and Heleidae) are common wetland inhabitants. Only the larvae and pupae of these groups actually occur in the wetland water column because the adults emerge from their pupal skins and lead a terrestrial life feeding on nectar or blood. The nuisance potential of mosquitoes in wetlands and normal control methods are discussed here.

### Mosquitoes

Mosquitoes present a biting nuisance for both humans and animals. However, the bigger difficulty for treatment wetlands can be the potential for transmission of human diseases. Wetlands and other shallowly flooded habitats provide breeding habitat for mosquitoes. These mosquitoes are a nuisance and may transmit debilitating diseases to humans and livestock. Nonetheless, mosquitoes are an essential part of wetland biodiversity (Schäfer, 2004).

Females of most mosquito species require a blood meal from a vertebrate host to stimulate the development of each clutch of eggs, and this behavior potentially places the surrounding human population at risk of mosquito contact. Species associated with wetlands exhibit ambushing or hunting behavior to acquire blood. Although they engage in hunting behavior during crepuscular and nocturnal periods, *Aedes* and *Psorophora* also ambush hosts during the day and generally will attempt to feed whenever large mammals such as humans or domestic animals enter their immediate surroundings. These species are avid and persistent feeders, produce a painful bite, and will pursue potential hosts after they are encountered. Species that ambush during the day create the greatest demand for control due to public awareness but generally do not fly far during the day and infrequently are involved in pathogen transmission.

Eastern equine encephalomyelitis (EEE) currently occurs east of the Mississippi River in the United States. It operates in a transmission cycle among *Culiseta melanura* mosquitoes and wild birds. In the southwestern United States, only Western equine encephalomyelitis (WEE) and St. Louis encephalitis (SLE) viruses have caused widespread illness in humans and are likely to be transmitted by mosquitoes associated with wetlands. WEE is a zoonosis maintained in a cycle involving *Culex tarsalis* mosquitoes and birds such as sparrows, doves, and quail. Transmission to humans and domestic animals is incidental to the basic cycle. SLE is distributed widely throughout the Western Hemisphere, but most outbreaks typically have occurred at warm latitudes in North America. It may involve any or all of five *Culex* species and also operates through avian intermediate hosts. Human malaria is caused by infection with a blood parasite and was introduced into North America where it is transmitted effectively by several local *Anopheles* mosquito species. Malaria in the United States is typically associated with tourists or immigrants who acquire their infection abroad or relapse a previous infection. In North America, the West Nile (WN) virus has spread from east to west with startling rapidity, beginning in 1999. It follows a cycle that includes many types of birds as reservoirs and is transmitted to humans by biting. Severe cases of the disease are infrequent (about 1 in 150), but some severe cases result in death.

**TABLE 22.4**
**Mosquito Density by Site and System in the NADB v.2.0 (1998)**

| Site Name | System | System Name | Average Density (#/m$^3$) |
|---|---|---|---|
| **Constructed Wetlands** | | | |
| Arcata, California | 1 | Arcata Wetlands | 217 |
| Santa Rosa, California | 1 | Kelly Farm | 915 |
| Hemet/San Jacinto, California | 1 | Hemet/San Jacinto | 1,337 |
| Sacramento, California | 1 | Sacramento Wetlands | 1,012 |
| **Natural Treatment Wetlands** | | | |
| Reedy Creek, Florida | 1 | Reedy Creek WTS1 | 791 |
| Reedy Creek, Florida | 3 | Reedy Creek WTS2 | 4,830 |
| Poinciana, Florida | 1 | Boot | 288 |

Mosquitoes are viewed quite differently in different climatic regions. In the far north and in moist temperate zones plus the humid subtropics, mosquitoes are pervasive and accepted as a part of the environment. However, in arid climates, mosquitoes are much less abundant and viewed as a controllable nuisance. States such as California and Arizona have mosquito control agencies, which operate under strict guidelines. Treatment wetlands in those states have been heavily regulated for mosquito control. In states where mosquitoes are naturally abundant, there is little or no regulation. Therefore, the sections that follow are directed to treatment wetland projects that will be subject to heavy regulation. Disease transmission by mosquito vectors is virtually nonexistent in cold climates.

### *Mosquitoes and Wetlands*

Mosquito species found in treatment wetlands can be classified into two groups based on their egg laying and hatching behavior. Females of some species lay their eggs directly on the water surface or on the leaves of aquatic plants. The eggs hatch usually within a few days and do not need an external hatching stimulus. These behavioral traits are characteristic of mosquitoes of the following genera (or subgenera): *Anopheles, Coquillettidia, Culiseta*, *Culex*, *Mansonia,* and *Uranotaenia*. By contrast, the eggs of floodwater mosquitoes in the genera *Aedes* and *Psorophora* normally are deposited on moist soil or debris on the shore around aquatic systems and do not hatch until submerged by rising water levels.

Reduced dissolved oxygen levels limit the effectiveness of natural aquatic predators such as mosquitofish (*Gambusia affinis*) and aquatic insects (dragonfly and damselfly larvae and beetles). Thick stands of surface vegetation may also limit the access of predaceous fish to mosquito larvae. The abundance of *Culex tarsalis* larvae in a wildlife habitat marsh wetland near Los Angeles, California, was found to correlate with the presence of floating mats of dead cattails and was less related to stem density of cattails (*Typha* spp.) or bulrush (*Scirpus californicus*) (Walton *et al*., 1990).

Generally speaking, adequately designed and properly operated FWS treatment systems will produce the same amount of mosquitoes as equivalent natural wetlands. Immature mosquito populations in treatment wetlands typically indicate low average densities (Table 22.4).

Mosquitoes can travel 1–5 km from their hatching location (Service, 1993). Mosquito population densities decrease exponentially with distance from their hatching site. The distance covered by hunting females tends to be species-specific but also depends upon host availability, mosquito population size, landscape features, and wind speed. Therefore, although flight is variable and influenced by landscape topography, mosquitoes developing in treatment wetlands have the potential to disperse long distances and infiltrate surrounding residential areas during host-seeking flights.

It is not possible to exclude egg-laying adult mosquitoes from FWS wetlands. Mosquito production is greatest in thick, heavy, monotypic stands of vegetation. Dense stands of monotypic vegetation physically obstruct predator access to mosquitoes and hinder control efforts. Unfortunately, plant species commonly used in FWS treatment systems, such as cattail, bulrush, and common reed, are favorable for mosquito production (Knight *et al*., 2004).

Heavy accumulation of plant detritus results in areas where mosquito larvae are sheltered from predators. Open-water areas with water depths greater than 30–40 cm rarely support mosquitoes (Sinclair Knight Merz, 2000). Deep zones provide refuge for mosquito fish and other predators, but unfortunately they are also preferred habitat for nuisance rodents.

## Control Strategies

The goal of system operators should be to minimize the number of larvae that can successfully survive to become adult, biting mosquitoes. This is generally achieved by either creating conditions favorable for mosquito larvae predators or through chemical/biological control of the mosquito larvae. Predators of mosquito larvae include a variety of aquatic insects (Greenway and Chapman, 2002). However, the most commonly used predator is the mosquito fish (*Gambusia holbrooki*). Mosquito fish do not survive in cold climatic zones, and their northern analogs (the top-minnows) are not nearly as effective for control of immature mosquito stages.

Two bacteria species, *Bacillus thuringiensis* variety *israelensis* (Bti) and *Bacillus sphaericus*, are commonly used for mosquito control in treatment wetlands (Williams *et al*., 1996). Pellet and granular forms are available for vegetated environments but are effective only if they can be distributed into areas where the mosquito larvae are present. As a result, effective use of *Bacillus* toxins requires functional access points into the wetland, including:

- Drivable access around all wetland cells
- Boat ramps, boardwalks, or other structures to allow access into deep-water zones
- Embankments and wetland areas that are free of power lines and other obstructions that might limit aerial spraying
- Piping and valving that allows application of chemical and biological control agents directly into the flow

Because water is continually flowing through the treatment wetland, control agents may have to be reapplied periodically.

Conventional chemical pesticides, such as the organophosphate compound temephos, are used for vector control operations in the United States because the chemicals are effective against a wide range of insects and remain economical for larviciding in large-scale abatement efforts (FCCMC, 1998). Such chemicals are not particularly toxic to mammals. Larviciding using either mosquito-specific or rapidly degrading control agents is preferred to adulticiding programs that use residual insecticides such as organophosphates (e.g., malathion, chlorpyrifos) or natural and synthetic pyrethroids (e.g., pyrethrin, resmethrin). After adult mosquitoes emerge from developmental sites and disperse into the region surrounding a wetland, control is more difficult than efforts focused on the developmental sites of the immature stages. However, adulticiding may be necessary if control efforts focusing on the aquatic, immature stages are ineffective.

Emergent vegetation in constructed treatment wetlands is an integral component of the water treatment process; however, dense emergent vegetation promotes mosquito production and impedes mosquito abatement efforts. Therefore, vegetation management activities may be a consequence of mosquito control. Forms of vegetation management by physical control include herbiciding, harvesting, and controlled burning.

## 22.5 VEGETATION MANAGEMENT

There are two perceptions that the treatment wetland operator should be aware of, and deal with appropriately: (1) the wetland only works if it is a large green expanse, and (2) the plants that were originally planted are the only ones that should be in the wetland. The first may be termed the "cosmetic concept." Although it is certainly true that full vegetation promotes the biogeochemical cycles that underlie many treatment processes, it is also true that the brown component (detritus and litter) is of equal or greater importance than the green leaves. This is especially true when considering cold climate wetlands, which, despite being brown (or even white) during many months, continue to provide treatment during those months. Thus, although human values place a premium on visible green plants, the correlation to treatment is not necessarily strong.

The second idea may be termed the "cornfield concept." As a carryover from agriculture, if the intent is to raise a particular crop, then anything else is a weed and should be removed so as to prevent interference with the crop. However, treatment wetlands are not generally intended to provide a specific crop but rather to promote utilization of nutrients, uptake of chemicals, and formation of microbial substrate. Therefore, it is often less important what species are present than their contributions to treatment.

### General System Care

Treatment wetlands require access facilities including walkable or drivable levees. These need to be kept in good condition. Berms may need mowing, particularly if local regulations prescribe this, as is often the case for treatment lagoons. A perceived problem is the growth of trees in wetland berms, such as cottonwood (*Populus deltoides*), willow (*Salix* spp.), and salt cedar (*Tamarix chinensis*). These are most easily controlled by mowing. This requires that berms be designed with flat enough side slopes for mowing equipment. Systems that are designed for public use will require that walking trails and other facilities be kept in a safe and useable condition.

### Harvesting

The harvest of wetland plants for the purpose of removal of stored chemicals is often suggested but seldom practiced. As set forth in Chapters 9 and 10, it is not effective in heavily loaded systems, because only a small fraction of the removal operates through the vegetation. Any consideration of harvest is therefore confined to agronomic (lightly loaded) systems. As noted in Chapter 3, there are drawbacks to harvesting. Techniques for harvesting large-scale systems are generally not available. Disposal of harvested plant material has associated costs, with few opportunities for product recovery. Harvest removes vegetation that would otherwise form litter and serve valuable treatment functions in the system, such as insulation via snow trapping and carbon supply for denitrification. Techniques for harvesting would presumably involve system dry-down or on-ice methods.

### Weeds

Treatment wetland vegetation is intended to enhance water quality functions in the system and not necessarily to provide habitat. Plants that are weeds from a habitat perspective may be desirable from a treatment perspective, but the two

**TABLE 22.5**
**Wetland Species That Are Sometimes Considered to Be Weeds from a North American Habitat Perspective**

| Scientific Name | Common Name |
|---|---|
| *Eichhornia crassipes* | Water hyacinth |
| *Hydrilla verticillata* | Hydrilla |
| *Lythrum salicaria* | Purple loosestrife |
| *Phalaris arundinacea* | Reed canary grass |
| *Phragmites australis* | Common reed |
| *Salix* spp. | Willow |
| *Typha latifolia* | Cattail |
| *Typha domingensis* | Cattail |
| *Typha angustifolia* | Cattail |
| *Elaeagnus angustifolia* | Russian olive |
| *Sapium* spp. | Chinese tallow |
| *Schinus terebinthifolius* | Brazilian pepper tree |
| *Casurina* spp. | Australian pine |
| *Tamarisk* spp. | Salt cedar |
| *Alternanthera philoxeroides* | Alligator weed |
| *Spiraea douglasii* | Douglas spiraea |
| *Ranunaulus repens* | Creeping buttercup |
| *Juncus effusus* | Soft rush |
| *Cytisus scoporius* | Scot's broom |
| *Lysinachia terrestris* | Yellow loosestrife |
| *Iris pseudacorus* | Yellow iris |

*Source:* From Hayes *et al*. (2000) *Wetlands Engineering Handbook*. ERDC/EL TP-WRP-RE21, U.S. Army Corps of Engineers: Washington, D.C.

categories are not mutually exclusive. It is useful to understand which categories of plant species are considered undesirable from a habitat perspective and what measures are recommended for their control (U.S. Army Corps of Engineers, 2000). Table 22.5 lists categories that include exotic species and some native species from the North American habitat perspective. If a treatment wetland is intended to serve multiple purposes including the ancillary benefit of providing optimal habitat, then these species may be excluded from the planting list and viewed as weeds if they subsequently volunteer into the wetland.

In general, control measures for undesirable species include burning, mowing, hand pulling, mechanical removal, and herbicide application.

However, there may also be reasons to favor the exclusion or eradication of some species from the point of view of enhancing treatment. Although there are not general rules, some plant species have disadvantages for the treatment of some pollutants. As an illustration, the Everglades protection wetlands (STAs) of South Florida target removal of phosphorus to ultra-low levels (≈10–50 μg/L). These are also uniquely large systems with attendant peculiarities of wind fetch. Huge amounts of research have shown that submerged aquatic vegetation is more effective than emergent plants. However, the wetlands are easily and often invaded by species that are moderately effective in phosphorus reduction, such as water hyacinths (*Eichhornia crassipes*), hydrilla (*Hydrilla verticillata*), and torpedo grass (*Panicum repens*). Hyacinths are moved by the wind and form dense windrows piled above water, which clog structures and create anoxic conditions. Torpedo grass and hydrilla form floating leaf canopies that block sunlight and oxygenation. Accordingly, these plants were regarded as undesirable and removed by herbiciding. Several different chemicals have been used, including 2,4-D, glyphosate, imazipyr, and diquat dibromide (Goforth, 2005).

## Burning

The value of prescribed fire in managing vegetation varies according to region and vegetation type. Fire may be used in treatment wetlands for the purpose of reducing and removing excessive thatch buildups that develop in arid climate systems. However, those climatic regions may also be prone to air pollution problems, which may preclude fire as a management tool. Fire is being used on a limited basis in some constructed FWS treatment wetland systems (e.g., West Jackson County, Mississippi) to periodically reduce the density of macrophyte growth. In arid regions, fire is often regarded as a necessary management tool to reduce the thatch in SSF wetlands (Wallace and Knight, 2006).

Fires are typically set in the late fall or winter after plant senescence, and wetland burns can be accomplished with or without shallow standing water in the wetland cells.

Fire mobilizes the stored nutrients and pollutants into water-soluble forms and permits the flushing of the materials the wetland was designed to store (Smith *et al*., 2001). The litter and sediment layers may be destroyed, thus necessitating a new microbial start-up period. If fire is used as a management tool, it is essential that the treatment wetland be designed with multiple parallel flow paths to allow bypassing flows around the cells receiving controlled burns. Following a planned fire in a wetland cell, water should be held for a period of several weeks to a few months before start-up and discharge to avoid the release of newly dissolved constituents.

White *et al*. (2002) studied the effects of fire in a FWS treatment wetland, the Orlando Easterly System in Florida. The vegetation communities included cattail, bulrush, and mixed species. A paired cell design was implemented with a burn in one cell, whereas the adjacent cell received no burn. Water quality parameters including soluble reactive phosphorus, total phosphorus, ammonium, total Kjeldahl nitrogen (TKN), and dissolved organic carbon were measured before and up to four months after the burn. A pre-and post-burn quantification of aboveground biomass was conducted to determine the net reduction in biomass as a result of the burn. The results showed that there was phosphorus released from the burned cell immediately after the fire, but over time, the concentrations dropped below that of the no-burn treatment, showing greater performance of the burned cell for treatment of water column phosphorus.



**FIGURE 22.18** This berm at the Hayward, California, treatment wetland is need of maintenance because of erosion.

## 22.6 MAINTENANCE OF STRUCTURES

It is obvious that wetland operation involves keeping the physical features in good condition, including inflow and outflow structures, pumps, piping, berms, and levees. Two features of FWS treatment wetlands may be singled out as distinctively different from many other civil works: berm maintenance and structure clogging.

Berm damage can occur as a result of animal activity, as discussed in the preceding text (see Figure 22.17). Animal control may be required in addition to repairs. Vegetative damage is much less likely, but nuisance species may need removal or mowing. Erosion damage from wave or rain action will occur if the original design and construction did not provide adequate stabilization (Figure 22.18, also see Figure 18.20). Repair and restabilization with riprap or appropriate berm plantings is required.

FWS treatment wetlands export plant debris, primarily as floating detritus (see Figure 18.29). Blockage of maintenance manholes, weirs, outflow piping, and rodent guards by dead floating leaf litter from emergent vegetation, floating aquatic plants (duckweed and algae), and large turtles can play havoc when trying to maintain flow rates and water levels. At the Brighton, Ontario, treatment wetland, stems and leaves break off from the dense growth of cattail and river bulrush around the perimeter of their wastewater lagoon and, along with mats of floating algae and duckweed discharges from the lagoon, are passed through a maintenance manhole prior to entering the wetland. However, much of this material remained in the maintenance manhole as a floating mat and built up to the point where it became thick enough to completely block the flow to the wetland (Figure 22.19). The wetland outflow was plagued by similar problems. Regular buildup of dead floating vegetation and other debris on the effluent rodent guard resulted in a blockage that would impede the outflow. In both cases, a simple fix was the installation of a snow fence barrier at the lagoon and wetland outflows. This allows the debris to become caught up against the barrier but allows the water flow to continue below it. Debris is routinely cleaned off the snow fencing to maintain the flow and water level in the wetland. For large FWS treatment wetlands, the situation is more complex, because of the size of the structures. Automatic self-cleaning trash racks can be used to span inlets that are several meters in width (Figure 22.20). Removal of the debris may require truck transport.




**FIGURE 22.19** Plant debris buildup in a control structure manhole at the Brighton, Ontario, project. (Photo courtesy J. Pries.)

## 22.7 LONG-TERM PROSPECTS

Part of the natural function of treatment wetlands is the buildup of sediments and soils in the system. This mechanism provides accretion of both incoming solids (settling) and of internally generated solids (see Chapter 7).

### FWS Wetlands

Sedimentation eventually compromises the operation of the system in two ways. Firstly, the sediment-trapping forebay



**FIGURE 22.20** The automated trash rack at the effluent pump station for the Everglades Nutrient Removal Project. Rakes pull the clogging floating vegetation up the inclined bar screen and deposit it on a conveyor, which removes it to a stockpile. The drained material is removed by truck.

or inlet zone can become filled and no longer provide the needed vertical-settling depth (see Figure 7.13 of Chapter 7). Secondly, solids accrete internally to the vegetated zones, regardless of how effective a sediment forebay or pond may be, because of internal wetland generation of solids. As noted in Chapter 7, the rate of accretion may be on the order of 1 or 2 cm/yr. Mass balance calculations may then be applied to determine the "life" of the wetland, in terms of the time required to fill the basin to a certain depth.

There is a major problem with such average accretion computations: the deposition is not uniform. If significant treatment is occurring, there may be more deposition in the wetland inlet region; in other words, there is likely to be a longitudinal gradient in the buildup of sediments and soils. The water in the inlet region can become shallower and consequently move with higher velocity. Transverse nonuniformity also creates difficulties, because channels can form in the new sediment layers, especially under shallow high-velocity conditions. In combination, the wetland may lose some of its original hydraulic efficiency. These processes are exacerbated by the fact that new sediments are slow to consolidate and usually create a "floc" layer that is relatively mobile. Thus, accretion can compromise the operation of a treatment wetland long before the containment (freeboard) is jeopardized.

The maintenance needed to correct for accretion is daunting. The wetland (or flow path) must be shut down, and the excess material excavated and disposed. Afterward, vegetation must be reestablished. This procedure has been applied to a portion of the Orlando Easterly Wetland system, with good results (Sees, 2005; Wang *et al*., 2006). That system had been in operation for about 18 years at the time of rejuvenation. In early 2007, after approximately 15 years of operation, some cells of the ENRP were selected for removal of accreted sediments. There is presently insufficient information to generally evaluate the trade-off between reduced efficiency and the cost of renovation.

### SSF Wetlands

Deposition of particulate matter, plant detritus, and microbial biofilms also function as clogging mechanisms in subsurface flow wetlands. For HSSF wetlands, this is observed as a marked reduction of the hydraulic conductivity of the inlet zone (Figure 2.22), which eventually leads to clogging and overland flow (Figures 2.26, 2.28, and 7.25). Design approaches such as reducing the organic loading over the inlet cross section (Figure 21.5) may reduce or delay clogging, but there is no evidence that inlet zone clogging can be avoided, as sedimentation is a cumulative process and the majority of the particulate matter is deposited in the inlet region (Figure 7.25) (Puigagut *et al*., 2006). In a recent survey of 255 SSF wetlands in the United Kingdom, with a median age of about ten years, 52% were reported to exhibit some degree of overland flow (Cooper *et al*., 2006a). Based on the performance of existing HSSF wetlands, it appears that inlet zone maintenance will be needed approximately every ten years to avoid a large degree of overland flow.

From an operational standpoint, bed clogging can be diagnosed as overland flow in the inlet region, and adjustments to the effluent water level control structure will not correct the problem (Figure 22.21).

At the present time, means of dealing with bed clogging are in the experimental stage. The basic options appear to be:

- Remove the clogged-bed media and replace with new material. It is an expensive option (Table 22.6) because not only must the replacement media be purchased, but there will likely be disposal costs associated with the clogged media.
- Removing the clogged-bed media, washing it, and returning it to the wetland bed (Cooper *et al*., 2006a). This method may be preferable to purchasing new bed media, but the washing equipment is only recently available and there will likely

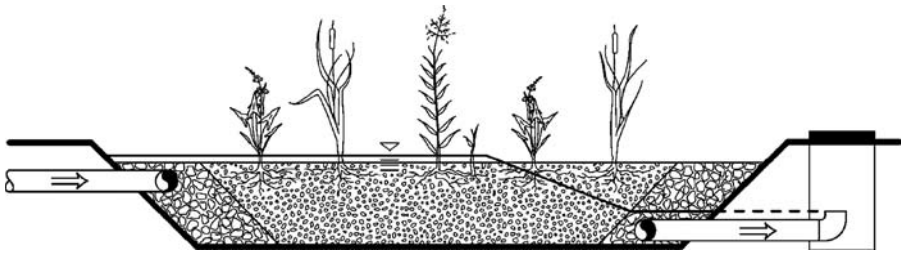



**FIGURE 22.21** Clogged-bed water level response in HSSF wetlands. (From Wallace and Knight (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements*. Final Report, Project 01-CTS-5, Water Environment Research Foundation (WERF): Alexandria, Virginia. Reprinted with permission.)

be disposal costs associated with the extracted biosolids.

- Using hydrogen peroxide ($H_2O_2$) to chemically oxidize the biosolids *in situ* (Hanson, 2002b; Behrends *et al*., 2006). In a study on a pilot-scale reciprocating bed wetland, Behrends (2006) attempted to rehabilitate the beds through the application of concentrated (30%) hydrogen peroxide at a dosage rate of approximately 9 L/m$^3$ of bed. Noticeable bubbling occurred, and sludge was redistributed, with some moving to the bed surface. No indications were given of improvement of either hydraulic conductivity or treatment efficiency. The success of this method has recently been demonstrated in Lake Elmo, Minnesota.

VF wetlands treating primary or secondary municipal wastewater will accumulate solids on the surface of the bed (Figure 7.30). However, VF systems that are treating waste streams inherently low in TSS, such as settled leachates, pretreated food wastes, or tertiary applications for municipal wastewater, will likely not form the same solids layer.

Accumulated solids must eventually be removed. Accumulation appears to be a function of the organic-loading rate and operational load-and-resting regime, and methods to quantify and predict solids accumulation have yet to be developed. Based on current information, a cleanout frequency of eight to ten years seems reasonable (Molle *et al*., 2004a) for the French raw sewage VF wetlands.

For biosolids dewatering wetlands, this accumulation is deliberate and part of the normal functioning of the system. The amount of dewatered sludge that can be accumulated in this way is a function of the available freeboard of the wetland cell, with accumulations of approximately 1.5 m over ten years of operation (Nielsen, 2002). The resulting material has a solids content of approximately 40% and is removed from the basin with an excavator or skid-steer loader. Some basins are constructed with concrete sidewalls to facilitate sludge removal (Figure 1.15).

**TABLE 22.6**
**History of Inlet Clogging for Selected HSSF Wetlands in Minnesota**

| System | Media $d_{10}$ (mm) | Cross-Sectional Organic Loading (g BOD/m$^2$·d) | Approximate Time for Inlet Zone Surfacing to Develop (years) | Cost of Inlet Zone Maintenance |
|---|---|---|---|---|
| St. George[a] | 2.1 | 448 | 2 | 19% |
| Spring Hill[b] | 5.0 | 353 | 4 | — |
| Prairie Hamlet[c] | 5.1 | 345 | 6 | — |
| Fields of St. Croix[b] | 4.0 | 304 | 6 | — |
| Jackson Meadow[a] | 5.8 | 287 | 6 | 10% |
| Lutsen[c] | 5.2 | 189 | 9 | — |

[a] Inlet zone maintenance is expressed as a percentage of the initial construction cost. Maintenance for these systems consisted of excavating clogged material, replacing with clean rock, lime stabilization to eliminate pathogens from the clogged rock, and reconfiguration of the inlet zone distribution to reduce the cross-sectional organic loading.

[b] Inlet zone maintenance was done on these systems, but cost data is not available.

[c] Some inlet clogging observed, inlet zone maintenance has not been required to date.

## 22.8 ASSISTANCE TO THE OPERATOR

It is not only the wetland itself that goes through start-up. Very often the system operator also goes through a kind of start-up, during which there is a steep learning curve. Familiarity with treatment wetlands in general is a prerequisite qualification for the wetland operator, but in many circumstances that may be "book knowledge." In virtually all cases, the operator will not have been involved in the wetland design, and therefore will not be familiar with the intended function of the system. Two steps can alleviate potential difficulties: (1) a start-up walk-through of the treatment system, from pretreatment to final discharge, and (2) preparation of a comprehensive operations and maintenance manual. The walk-through includes a discussion between the designer and operator, of the intended function of structures and controls, and of the wetland vegetation, detritus, and soils. It serves the purpose of defining what activities may be required on a seasonal basis.

The project operation and maintenance (O&M) manual is the reference document that the operator may consult as needed during future operation of the facility (Tables 22.7 and 22.8). It describes the design intent of the project. Start-up and normal operations are explained. Monitoring requirements, protocols, and reporting are listed. Routine O&M steps are outlined. Some of the foreseeable emergency situations, notably high-rainfall events, are discussed, and emergency measures defined. For later reference, it includes the as-built drawings for the wetland and the current regulatory permits.

For large projects, there may be a requirement for a project operation plan. For example, the stormwater treatment areas (STAs) of South Florida all have such documents. A typical outline for STA6 (SFWMD, 2004) is

I. Project description
II. Structure and canal descriptions
III. Operations
IV. Downstream effects
V. Additional flows
VI. Coordination with other plans
VII. Permits for operation

**TABLE 22.7**
**Example Table of Contents Sections for an FWS Wetland Operations and Maintenance Manual**

**Introduction**
Background
User's guide
Treatment wetland processes

**Unit Processes**
Inflow control and measurement
Wetland flow
Wetland function in shallow zones
Wetland function in deep zones
Outflow and level control

**Start-Up Procedures**
Treatment wetland
Planting
Start-up and start-up flow diagram

**System Operation**
Operations
Winter operations
Mosquito control
Sampling procedures
Grab sampling
Sampling and monitoring
Start-up period
Steady state sampling
Laboratory tests
Recording data
Daily/weekly operating logs

**Monitoring**
Sampling procedures
Grab sampling
Sampling and monitoring
Start-up period
Steady state sampling
Laboratory tests
Recording data
Daily/weekly operating logs
Laboratory records
Monthly reporting
Annual report
Maintenance records
Operating costs
Emergency conditions
Trend charts
Laboratory records
Monthly reporting
Annual report
Maintenance records
Operating costs
Emergency conditions
Trend charts

**Treatment Wetland Maintenance**
Process control problems
Water control structures
Short-circuiting and deep zones
Vegetation
Embankments
Piping
Warranty provisions

**Emergency Plans and Procedures**
Hydraulic overload
Process failure
Safety

**Appendices**
A. Design criteria
B. Treatment wetland design drawings
C. Sample volumes and storage requirements
D. Operating log

*Source:* From J. Pries, CH2M Hill, Kitchener, Ontario; for the Nanticoke FWS wetland.

**TABLE 22.8**
**Selected Table of Contents from a HSSF Wetland Operations and Maintenance Manual, Cambridge–Isanti School District, Minnesota**

**HSSF Wetland O&M Topics**

- Inlet zone surfacing
- Water level adjustment
  - Winter
  - Summer
- Freeze protection of hydraulic control structures
- Control of rodents
- Erosion repairs
- Liner repair procedures
- Vegetation maintenance
  - Desirable plant species
  - Listed noxious weeds and control methods
- Nitrogen processing
- Operational adjustments
- Sampling methods
- Operating logs
- Reporting monitoring results

**Appendix A:** Design basis
**Appendix B:** Record drawings
**Appendix C:** Operator certification requirements
**Appendix D:** Example O&M checklists

The operations section of the overall plan discusses various modes of operation. The design intent for normal operation is specified, as to which structures are operating and what water depths are to prevail. However, the design included operations during the "standard project storm" of 55 cm of rain over a 24-hour period. The operating plan, therefore, includes information on extreme flow operations. The design calculations also indicated frequent dry periods, and the operations plan includes provisions for dealing with this lack of water. Should it become necessary to take treatment cells out of service (this has not happened in the first ten years of project operation for STA6), the document contains the design intent for earthwork to accommodate the temporary reconfiguration. The design of the project also contemplated expansion of the treatment wetland, by adding a third flowpath, and that intent is spelled out in the operation plan.

## SUMMARY

Treatment wetlands are sufficiently different from other treatment technologies to make learning their operation a nontrivial effort. Although they involve few sophisticated controls, they offer hydraulic and ecological challenges. Thus, although they are "low tech," they are "high bio." Given a conservatively designed wetland treatment system, successful operation is not a challenging task. There are few operational controls in most treatment wetlands, and these controls must be used wisely and with common sense to maintain top performance. Typically, the only system controls available are variation of inflow-loading rates through storage or diversion of flows, system rotation between alternate discharge locations or wetlands, and control over water levels within the wetland.

A certain level of routine maintenance measures is necessary, including mowing, structure cleaning, and berm inspection. Vegetation management might include depth management, selective herbiciding, and the controlled use of fire. Animal control is increasingly understood to be an important facet of FWS wetland operation. In particular, rodents can be quite destructive and require control in some circumstances. Mosquito control is an important issue in some wetland treatment projects, notably those in warm, arid climates. The owner and operator of these wetland systems should be prepared to deal with this possible issue.

It is becoming apparent that HSSF wetlands will require declogging one or more times during a 20-year life span. Methods for removing solids are currently under development.

Monitoring is one of the most important aspects of treatment wetlands operation. Monitoring of inflow and outflow water quality provides a system-level barometer of wetland health and performance. Monitoring of the internal wetland structure provides a reference for correlating changes in water quality performance with system structure. Routine monitoring and data analysis are essential for making informed decisions concerning control of operational variables such as water depth and mass loadings. Additional monitoring beyond regulatory requirements may be performed to accomplish specific operational goals. Because of its importance in optimizing system performance and averting unacceptable biological changes, some monitoring should be conducted in all treatment wetlands.

Although the totality of potential management issues is large, all of the issues identified in this chapter do not occur at any one wetland. In general, treatment wetland systems are operator-friendly, but they cannot be left unattended for prolonged periods.

# 23 Economics

The economics of treatment wetlands consists of two major factors: capital costs and operating costs. The capital cost components of free water surface (FWS) and subsurface flow (SSF) wetlands are essentially the same, except for the cost of the gravel required for SSF wetlands (Campbell and Ogden, 1999; U.S. EPA, 2000a). However, SSF systems have generally been implemented for smaller flows than FWS systems. For instance, the system database compiled by the Water Environment Research Foundation (WERF) has 214 FWS wetlands with a median design flow of 1,050 $m^3/d$, 707 HSSF wetlands with a median design flow of 9.5 $m^3/d$, and 566 VF wetlands with a median design flow of only 2.1 $m^3/d$ (Wallace and Knight, 2006). System areas from the WERF database show 330 FWS wetlands with a median area of 1.6 ha (16,000 $m^2$), 710 horizontal subsurface flow (HSSF) wetlands with a median area of 140 $m^2$, and 544 vertical flow (VF) wetlands with a median area of 44 $m^2$. Therefore, SSF wetlands do not enjoy the economy of scale experienced by FWS wetlands. There is now enough information to determine approximate capital cost functions that represent these scale effects.

Because wetland systems are constructed using local labor and local materials, it is not possible to offer precise universal cost estimates that will apply to all treatment systems. Generally, the basic components of a wetland treatment system—earthwork, gravel (in the case of SSF wetlands), liners, and plants—are produced in regional markets that are distance sensitive. For instance, the installed cost per cubic meter of gravel is highly dependent on the distance between the source of supply (a local gravel pit) and the site of wetland construction. Labor costs are also highly variable. To assess the feasibility of a wetland treatment system, local cost figures should be used to compare the capital and operating costs of a wetland system against that of other treatment technologies.

Within the United States, Construction Cost Indices (CCI) are published by the Engineering News Record (ENR). These cost indices track inflationary changes within the construction industry over time. The ENR CCI started at 100 in the year 1913 and has increased to 7,856 in March 2007. Cost indices are available as a national average, but are also published for 20 major metropolitan areas in the United States. For the purposes of this chapter, costs are based on the 2006 United States national average ENR CCI of 7,751. As a side note, construction in high-cost metropolitan areas is almost double that of low-cost metropolitan areas.

In general, capital costs of treatment wetlands are comparable to alternative technologies for accomplishing the same task. However, the costs of operating a treatment wetland are typically much lower than for competing technologies. Mechanical devices are always more energy intensive, and will always be more expensive to operate, than a passive wetland system (Type *A*) (Brix, 1999). The basic exchange is land for energy (Campbell and Ogden, 1999). As a consequence, the lifecycle cost of a wetland project, as represented by the present worth of capital and operating expenses, is very often quite favorable compared to alternative treatment technologies.

Operating costs can be quite low, especially for Type *A* passive systems. Energy costs are typically close to zero for gravity-driven FWS wetlands, and are generally low for all types of treatment wetlands (see Table 1.1). Water quality monitoring is often a principal part of O&M costs. However, operation and maintenance (O&M) costs can become appreciable if it is attempted to maintain specific vegetation types, thus encountering "weeding" costs.

## 23.1 CAPITAL COSTS

Although it is not possible to offer universal cost guidelines, every system shares a similar set of construction components. Therefore, it is possible to estimate the cost of each component within a regional market. The basic direct cost components of a wetland treatment system include:

- Land
- Site investigation and system design
- Earthwork
- Liners
- Media
- Plants
- Water control structures and piping
- Site work (site preparation, fencing, access roads, etc.)
- Human use facilities

These costs include material, labor, overhead, and profit, and represent the contractors installed cost. Additionally, there are indirect costs associated with permitting, engineering, financing, mobilization, and construction management. In general, these costs are all incurred prior to system start-up. Detailed estimates are usually made after final sizing and siting. More precise economic estimating is possible after final design drawings have been prepared.

### Regional Variation

Economics vary geographically because of differing unit costs, and because of differences in the selection of materials

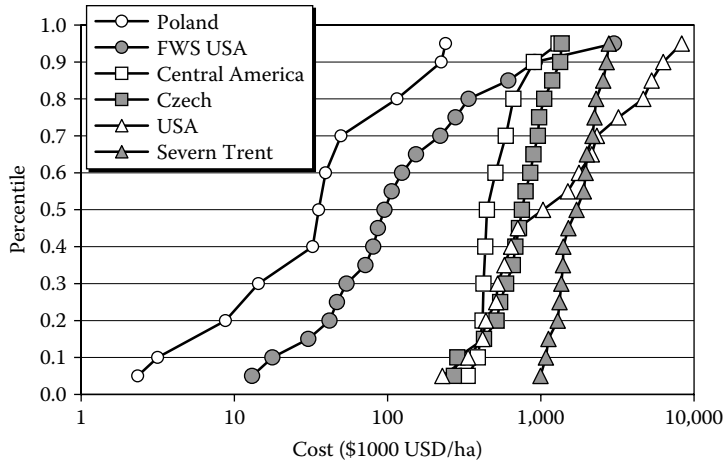



**FIGURE 23.1** Capital costs distribution for treatment wetlands. All are HSSF systems except FWS (United States). Costs adjusted to 2006 using CCI = 7,751 and the 2006 exchange rate. Data for Poland from Kowalik and Obarska-Pempkowiak (1998) In *Constructed Wetlands for Wastewater Treatment in Europe*. Vymazal *et al*. (Eds.), Backhuys Publishers, Leiden, The Netherlands, pp. 217–225. Data for the Czech Republic from Vymazal (1996) *Ecological Engineering* 7: 1–14. Data for Severn Trent from Green and Upton (1994) *Water Environment Research* 66(3): 188–192. Data for Central America from Platzer *et al*. (2002) *Investigations and Experiences with Subsurface Flow Constructed Wetlands in Nicaragua*, *Central America*. Mbwette (Ed.). Proceedings of the 8th International Conference on Wetland Systems for Water Pollution Control, 16–19 September 2002, Comprint International Limited: University of Dar Es Salaam, Tanzania, pp. 350–365. Data for the United States: Various.

and other design features. The cost of labor and materials within a particular regional market plays a large role in the cost of a wetland treatment system. Cost differentials are even greater when comparing across worldwide geographic locations and their accompanying economies. The effect of regional market cost factors is illustrated in Figure 23.1, which demonstrates treatment wetland capital cost distributions for various locations. HSSF wetland systems in Poland (Kowalik and Obarska-Pempkowiak, 1998) exhibit lower capital costs than those in Nicaragua (Platzer *et al*., 2002) or the Czech Republic (Vymazal, 1996). Severn Trent tertiary systems in the United Kingdom (Green and Upton, 1994) are more expensive than those in the Czech Republic; HSSF systems in the United States have a distribution of capital costs that spans the range from the inexpensive Czech systems to those that are more expensive than the U.K. systems.

Some of the regional differences have to do with design sizing criteria. Different criteria are used in different countries, and also at different times in the same country. Others have to do with structural specifications. For instance, a number of U.K. systems are sited in basins lined with brick or concrete, and some have stainless steel level and flow control elements. Clearly, such systems will have greater capital costs than those sited in earthen basins with plastic piping and simplified flow control elements.

There are also economies of scale, which will be addressed in a subsequent section. However, it is useful to first examine the various components of capital costs in more detail.

## Direct Costs

### Land

Land costs are highly site specific. Information on land availability and land costs is generally obtained with the assistance of real estate professionals who are familiar with local market conditions. Wetlands are more land intensive than many other wastewater treatment processes. If land needs to be purchased for the project, this can be a significant cost component. In the United States, treatment wetland land purchase prices have ranged from $3,000/ha in remote locations with low population density and low agricultural utility to over $100,000/ha in urbanizing agricultural landscapes. Land acquisition in or near urban areas is sometimes viewed as preservation of green space, and valued for ancillary benefits.

Land costs can be a significant fraction of the total capital cost. For example, Wossink and Hunt (2003) identify three categories of land cost for urban stormwater wetlands in North Carolina: zero cost because the project is required for community green space, $125,000/ha for vacant land that may be used for residential development, and $550,000/ha for land that may be used for commercial development.

As will be discussed, land costs occupy a special role in the present worth analysis of a wetland system.

### Site Evaluation

The design and construction of a wetland requires that site characteristics be well understood—including soils,

groundwater elevations, and site topography. These activities usually precede or accompany engineering design, but are additional to that design.

### *Topographic Survey*

The ground elevation of the site is a critical factor in design, because it controls the cut-and-fill calculations that normally dictate the elevations of berms and bottoms. Associated with balancing of earth import and export is the issue of the potential for gravity flow. Sometimes proper cell elevations can eliminate the need for one or more pumps, and thus may interact with cut-and-fill considerations. The topographic survey will influence the direction of flow and the sequencing of cells, and the need for long or short distribution and collection canals, which for large systems can influence the project cost. The potential need to level cell bottoms for purposes of evening the hydraulic flow distribution can only be evaluated with detailed site topography. The shallow water depths in wetlands, especially large FWS systems, creates a need for accurate as-built topography as an aid to understanding water depth and movement. The costs for such surveys are typically \$50/ha–\$500/ha, depending on scale and grid size requirements. For instance, the topographic survey for the Incline Village, Nevada, treatment wetland cost \$370/ha for the 175-ha wetland, adjusted to 2006. However, the survey work for cell 4 of STA2 of the Everglades phosphorus control project required only \$45/ha.

### *Geotechnical Investigations*

Many small-scale wetland projects are designed in conjunction with soil infiltration systems. In these situations, it is common to use shallow soil borings or backhoe pits to determine local soil characteristics. Costs for these initial site investigations vary with the size of the project. In the U.S., the cost for a site investigation can range from a few hundred dollars (for a single-home system) up to several thousand dollars (for a system serving a small community).

Larger-scale projects, including those that discharge to surface waters, also require soil investigations. Even though such large projects rarely require liners, there is a need to assess the potential for seepage from the project, and hence the need for seepage collection canals. It is critical to determine if site soils are adequate for the berms and levees, and if so how much compaction may be required, and how much allowance for subsidence.

Soils that may be used for rooting media in FWS systems, as well as the gravel substrate for SSF systems, may need to be assessed for contaminants of concern, including nutrients. A modest amount of chemical testing may be required to identify any potential problems, or to form the basis for forecasting the sorption life expectancy for contaminants.

### *Hydrogeological Investigations*

The location of the groundwater table and direction of groundwater movement can be a critical factor in wetland design. For small community and on-site systems involving soil infiltration, the depth to groundwater is an important consideration. For "normal" infiltration beds (septic fields), there is a required minimum unsaturated depth that is likely to carry over to the wetland/infiltration situation.

For a wetland with a liner, it is also necessary to know the depth to groundwater, because rising, shallow water tables can lift a synthetic wetland liner, displacing air from the soil environment. This has the potential to create large underliner bubbles that push the liner up out of the water; these are commonly called "whale backs." The solution to such potential difficulties is the construction of an under-drain system beneath the wetland liner that vents air and controls the level of the water table. The need for this feature, and its associated cost, will be determined through site-specific hydrogeological investigations.

In some cases, there may be a concern for regional use of the groundwater as a potable water supply. The treatment wetland might be viewed as a possible source of contamination if partially treated water entered into the unprotected drinking water aquifer, and moved to the wells that withdraw potable water. This would usually occasion the need for a hydrogeological study to ascertain the depths and directions of regional groundwater flow, and the consequences of even small leaks from the treatment wetland to the water quality of the aquifer. This was the case at the Columbia, Missouri, project. The hydrogeological study included calibration and modeling to address this issue, at a cost of \$750/ha for the 36-ha wetland, adjusted to 2006 (Brunner *et al.*, 1992).

## Earthwork

The construction of a wetland treatment system requires excavation and grading of the site to produce level basins that are enclosed by earthen berms. For small systems (generally less than 0.05 ha) backhoes or similar types of excavation equipment are commonly used. Larger basins are generally constructed using bulldozers or construction scrapers. Area-specific earthwork costs are the product of two components. The first component is the cost to move earth, which is a volumetric (per cubic meter) cost. This cost component is a function of equipment costs, labor costs, and the source of earth supply (on-site or imported). The second component is the amount of earth that must be moved to grade the site. This areal requirement ($m^3$ of earth per $m^2$ of wetland area) is a function of the project site conditions. Earthwork costs are lowest on flat sites that require minimal grading and have suitable soils on-site (low areal grading requirement and volumetric cost). Sloping a site requires terracing, which increases earthwork costs (due to the large areal grading requirement). If the fill needs to be imported, earthwork costs will increase (due to the high volumetric cost). Earthwork construction techniques are essentially the same for FWS and HSSF wetlands, and hence volumetric earthwork costs for the two types should be comparable for similar-sized wetlands.

**TABLE 23.1**
**Earthmoving Costs for HSSF Wetlands in Minnesota**

| System Name | Design Flow ($m^3/d$) | Wetland Area ($m^2$) | Earthwork Volume ($m^3$) | Cost Volumetric ($/$m^3$) | Grading Req. ($m^3$ earth per $m^2$ wetland) | Cost (Areal) ($/$m^2$) |
|---|---|---|---|---|---|---|
| St. George, Minnesota | 25 | 595 | 631 | 8.54 | 1.06 | 9.06 |
| Darfur, Minnesota | 38 | 1,301 | 383 | 5.70 | 0.29 | 1.68 |
| Northern Tier High Adventure Base, Minnesota | 34 | 297 | 306 | 13.56 | 1.03 | 13.97 |
| Lakes of Fairhaven, Minnesota | 59 | 1,828 | 383 | 7.56 | 0.21 | 1.58 |
| Delft, Minnesota | 22 | 664 | 306 | 18.15 | 0.46 | 8.36 |
| St. Croix Chippewa, Wisconsin | 251 | 6,141 | 6,885 | 4.51 | 1.12 | 5.06 |
| Prinsburg, Minnesota | 206 | 4,094 | 7,069 | 2.17 | 1.73 | 3.75 |
| **Median Values** | | | | **7.56** | **1.03** | **5.06** |

### *Clearing and Grubbing*

If the undeveloped site has undesirable vegetation, buildings, or other existing features that are incompatible with the wetland, these will need to be removed as part of the earthwork process. Brush and trees were removed from the sites at Ouray, Colorado; Sorrento, Louisiana; and West Jackson County, Mississippi, at an average cost of $9,800/ha. On larger projects, roads and ditches may require degrading, and buildings may require removal.

### *FWS Wetlands*

In a study of two municipal FWS wetland systems (West Jackson County, Mississippi and Gustine, California) the U.S. Environmental Protection Agency (EPA) estimated a volumetric cost of $10.80/$m^3$ when adjusted to 2006 USD (U.S. EPA, 2000a). The 20.2-ha wetland system in West Jackson County, Mississippi, had an areal grading requirement of only 0.26 $m^3/m^2$, and the 9.7-ha system in Gustine, California, had an areal grading requirement of 0.35 $m^3/m^2$.

Larger treatment wetlands require far less earthmoving on a per-hectare basis than do small systems. For instance, Cell 4 of STA2 of the Everglades phosphorus control project required only 0.056 $m^3/m^2$ of earthmoving based on the 816-ha wetland footprint. However, the construction of such a large system does not involve scraping a thin layer from the entire footprint. Rather, the inlet spreader canal, outlet collection canal, and seepage return canal are the source of the fill material for the containment levees.

Even large wetlands may require significant earthmoving if built on sloping terrain. For example, the Inman Road treatment wetland in Clayton County, Georgia, has 22 wetted hectares in a terraced arrangement of 22 cells. The site presented slopes of 2–25%, and about 30 m of vertical variation (Inman *et al.*, 2001, 2003). Construction required moving 420,000 $m^3$ of earth, or 1.9 $m^3/m^2$.

Regional factors and site conditions can alter the cost of earthmoving. Cell 4 of STA2 of the Everglades phosphorus control project moved 460,000 $m^3$ of material, in a nearly balanced cut and fill. The breakdown of per cubic meter costs was: $2.61 to blast rock, $2.13 to excavate, and $2.09 to build levees, for a total of $7.96/$m^3$ (adjusted to 2006 USD).

### *HSSF Wetlands*

Earthwork costs for some HSSF wetland systems in the Minnesota–Wisconsin regional market are summarized in Table 23.1. Data from these seven HSSF wetland systems clearly show the impact of local site conditions on earthwork costs. Areal grading requirements varied from 0.21 $m^3/m^2$ to 1.73 $m^3/m^2$, with a median value of 1.03 $m^3/m^2$.

Volumetric earthwork costs for the systems in Table 23.1 varied between $2.17/$m^3$ and $18.15/$m^3$, with a median value of $7.56/$m^3$ (cost adjusted to 2006 USD, ENR CCI = 7751). Combining areal grading requirements with volumetric earthwork costs resulted in areal earthwork costs ranging between $1.68/$m^2$ and $13.97/$m^2$, with a median value of $5.06/$m^2$ (adjusted to 2006 USD).

## Liners

The decision to install a liner in a wetland system, and which type of liner to use, depends on the project goals, regulatory requirements, and feasibility. Some wetlands are unlined, either because the *in situ* native soils are deemed to have sufficient sealing properties, or because groundwater recharge is a function of the system. Very large FWS wetlands cannot be plastic lined because it is not feasible for systems of more than a few hectares, but clay lining has been implemented on systems up to 40 ha, such as Columbia, Missouri. If the wetland must be lined, there are a variety of liner materials available. The two most common liner materials used are 0.76-mm polyvinyl chloride (PVC) and 1.0-mm high-density polyethylene (HDPE). PVC liners are generally factory seamed, one-piece liners used on small projects less than 0.1 ha in size. HDPE liners generally come in rolls and are field seamed for larger projects.

The total *installed* cost of the liner includes not only the material cost but also the labor cost associated with field seaming, seam testing, material inspection, and leak testing. If local soil conditions include sharp or angular rocks,

**TABLE 23.2**
**Approximate Cost of Installed Liners (2006 USD)**

| Material | Thickness | Installed Cost ($/m²) |
|---|---|---|
| Bentonite | 10 kg/m² | 7.96 |
| Native clay[a] | 30 cm | 7.50 |
| Clay geotextile sandwich | NS | 4.84 |
| Polyvinyl chloride | 0.76 mm | 4.09 |
| High-density polyethylene | 1.02 mm | 4.73 |
| Polypropylene | 1.02 mm | 5.92 |
| Reinforced polypropylene | 1.14 mm | 6.89 |
| Hypalon | 0.76 mm | 6.89 |
| Hypalon | 1.52 mm | 8.07 |
| XR-5 | NS | 11.19 |

[a] Assumes $25/m³ delivered, placed, and compacted.

*Source:* Data from U.S. EPA (2000a) *Constructed wetlands treatment of municipal wastewaters*. EPA 625/R-99/010, U.S. EPA Office of Research and Development: Washington D.C.; and Interstate Technology and Regulatory Council (2003) Technical and Regulatory Guidance Document for Constructed Treatment Wetlands. *http://www.itrcweb.org/WTLND-1.pdf.*

a layer of sand or other granular material may have to be placed before the liner can be installed. If the gravel used to line the bed has sharp or angular edges, it may be necessary to cover the interior of the liner with a protective layer of geotextile fabric, since sharp rock can puncture a liner. Table 23.2 shows approximate installed costs of a variety of liner materials.

Liner costs for the FWS wetland system at Ouray, Colorado, were $6.97/m², and $13.23/m² for the HSSF wetland system at Ten Stones, Vermont (both adjusted to 2006 USD) (U.S. EPA, 2000a). Liner and geotextile fabric costs for 12 HSSF wetlands in the Minnesota–Wisconsin regional market are summarized in Table 23.3. Data from these wetland systems indicate that installed liner costs (adjusted to 2006 USD; ENR CCI = 7751) ranged from $3.74/m² to $19.71/m², with a median cost of $8.66/m². The use of a geotextile fabric will increase the installed liner cost. The data in Table 23.3 indicate that the installed cost will increase by approximately $3.00/m² when a geotextile fabric is used.

If soil at the site contains rocks or other debris that could damage the liner during installation, a layer of sand or similar granular material is often placed prior to installing the liner. For example, 8 cm of sand was placed prior to installing the liner in three Minnesota HSSF wetlands. The installed cost for sand bedding ranged from $1.38/m² to $3.08/m², with a median cost of $2.24/m² (2006 USD; ENR CCI = 7751).

## Media and Mulch

The major cost variation between FWS and SSF wetlands results from differences in the rooting media. FWS systems typically use 30–40 cm of soil, which may be entirely available on the site, whereas SSF wetlands use 50–80 cm of gravel or other similar media, which must usually be purchased and transported to the site.

### *Soils for FWS Systems*

If *in situ* native soils are suitable, they are typically used as the rooting substrate for emergent wetland plants in FWS systems. If *in situ* native soils are unsuitable for use as rooting media, soil may be imported and mixed with existing soil to create conditions adequate for rooting wetland plants. For these reasons, the cost of the rooting medium is usually reflected in the volumetric earthwork costs. However, it is sometimes advantageous to add organic material to the rooting soil for a FWS system, to provide sorption capacity immediately upon start-up (Figure 23.2). Such a blending

**TABLE 23.3**
**Liner and Geotextile Costs for Example HSSF Wetlands**

| System Name | Design Flow (m³/d) | Wetland Area (m²) | Liner ($/m²) | Geotextile ($/m²) |
|---|---|---|---|---|
| St. George, Minnesota | 25 | 595 | 8.04 | — |
| Darfur, Minnesota | 38 | 1,301 | 7.36 | 3.35 |
| Northern Tier High Adventure Base, Minnesota | 34 | 297 | 8.93 | 1.84 |
| Opole, Minnesota | 35 | 725 | 6.57 | 3.94 |
| Tamarack, Minnesota | 26 | 418 | 19.71 | 3.29 |
| Lakes of Fairhaven, Minnesota | 59 | 1,828 | 6.84 | 3.02 |
| Delft, Minnesota | 22 | 664 | 13.70 | 12.45 |
| Cedar Mills, Minnesota | 35 | 1,073 | 9.95 | 1.76 |
| St. Croix Chippewa, Wisconsin | 251 | 6,141 | 3.74 | 0.82 |
| Prinsburg, Minnesota | 206 | 4,094 | 10.19 | — |
| Mulberry Meadows, Minnesota | 72 | 1,580 | 8.39 | 2.47 |
| Cambridge-Isanti School District, Minnesota | 39 | 1,196 | 9.07 | 3.13 |
| **Median Value** | | | **8.66** | **3.08** |

*Note:* Geotextile is a nonwoven, needle-punched polypropylene material (230 g/m² fabric weight) used as a protective layer on top of the liner.



**FIGURE 23.2** Organic materials may be added to the rooting soils to promote early sorption potential. At the Hillsdale, Michigan, site, soils were borrowed from an adjacent floodplain, creating a pond and island complex in the borrow site.

operation may involve approximately 10 cm of amendment material, thus adding 0.1 $m^3/m^2$ of earthmoving to the project, plus the cost of the composting material. Municipal yard waste compost has been successfully used at Isanti-Chisago, Minnesota, and at Saginaw, Michigan, treatment wetlands. The use of purchased topsoil, either entirely or as an amendment, is a very expensive option, because high-quality (horticultural) topsoil typically sells for up to $\$25/m^3$.

### *Media for SSF Wetlands*

A number of innovative types of media have been used in SSF treatment wetlands. Examples include recycled glass fragments, such at Millersylvania State Park, Washington, blast furnace slag (Mann and Bavor, 1993; Drizo *et al.*, 1999), and lightweight expanded clay aggregates (LECA) (Zhu *et al.*, 1997; Jenssen and Krogstad, 2003). The cost of LECA is quite high; for instance, Scholz *et al.* (2001) report $\$204/m^3$ (2006 costs), but it is available in large units (Figure 23.3).




**FIGURE 23.3** Light expanded clay aggregate (LECA) can be purchased in large units in Europe.

The wide variety of SSF wetland media leads to wide variations in costs, but gravel is perhaps the most common material used in Europe and North America (Figure 23.4).

In a HSSF wetland, the rooting medium comprises the material used in the main wetland bed (typically gravel) and the coarser material used in the inlet and outlet zones (typically coarse rock). Costs for these materials are a function of regional market conditions as well as the distance between the source of supply and the project site. For instance, the 4.0-ha VF wetland at Connell, Washington, utilized 36,000 $m^3$ of 2.6-mm coarse sand that was available immediately adjacent to the bed, thus incurring only a cut-and-fill cost (Burgoon *et al.*, 1999). That situation is extremely rare, and most often media will be mined, screened, washed, and transported to the wetland site at considerable cost.

Media costs for 12 selected HSSF wetlands in the Minnesota–Wisconsin regional market are summarized in Table 23.4. This region was glaciated and typically has ample sources of gravel within a reasonable distance of a project. Data indicate that installed gravel media costs (used in the main portion of the bed) range between $\$15.95/m^3$ and $\$70.26/m^3$, with a median cost of $\$41.87/m^3$ (adjusted to 2006 USD). The gravel used in these systems was 9–25 mm in size. Within the Minnesota–Wisconsin regional market, distances between the source of supply (local gravel pit) and the project site were the primary factor in determining unit costs. As the depth of the bed media is established during the design process, calculation of areal costs is relatively straightforward. All of the HSSF systems in Table 23.4 were designed with a bed depth of 0.45 m, resulting in a median areal cost of $\$18.84/m^2$.

Media costs presented in Table 23.4 are higher than those presented in the literature for other HSSF systems. For three HSSF wetlands (Mesquite, Nevada; Carville, Louisiana; and Ten Stones, Vermont), U.S. EPA (2000a) reported bed media costs of \$14.33, \$23.58, and $\$14.74/m^2$, respectively (2006 USD). These were deeper beds (0.8, 0.75, and 0.75 m, respectively), and the volumetric cost of the media was less.



**FIGURE 23.4** Gravel media being placed at the Grand Lake, Minnesota, HSSF wetland.

### *Mulch*

In cold-climate regions, it may be necessary to insulate a HSSF system. This may be done with straw, or cover blankets for small systems. Mulch may be used as an insulating layer for cold-climate wetlands, and is a common design feature of HSSF wetlands in Canada and the northern regions of the United States (Figure 23.5). All of the wetlands in Table 23.4 were insulated with 0.09 m of reed-sedge peat. Installed costs for this peat material (2006 USD) ranged between \$23.94/$m^3$ and \$82.98/$m^3$, with a median cost of \$49.89/$m^3$. This converts to an areal cost of \$4.49/$m^2$.

### *Coarse Stone*

The berm slopes of FWS wetlands may be armored against burrowing animals and wave erosion by the use of rip rap. Coarse rock is a common choice (see Figure 18.19), but concrete matting has also been used (see Figure 18.23).

HSSF wetlands commonly use a coarser material (drain rock) in the inlet and outlet portions of the bed (Figure 23.6). In the United Kingdom, the stone inlet and outlet zones are typically 0.5 m wide, at full bed depth, and packed with 50–200 mm stone (Cooper *et al.*, 1996). Drain rock used in the 12 HSSF wetlands summarized in Table 23.4 was 20–75 mm in size. Installed drain rock costs ranged from \$25.53/$m^3$ to \$74.41/$m^3$, with a median cost of \$47.59/$m^3$ (adjusted to 2005 USD). U.S. EPA (2000a) reports that costs for outlet materials were \$10.39/$m^3$ (50-mm stone) and \$24.24/$m^3$ (100-mm stone) for the HSSF wetland at Ten Stones, Vermont (2006 USD).

## Plants

The plant component of capital cost varies according to the method chosen for vegetation establishment. It is presupposed that the media is in place, either the soil layer in a FWS system or the gravel in an SSF system. The FWS

**TABLE 23.4**
**Media Costs for Selected HSSF Wetlands**

| System Name | Design Flow ($m^3$/d) | Wetland Area ($m^2$) | Gravel (\$/$m^3$) | Mulch (\$/$m^3$) | Coarse Stone (\$/$m^3$) |
|---|---|---|---|---|---|
| St. George, Minnesota | 25 | 595 | 31.32 | 82.98 | 48.00 |
| Darfur, Minnesota | 38 | 1,301 | 45.56 | 81.36 | 47.18 |
| Northern Tier High Adventure Base, Minnesota | 34 | 297 | 70.26 | 76.64 | 74.41 |
| Opole, Minnesota | 35 | 725 | 15.95 | 49.47 | 31.11 |
| Tamarack, Minnesota | 26 | 418 | 23.94 | 23.94 | 25.53 |
| Lakes of Fairhaven, Minnesota | 59 | 1,828 | 33.27 | 45.36 | 52.92 |
| Delft, Minnesota | 22 | 664 | 42.33 | 37.80 | 36.30 |
| Cedar Mills, Minnesota | 35 | 1,073 | 39.83 | 64.02 | 52.65 |
| St. Croix Chippewa, Wisconsin | 251 | 6,141 | 41.40 | 32.44 | 30.31 |
| Prinsburg, Minnesota | 206 | 4,094 | 45.53 | 55.48 | 52.65 |
| Mulberry Meadows, Minnesota | 72 | 1,580 | 51.65 | 46.22 | 51.66 |
| Cambridge-Isanti School District, Minnesota | 39 | 1,196 | 46.22 | 50.30 | 44.87 |
| **Median Value** | | | **41.87** | **49.89** | **47.59** |



**FIGURE 23.5** Peat mulch in place at the Grand Lake, Minnesota, wetland.

wetland offers the largest number of options, including natural recruitment, seeding, and planting.

### *Natural Recruitment*

This option is the least costly but the least controllable, and usually of the longest duration. It is not free from cost, because moist soil conditions must be maintained, which in turn requires water management on the wetland. Virtually all of the Florida stormwater treatment wetlands (STAs) were established in this way. A newer strategy for these STAs is the use of a seeded sacrificial rice cover crop (*Oryza sativa*), which assists in soil stabilization and initial nutrient immobilization. This cover crop lasts only one growing season, does not renew itself, and gives way to other wetland vegetation.



**FIGURE 23.6** Inlet coarse rock zone at the Fish and Royer, Indiana, HSSF wetland.

### *Seeding*

Seeding is the next least expensive method of vegetation establishment. The techniques range from scattering in the wind (Figure 23.7), to back-pack broadcasting, to the use of seed drills. The seed is then pressed into the soils using a roller or cultipacker, or by light raking. The use of foot travel has also been used effectively, at the Isanti-Chisago site in Minnesota (Loer *et al.*, 1999). Volunteer labor has been used successfully at Brighton, Ontario, and Roblin, Manitoba, where grades 9 and 10 science students participated in seeding (PFRA, 2002). Hydroseeding has been used successfully (U.S. EPA, 2000a) but requires the addition of detergent to loosen the seed covering. In the Great Lakes region of the United States, the optimum seeding time is autumn through late spring.

Seeds for wetland plants may be harvested from local sources as part of the project work, particularly for *Typha*, which produces large seed heads with extremely numerous seeds, and which are easily picked in autumn. Seeds are also available from wetland nurseries in the United States, for over a hundred wetland species. The cost is considerable, ranging from $125 to $1,500 per kilogram of live seeds for common wetland plants in the northern United States (Table 23.5). The seeding rate is typically on the order of 2–4 kg of live seed per hectare. The purchased price of the seed is consequently in the range of $400–$3,000 per hectare, with a median of about $1,200 per hectare. The cost of installation includes the seeding itself (minor) and the fostering of germination and growth, via soil moisture/water management (major). Dry seeding has been estimated at ten man-hours per hectare, and hydroseeding at $145/ha (U.S. Army Corps of Engineers, 2000). Such required activities are usually inexpensive, and are less than the cost of the seed itself.



(a) (b)

**FIGURE 23.7** Seeding of the Brighton, Ontario, wetland (a) and the resulting growth (b). (Photos courtesy J. Pries.)

### *Planting*

The choice to plant originates with the desire to propagate a selected suite of plant species in a short period of time. Planting costs are a function of several different factors. These include the form of plant material (plugs or rootstock), the source of material (locally harvested, on-site nursery, or commercial nursery), the method of planting (manual or mechanical, volunteer or, contractor labor), and the overall planting density (number of plants per square meter). Because

**TABLE 23.5**
**Examples of Prices for Plants and Seeds (2007 USD)**

| | | Seed | | Bare Root | | Plugs | |
|---|---|---|---|---|---|---|---|
| | | ($/kg) | ($/ha) | ($/Plant) | ($/ha) | ($/Plant) | ($/ha) |
| **Individual Species** | | | | | | | |
| *Phragmites australis* | Common reed | — | — | 0.65 | 6,500 | 0.95 | 9,500 |
| *Phalaris arundinacea* | Reed canarygrass | — | — | 0.55 | 5,500 | — | — |
| *Typha latifolia* | Broad leaf cattail | 127 | 381 | 0.80 | 8,000 | 0.98 | 9,750 |
| *Typha angustifolia* | Narrow leaf cattail | 127 | 381 | 0.80 | 8,000 | 1.08 | 10,750 |
| *Scirpus validus* | Great bulrush | 485 | 1,454 | 0.65 | 6,500 | 0.80 | 8,000 |
| *Scirpus acutus* | Hard-stemmed bulrush | 717 | 2,152 | 0.85 | 8,500 | 0.88 | 8,750 |
| *Scirpus atrovirens* | Dark green rush | 329 | 986 | 0.85 | 8,500 | 0.88 | 8,750 |
| *Scirpus pungens* | Common three-square | 1,072 | 3,217 | 0.65 | 6,500 | 0.90 | 9,000 |
| *Scirpus cyperinus* | Woolgrass | 326 | 978 | 0.60 | 6,000 | 0.83 | 8,250 |
| *Iris virginica* | Blue-flag iris | 357 | 1,070 | 0.85 | 8,500 | 1.15 | 11,500 |
| *Leersia oryzoides* | Rice cutgrass | 419 | 1,258 | 0.40 | 4,000 | 0.80 | 8,000 |
| *Juncus effusus* | Common rush | 599 | 1,797 | 0.55 | 5,500 | 0.88 | 8,750 |
| *Sagittaria latifolia* | Common arrowhead | 374 | 1,121 | 0.55 | 5,500 | 1.25 | 12,500 |
| *Pontederia cordata* | Pickerel weed | 317 | 952 | 2.00 | 20,000 | 3.13 | 31,250 |
| *Sparganium eurycarpum* | Common bur reed | 431 | 1,293 | 0.55 | 5,500 | 1.28 | 12,750 |
| **Emergent Wetland Mix** (20 species plus cover crop) | | — | 3,088 | — | — | — | — |
| **Stormwater Wetland Mix** (18 species plus cover crop) | | — | 1,914 | — | — | — | — |
| **Rush/Bulrush Mix** (5 species) | | — | 556 | — | — | — | — |
| **Median** | | | **1,189** | **0.65** | **6,500** | **0.93** | **9,250** |

*Note:* The planting rate is assumed to be 10,000/ha, and the seeding rate is assumed to be 3.0 kg/ha. The cover crop is 25 kg/ha common oats (*Avena sativa*) and 8 kg/ha of annual rye (*Lolium multiflorum*).

*Source:* Data based on averages from J.F. New, Walkerton, Indiana (http://www.jfnew.com/) and Southern Tier Consulting, West Clarksville, New York (http://www.southerntierconsulting.com/).



(a) (b)

**FIGURE 23.8** Homemade planter used for the Hillsdale, Michigan, treatment wetland (a). The 1.4 ha of basins were quickly planted (b). See Figure 19.6 for a view of the wetland one year later.

of these variables, planting costs vary widely among wetland projects. In projects that employ hand planting, plant material costs may be a small fraction of the installed cost.

Commercial nurseries sell a wide range of wetland plants as potted "plugs" or bare root stock (Table 23.5). The median (2007) cost of bare root propagules was \$0.65 (n = 14), and for plugs, \$0.93 (n = 13). Transportation costs must be added to these, leading to a delivered cost of about \$1.20 per plant. It is sometimes possible to obtain plants from existing stands, such as cattails from roadside ditches. Extraction requires mechanical digging, typically with a backhoe, to a depth of about 30 cm, so as to gather the majority of the root mass. The plants are then separated from the mass by hand, with each propagule containing at least one healthy shoot and at least 20 cm of associated rhizome. The shoot is topped to a height of 30–40 cm, to reduce the amount of foliage that the transplant must maintain after the trauma of transplanting. On-site nurseries may also be established prior to wetland construction, for the purpose of supplying plant materials. However, the planting contractor may prefer to avoid the effort needed to extract and prepare the plants, and opt for the use of purchased plants, as was the case at Columbia, Missouri (Brunner and Kadlec, 1993). Established treatment wetlands may serve as sources of plant materials for new systems. For example, the plants for the Connell, Washington, system (Kadlec *et al.*, 1997) were extracted from the Arcata, California, project, and successfully established in the new wetland. Marsh establishment on the Texas coast required 11.3–29.3 man-hours per 1,000 plants to hand-dig, separate, and transplant various propagule types of 11 marsh species (Dodd and Webb [1975], as referenced by U.S. Army Corps of Engineers [2000]).

The total cost of plants depends on planting density, which ranges over about 0.25–4.0 plants per square meter (spacing of 0.5–2.0 m). If the plants are on 1-m centers, there are 10,000 per hectare. This value is used in Table 23.5 to establish the per-hectare cost of nursery-purchased plant plugs, which ranges from \$8,000 to \$31,000, with a median of \$9,250/ha.

The cost of putting the plants into the ground can be extremely variable. In some instances, innovation has been employed to quickly and efficiently plant a wetland with mechanical equipment (Figure 23.8). However, most small wetlands are planted by hand, with rates of up to 150 plants per person per hour for experienced planting crews. Allowing for lesser efficiency of 50 plants per person per hour, thus requiring 200 man-hours per hectare at 10,000 plants per hectare. This matches estimates provided by the U.S. ACE publication (U.S. Army Corps of Engineers, 2000). For labor at \$20/h, insertion costs would be about \$2,000/ha at 10,000 plants per hectare, or \$0.40 per plant. When combined with the per-plant purchase price, the cost would be \$1.60 per plant installed.

U.S. EPA (2000a) suggests a planting cost for FWS wetlands of \$0.83 per plant installed (adjusted to 2006 USD). This may be a reasonable estimate for locally harvested plants, but is probably too low for nursery-purchased plants.

Table 23.6 summarizes planting costs for HSSF wetland systems in the Minnesota–Wisconsin, U.S. regional market. All of these systems were hand-planted with rootstock (primarily *Scirpus atrovirens*, *Scirpus fluviatilis*, and *Scirpus cyperinus*) purchased from commercial wetland plant nurseries in Wisconsin. All 12 systems were planted at a density of 2.7 plants/per square meter. This represents an aggressive planting approach that uses well-established rootstock and is designed to achieve complete plant coverage in about two growing seasons (which is approximately 18 weeks per year in this climatic region). The median of \$4.58 per plant installed, in Table 23.6, is much higher than the estimate mentioned previously. The difference may be attributable not only to labor costs and other regional market factors but also to the unfamiliarity of the contractors with planting as a construction activity. On an areal basis, Table 23.6 shows a median of \$12.32/m$^2$. In another adjacent climatic region, for the Minoa, New York, HSSF wetland, planting costs were \$9.45/m$^2$.

For small to medium size wetlands (less than 10 ha), planting costs are less than 10% of the total capital cost. This

**TABLE 23.6**
**Planting Costs for Selected HSSF Wetlands[a]**

| System Name | Design Flow ($m^3/d$) | Wetland Area ($m^2$) | Plants ($/plant) | Installed Cost ($/$m^2$) |
|---|---|---|---|---|
| St. George, Minnesota | 25 | 595 | 4.55 | 12.24 |
| Darfur, Minnesota | 38 | 1,301 | 4.36 | 11.73 |
| Northern Tier High Adventure Base, Minnesota | 34 | 297 | 6.11 | 16.44 |
| Opole, Minnesota | 35 | 725 | 12.22 | 32.87 |
| Tamarack, Minnesota | 26 | 418 | 8.54 | 22.97 |
| Lakes of Fairhaven, Minnesota | 59 | 1,828 | 2.89 | 7.77 |
| Delft, Minnesota | 22 | 664 | 4.63 | 12.45 |
| Cedar Mills, Minnesota | 35 | 1,073 | 4.61 | 12.40 |
| St. Croix Chippewa, Wisconsin | 251 | 6,141 | 2.72 | 7.32 |
| Prinsburg, Minnesota | 206 | 4,094 | 4.71 | 12.67 |
| Mulberry Meadows, Minnesota | 72 | 1,580 | 3.91 | 10.52 |
| Cambridge-Isanti School District, Minnesota | 39 | 1,196 | 3.39 | 9.12 |
| **Median Value** | | | **4.58** | **12.32** |

[a] Costs in 2006 USD.

fraction applies to all regions, such as Spain (Nogueira *et al.*, 2006). For very large FWS wetlands, planting is too expensive, and natural recruitment is relied on.

## Structures

Hydraulic control of wetlands is generally provided through distribution piping and water level control structures. Inlet distribution, outlet collection, and water level control are required in all systems.

Many FWS systems operated with parallel paths, thus a splitter structure is needed to apportion the flows among flow paths. Depending on the size of the system and the complexity of the structure, costs will vary, but may be about $5,000–$10,000 for a 1-ha wetland (Wallace and Knight, 2006). In FWS wetlands, inlet distribution is normally accomplished using an inlet deep zone across the width of the wetland, also called an inlet spreader canal. The cost for this is part of the cut and fill for the overall project. These deep zones may be filled with coarse rock to discourage rodents, thus incurring the cost of the rock.

Sometimes a perforated pipe is placed at the bottom of the deep zone, to aid in uniform introduction of the water. The cost of 10-cm diameter perforated PVC is roughly $20/m. In warm climates, distribution piping may be above ground and above water (see Figure 18.23). Gated aluminum irrigation piping costs on the order of $30/m for 30 cm diameter, with lesser costs for smaller diameters. The 30-cm size can convey up to 15,000 $m^3/d$ without excessive headloss, and smaller diameters may be used for smaller flow rates.

Small HSSF wetlands may receive nearly raw wastewater, which will necessitate some form of inlet trash screening. In the United Kingdom, this may be a Copasac™ chamber with a filter bag; in other countries, a cleanable bar screen is used. If the water is delivered from a septic tank, or other settlement chamber, then this screening device may be unnecessary.

In small- to mid-sized HSSF wetlands, flow is generally distributed across the width of the bed in the inlet zone using perforated pipes or infiltration chambers. Perforated PVC pipe (100-mm diameter) is commonly used as a low-cost distribution method for small systems. The drawback of using perforated pipe is that the influent organic loading is concentrated at the perforations. Infiltration chambers (Figure 23.9) have a louvered sidewall which provides a greater number



**FIGURE 23.9** Placement of inlet distribution chamber at Lake Elmo, Minnesota. (From Wallace and Knight (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements*. Final Report, Project 01-CTS-5, Water Environment Research Foundation (WERF): Alexandria, Virginia. Reprinted with permission.)

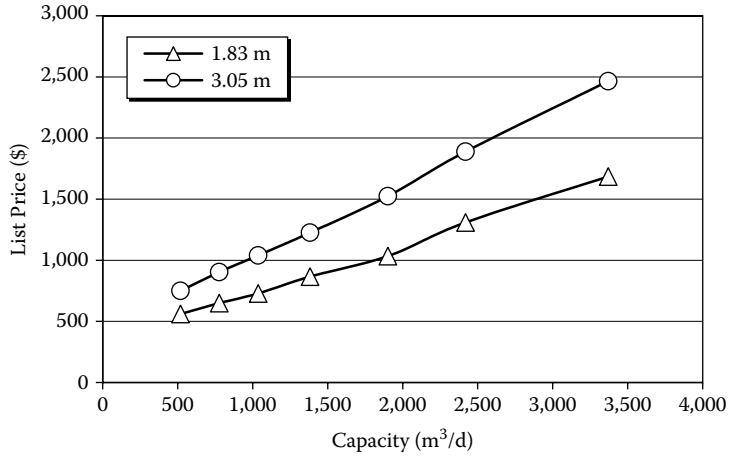



**FIGURE 23.10** Example pricing of simple in-line stoplog structures. The height of these contained units is variable; data are shown for 1.83 and 3.05 total structure height. The capacity is for a height over the weir of 25 cm. See Figure 18.26. (Data from Agri Drain Corporation, Adair, Iowa. Date: March 30, 2007, CCI = 7,856. http://www.agridrain.com/watercontrolproductsinline.asp.)

of openings and a larger cross-sectional area, both of which are desirable for evenly distributing the influent organic load. Generally, the same device (perforated pipe or infiltration chamber) is used at the outlet end of the bed for effluent collection. Installed costs for perforated pipe and infiltration chamber approaches in the northern United States are about $20/m and $40/m, respectively (Wallace and Knight, 2006).

The water level within a wetland cell is regulated by a downstream water level control structure. A typical outlet structure for small FWS wetlands is shown in Figure 18.27, and an outlet for a moderately large system in Figure 18.28. The costs for typical low-flow level control structures are given in Figure 23.10. A simple flexible pipe outlet level control structure for a HSSF wetland is shown in Figure 23.11. Some small systems use premanufactured "pond valves" that utilize stacking plates (with a 100 mm inlet and outlet pipe) to set the water level. Costs for this approach are below $1,000 for flows under 500 $m^3$/d (Wallace and Knight, 2006). Larger projects may use precast or cast-in-place concrete structures with a weir device (stop log, fixed plate, or adjustable weir gate). The ability to adjust the water level is an important control element, and providing operational control over the water depth in the wetland is well worth the incremental cost over fixed-level structures.



**FIGURE 23.11** Outlet level control structure at Pribraz, Czech Republic. Flexible pipes may be raised or lowered to adjust water level. There are two cells controlled by this device.

Large inlet and outlet structures for FWS systems can be fairly complicated, with many cost components (see Figure 18.22). These may involve remote sensing of water levels, sent to an operations center by telemetry, and remote control of opening and closing. Therefore, there is a need for electrical power, and the associated control equipment. The structure requires appropriate bedding, which usually means imported gravel. The cost breakdown for an example is given in Table 23.7. Despite the apparent complexity and cost, such structures are more economical when measured in terms of cost per unit flow rate. The structure of Table 23.7 cost approximately $0.42 per $m^3$/d, whereas the AgriDrain™ units cost about $0.71 per $m^3$/d.

**TABLE 23.7**
**Capital Cost of One Inlet Structure for STA2 Cell 4**

| Item | Description | Cost ($) |
|---|---|---|
| Piping | 40 m of 1.68-m diameter @ $983/m | 39,323 |
| Structure | Slide gate | 29,595 |
| Excavate, bed, install | 917 $m^3$ gravel | 9,095 |
| Sensors | — | 21,244 |
| Telemetry | — | 9,560 |
| Stilling wells | — | 26,644 |
| Electrical control | — | 36,115 |
| **Total** | | **171,577** |

*Note:* Adjusted to 2006 USD, CCI = 7,751. There are six such structures for the 816-ha wetland.

*Source:* Data from Brown and Caldwell (2005) *Basis of design report, Stormwater Treatment Area 2 Cell 4 Expansion Project*. Report to South Florida Water Management District. http://www.sfwmd.gov, West Palm Beach, Florida.

### Site Work

These site work costs generally include construction elements such as construction of access roads to the project site, fencing (if required), installation of electrical power to the project site, erosion control, seeding (outside wetland areas), and surface restoration. As the nature and extent of these site work requirements change from project to project, these costs will also vary. Fencing is a common requirement, and three-meter cyclone fencing costs approximately $50/m installed (Marsh *et al.*, 1998). Rodent exclusion fencing cost an additional $75/m installed at the 2-ha Hillsdale Michigan site (Kadlec *et al.*, 2007). Site work costs for 12 HSSF wetlands in the northern United States, ranging from 400 to 6,000 $m^2$, had a median of $46.08/$m^2$ (2006 USD), which was 8.1% of the total project cost (Wallace and Knight, 2006).

### Ancillary Benefits

As discussed in Chapter 19, ancillary benefits are sometimes part of the design philosophy of a treatment wetland, particularly a FWS system. Boardwalks, viewing platforms, hiking trails, information kiosks, visitor centers, and parking lots are typical added features for those treatment wetlands, and can add considerable expense. However, challenges to those expenditures are rare or nonexistent, because of the great public interest and acceptance of visitor features.

## Indirect Costs

### Engineering and Permitting

Engineering activities include conceptual design, final sizing, preparation of plans and specifications, preparation of the O&M manual, and permitting. These costs depend upon the size, complexity, and novelty of the project. In the United States, single-home HSSF wetland designs for domestic wastewater treatment wetlands can be produced for a few hundred dollars—in those locations where prescriptive design criteria have already been approved by local regulators. The time required to produce approved plans is a matter of weeks.

However, FWS wetlands, targeting other wastewater sources and differing cleanup goals, present a much different situation. Designs become performance based, and require significant expertise in conceptual design and sizing. There are likely to be no well-established regulatory procedures, which means that obtaining the necessary permits can become a challenging and time-consuming process. Because of scale, the number and complexity of drawings increases. A larger project is likely to go through several design phases and produce a conceptual design, a basis of the design report, and 10%, 30%, 90%, and final design reports—each subject to scrutiny and possible revision. The project may be subjected to value engineering, a process by which outside experts examine the project plans to ascertain if further economies might be realized.

Design costs for larger projects are usually expressed as a percentage of the overall construction costs. The cost of the routine engineering design has often been estimated at approximately 10–15% of the overall construction cost (U.S. EPA, 2000a). This amount is probably insufficient to deal with a long, drawn-out permitting process.

### Nonconstruction Contractor Costs

In addition to the construction elements of site work for a wetland system, the project will incur other project costs. Contractor expense costs are incurred by the contractor, but are not directly connected to the wetland construction. These typically include mobilization, bonding, insurance, traffic control (if required), and construction surveying and staking. These costs are dependent on factors such as the distance to the project site the contractor has to mobilize, the contractor's previous history with insurance carriers and bonding companies, bonding requirements in the construction contract, whether the construction work will impact existing traffic patterns, and the extent of surveying required to set reference stakes for site grading and pipe installation. Contractor overhead, or burden, is the cost of doing business that is not attached to any particular project, and it is commonly associated with the direct labor charges as a percentage. Because these items change from project to project, some variation in nonconstruction costs can be expected. Nonconstruction contractor costs for 11 HSSF wetlands in the northern United States ranged from 0.8% to 6.8% of the overall construction cost, with a median value of 4.0% (Wallace and Knight, 2006).

### Construction Observation and Start-Up Services

Services provided during construction and start-up typically include construction observation, inspections, testing, start-up assistance, and operator training. These components constitute approximately 10% of the overall construction cost (U.S. EPA, 2000a). Construction supervision needs to involve the services of a knowledgeable treatment wetland specialist, as a complement to the usual supervision of the implementation of civil works. Start-up also requires such services for the period of vegetation establishment. Operator training should include a good foundation of the general principles of treatment wetlands, as well as the specific characteristics of the project.

### Contingency and Escalation

Escalation is an allowance for inflation. Contingency is a percentage of the base cost to cover errors in human judgment, and it is intended to be spent. The amount of money set aside or planned for unforeseen events is a function of the level of the economic estimate being formulated. A project's risk elements may include accidents, vandalism, theft, work quantity and productivity variances, unfavorable weather, and bankruptcies. Contingency allotments of 10–30% are typically used.

### Illustrations

It is instructive to combine the various elements of capital cost into a summary by category. There are too many variables to do this in a general way, and consequently a few illustrations are presented here.

Table 23.8 shows a capital cost list for a hypothetical small-scale FWS treatment wetland. The 1-ha system might receive a hydraulic loading of 300 $m^3$/d (3 cm/d). The direct cost is $320,000/ha. It is seen that over 50% of the direct cost is for earthwork and liner, and another 28% is associated with planting soil and plants. The cost of land has been included, but is a small fraction of the total for this small wetland. The indirect costs are 50% of the direct costs.

Table 23.9 is the listing of estimated direct capital costs for STA2 Cell 4 of the Florida Everglades protection project, an 816-ha system construction-completed in 2006, and Table 23.10 lists the indirect costs (Brown and Caldwell, 2005). The direct cost is $8,513,000, or $10,433/ha. Most of the cost is associated with earthwork (31%), site preparation (28%), and structures (28%). There are no liner costs, and no planting soil or plant costs. Land costs have not been included, because the property was acquired through a convoluted set of purchases, sales, and exchanges. An approximate estimate is $12,500/ha, which would increase the capital unit cost to $23,000/ha. Regardless of the cost distribution, this very large system is much less expensive on a per-hectare basis than the 1-ha system, emphasizing that economy of scale is highly important. In fact, there is so much change with scale that any "universal" value for dollars per hectare, mean, or median is meaningless. The indirect costs for this project, or markups, total $6,144,000, i.e., 74% of direct costs excluding land, or 33% based on land plus construction.

**TABLE 23.8**
**Estimated Capital Costs for a Hypothetical 1-ha FWS Wetland System**

| Component | Unit | Quantity | Unit Cost ($) | Total Cost ($) |
|---|---|---|---|---|
| Land acquisition | ha | 1 | 10,000 | 10,000 |
| Site evaluation | lump sum | 1 | 2,000 | 2,000 |
| Clear and grub | ha | 1 | 8,000 | 8,000 |
| Earthwork | $m^3$ | 10,000 | 7 | 70,000 |
| Liner | $m^2$ | 12,000 | 8 | 96,000 |
| Planting soil | $m^3$ | 3,000 | 10 | 30,000 |
| Plants and planting | plant | 20,000 | 3 | 60,000 |
| Structures | lump sum | 5 | 2,000 | 10,000 |
| Conveyance | m | 400 | 35 | 14,000 |
| Site work | lump sum | 1 | 20,000 | 20,000 |
| *Total Direct Cost* | | | | *320,000* |
| Engineering | 15% | | | 48,000 |
| Construction observation | 5% | | | 16,000 |
| Start-up services | 5% | | | 16,000 |
| Nonconstruction costs | 5% | | | 16,000 |
| Contingency | 20% | | | 64,000 |
| *Total Indirect Cost* | | | | *160,000* |
| **Total Cost** | | | | **480,000** |

**TABLE 23.9**
**Estimated Direct Capital Costs for 816-ha STA2 Cell 4 in Thousands of Dollars[a]**

| Category and Item | Cost ($) | Subtotal ($) |
|---|---|---|
| **Canals** | | **970** |
| Inflow | 154 | |
| Collection | 146 | |
| Discharge | 292 | |
| Perimeter | 378 | |
| **Levees** | | **1,662** |
| Northeast | 151 | |
| North | 357 | |
| South | 276 | |
| South perimeter | 138 | |
| East | 713 | |
| Temporary | 27 | |
| **Site Preparation** | | **2,343** |
| Blast | 1,254 | |
| Clear and grub | 214 | |
| Degrade roads | 305 | |
| Fill ditches | 201 | |
| Remove buildings | 173 | |
| Remove periphyton test | 60 | |
| Soil placement | 50 | |
| Rip rap | 46 | |
| Seeding berms | 40 | |
| **Structures** | | **2,315** |
| Inlet | 311 | |
| Inflow | 517 | |
| Discharge | 587 | |
| Sensors | 209 | |
| Stilling wells | 157 | |
| Electrical control | 213 | |
| Power feed | 167 | |
| Roads | 153 | |
| **Miscellaneous** | | **1,223** |
| Mobilization | 854 | |
| Vehicles | 99 | |
| Survey | 37 | |
| Dewatering | 197 | |
| Temporary facilities | 37 | |
| **Total** | | **8,513** |

*Note:* Land costs are not included. There is no liner and no planting. Flow is by gravity.

[a] Costs in 2006 USD.

Capital costs for HSSF wetland systems are generally estimated using the same "line item" cost approach discussed for FWS wetlands. Construction quantities are calculated from the design, and the cost for each of these items is estimated using local or regional cost information. An example of this approach, using excerpts from an actual bid tabulation, is shown in Table 23.11. This example does not include land costs, but the contractors indirect charges are included, and are dispersed throughout the unit costs. Not surprisingly, most of the construction cost is for the media (36%) and liner (24%).

**TABLE 23.10**
**Estimated Markups for STA2 Cell 4 of 816 ha in Thousands of Dollars[a]**

| Cost | Percent | Amount ($) | Subtotal ($) |
|---|---|---|---|
| **Net Direct Costs** | | | **8,144** |
| Labor | 23 | 1,839 | |
| Materials | 10 | 824 | |
| Equipment | 25 | 2,036 | |
| Subcontractor | 2 | 146 | |
| Other | 40 | 3,299 | |
| **Net Markups** | | | **1,478** |
| Small tools | 5 | 92 | |
| General contractor labor burden | 34 | 625 | |
| Trade labor burden | 28 | 515 | |
| Sales tax on material | 6 | 49 | |
| Fuel and maintenance | 6 | 122 | |
| Material escalation | 9 | 74 | |
| **Net Financial** | | | **4,666** |
| Contingency | 30 | 2,886 | |
| Reserve | 5 | 481 | |
| Overhead and profit | 10 | 962 | |
| Bonds | 4 | 337 | |
| **Total** | | | **14,288** |

[a] Costs in 2006 USD.

*Note:* Engineering costs are not included.

## Economy of Scale

A strong determinant in the overall cost of a wetland treatment system is the size of the system. Larger projects generally benefit from economies of scale, resulting in lower unit costs. The size of the project may be measured as the wetland area, or as the flow rate treated, or as the population served if the water is domestic wastewater. Cost functions relate capital expenditure to these measures, and commonly take the form of a power law, with an exponent of 0.6–0.7 often being appropriate. Cost data in the wetland literature is often ambiguous about what has been included or excluded. Land costs are often not included, and the same is true of indirect costs. For small HSSF domestic systems, costs of the collection system and pretreatment devices (septic tanks) may be included. Therefore, the reader is advised that the following compilations of reported costs are subject to considerable variability, and will generally be lower than the summation of the various categories enumerated previously.

Figures 23.12 through 23.15 present the trends of costs with size or flow capacity for FWS and HSSF systems in the United States. These are reasonably well represented by the following relationships:

Areal:

$$\text{FWS:}\quad C = 194A^{0.690}\quad R^2 = 0.79\quad 0.03 < A < 10{,}000 \tag{23.1}$$

$$\text{HSSF}\quad C = 652A^{0.704}\quad R^2 = 0.75\quad 0.005 < A < 20 \tag{23.2}$$

Flow:

$$\text{FWS:}\quad C = 518Q^{0.729}\quad R^2 = 0.79\quad 1 < Q < 5{,}000{,}000 \tag{23.3}$$

$$\text{HSSF:}\quad C = 561Q^{0.498}\quad R^2 = 0.76\quad 0.5 < Q < 10{,}000 \tag{23.4}$$

where

$A$ = wetland area, ha
$C$ = cost, 2006 dollars in thousands
$Q$ = flow rate, $m^3/d$ in thousands

**TABLE 23.11**
**Selected Bid Results for a 300-$m^2$ HSSF Wetland on the U.S.–Canadian Border[a]**

| | | | Engineer | | Contractor 1 | |
|---|---|---|---|---|---|---|
| **Item** | **Quantity** | **Unit** | **Unit Cost ($)[a]** | **Total ($)** | **Unit Cost ($)[a]** | **Total ($)** |
| Pea gravel | 135 | $m^3$ | 56.27 | 7,596.73 | 56.27 | 7,596.73 |
| Drain rock | 23 | $m^3$ | 72.35 | 1,664.12 | 61.10 | 1,405.27 |
| Peat | 45 | $m^3$ | 64.31 | 2,894.07 | 77.18 | 3,473.10 |
| Plants | 750 | each | 6.11 | 4,582.41 | 6.11 | 4,582.41 |
| Earthwork cut/fill | 228 | $m^3$ | 8.04 | 1,833.26 | 13.66 | 3,114.87 |
| 0.76-mm PVC liner | 446 | $m^2$ | 7.22 | 3,220.96 | 14.46 | 6,447.37 |
| Granular bedding | 23 | $m^3$ | 40.19 | 924.39 | 40.19 | 924.39 |
| Protective fabric | 372 | $m^2$ | 3.29 | 1,222.81 | 2.63 | 977.34 |
| 100-mm Sch 40 PVC (insulated) | 27 | m | 40.09 | 1,082.51 | 38.09 | 1,028.40 |
| 50-mm Sch 80 PVC (insulated) | 3 | m | 32.08 | 96.23 | 38.09 | 114.27 |
| 50-mm Sch 40 PVC (insulated) | 12 | m | 24.06 | 288.73 | 38.09 | 457.07 |
| 100-mm Sch 40 perforated PVC pipe | 34 | m | 48.11 | 1,635.71 | 26.06 | 886.20 |
| Water control structure | 1 | each | 672.09 | 672.09 | 916.48 | 916.48 |
| Water balance test | 1 | each | 9,775.82 | 9,775.82 | 2,443.95 | 2,443.95 |
| **Constructed Wetland Subtotal** | | | | **37,489.84** | | **34,367.86** |

[a] Costs in 2006 USD.

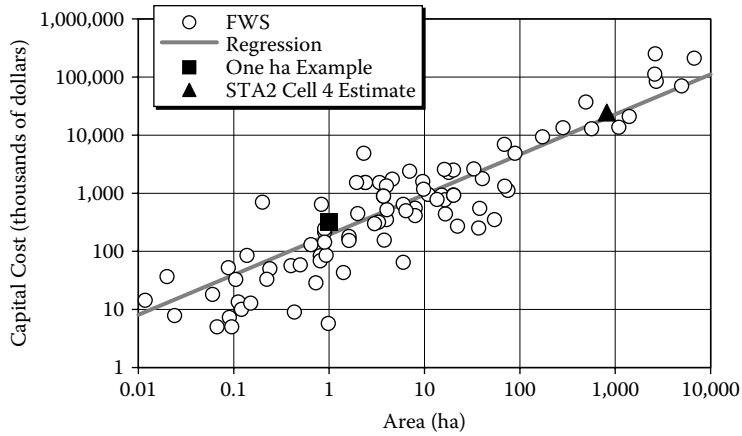



**FIGURE 23.12** The capital cost for FWS wetlands as a function of size (N = 84 wetlands). Points for $A < 300$ m$^2$ were excluded from the regression. $C = 194\,A^{0.690}$; $R^2 = 0.79$; $C$ in 1,000s of dollars.

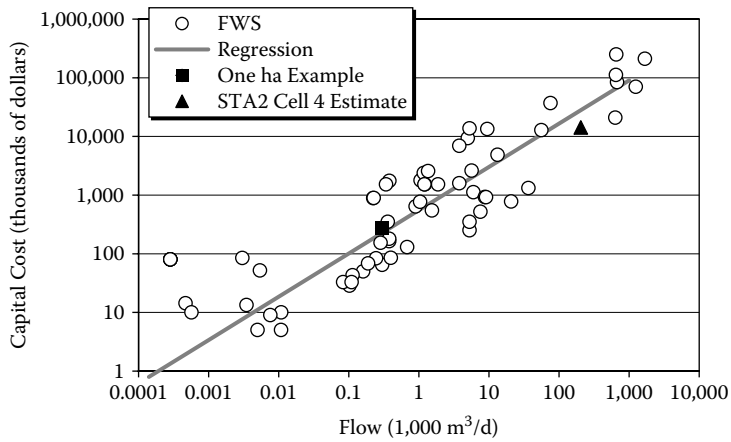



**FIGURE 23.13** The capital cost for FWS wetlands as a function of flow rate (N = 61 wetlands). Points for $Q < 1$ m$^3$/d are not considered in the regression. $C = 518Q^{0.729}$; $R^2 = 0.79$; $Q$ and $C$ in 1,000s of dollars.

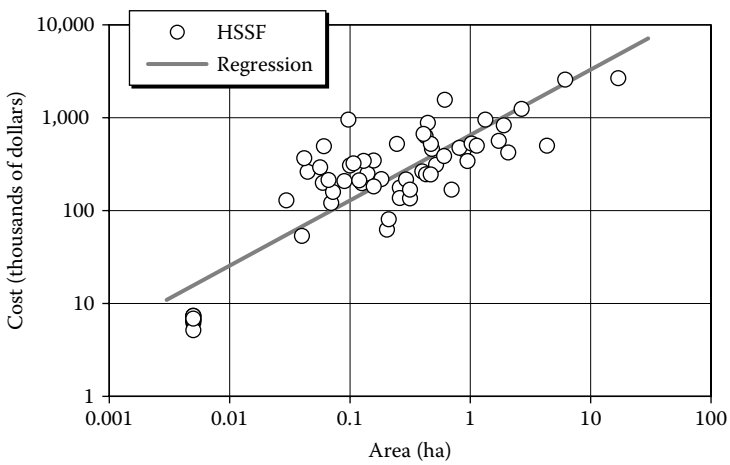



**FIGURE 23.14** The capital cost for HSSF wetlands as a function of size (N = 63 wetlands). $C = 652\,A^{0.704}$; $R^2 = 0.75$; $C$ in 1,000s of dollars.

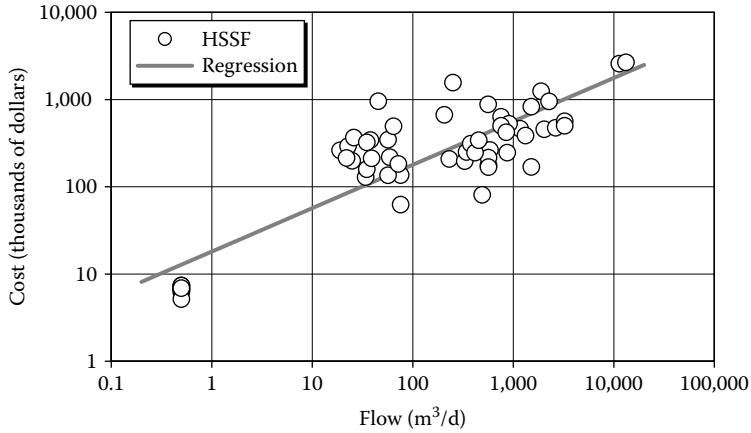



**FIGURE 23.15** The capital cost for HSSF wetlands as a function of flow rate (N = 58 wetlands). $C = 18Q^{0.498}$; $R^2 = 0.76$; $C$ in 1,000s of dollars.

The result of the nearly identical exponents for area is a nearly fixed ratio of costs for FWS and HSSF systems over the range of areas where both are used (about 0.1–10 ha). That ratio is 3.32 ± 0.14, with HSSF being the more expensive. The literature contains several estimates of the median cost per hectare for FWS and HSSF wetlands. For the data reviewed here, these medians are $98,000/ha for FWS and $1,230,000/ha for HSSF systems. These unit costs are not usable in a comparison, because they cover entirely different size ranges. As detailed in Chapter 16, treatment per unit area is about the same for FWS and HSSF systems, and as seen here HSSF costs are about triple those for the equivalent FWS system.

A direct comparison of costs on a flow basis is inappropriate, because FWS and HSSF systems are generally designed for different pollutants and different removals.

These cost functions may be used to obtain an approximate cost for a prospective project, but should not be used in lieu of detailed cost estimation.

In many cases, for domestic wastewater treatment, the measure of HSSF wetland size is the population served, partly because only design flows are known and partly because the cost to the individuals served is the desired result. Some results reported in the literature are:

$$\text{HSSF (Central America):} \quad C = 360\ PE^{0.755} \quad R^2 = 0.97 \qquad \text{Platzer } et\ al. \text{ (2002)} \qquad (23.5)$$

$$\text{HSSF (Spain):} \quad C = 490\ PE^{0.707} \quad R^2 = 0.71 \qquad \text{Nogueira } et\ al. \text{ (2006)} \qquad (23.6)$$

$$\text{HSSF (Portugal):} \quad C = 2{,}200\ PE^{0.679} \quad \text{estimated} \qquad \text{Junca de Morais } et\ al. \text{ (2003)} \qquad (23.7)$$

where
PE = Population equivalent
$C$ = Cost, 2006 USD

The high regional variability is apparent in these relations.

No matter which measure of scale is used, it is clear that an exponent of something like 0.7 is appropriate for scaling of these continuous flow systems.

Stormwater systems present a different situation, because the wetland size is often determined from the watershed size, and not any specific performance specification. Scaling to the size of the watershed may then be more appropriate (Wossink and Hunt, 2003). For the state of North Carolina, the following cost function has been derived by Wossink and Hunt (2003):

$$\text{Stormwater wetlands} \quad C = 6{,}910\ WS^{0.484} \quad 1 < WS < 100 \qquad (23.8)$$

where
WS = Watershed area, ha
$C$ = Cost, 2006 USD

This amounts to a set of formulas when converted to wetland area, because the guidelines for sizing in North Carolina specify wetland to watershed area ratios (WWARs) of between 1.0 and 6.5%.

## 23.2 OPERATION AND MAINTENANCE COSTS

Wetland systems have very low intrinsic O&M costs, including pumping energy, compliance monitoring, maintenance of access roads and berms, and mechanical component repair. These basic costs are much lower than those for competing concrete and steel technologies, by a factor of 2–10. Attempts at harvesting, or at maintenance of a particular vegetation species composition, can prove costly. Ancillary research costs—which can occur at the directive of the regulatory agencies—have sometimes cancelled out this large potential advantage, especially for natural treatment wetlands. Most maintenance tasks associated with operating a wetland facility deal with servicing pump headworks, and other conventional components of the treatment plant (U.S. EPA, 2000a).

### Free Water Surface Wetlands

The O&M costs for a FWS facility include pumping energy, compliance monitoring, dike maintenance, and equipment replacement and repairs. Dike maintenance consists of mowing and preservation of structural integrity. Equipment replacement and repairs pertain to piping and pipe supports, structures, and pumps. Pumping energy may be accurately quantified, as can the initial level of compliance monitoring, once a permit is issued. Mowing is primarily a matter of aesthetics, with secondary emphasis on visual detection of snakes and alligators. If public use is encouraged, there may be a need to maintain signage, trails, and boardwalks. Nuisance control or removal may be required, most often targeting mosquitoes, burrowing rodents, and bottom-stirring fish. Nuisance control, such as mosquito and rodent eradication, has sometimes proved to be troublesome.

The sum total of these activities is usually relatively inexpensive. No chemical purchases are involved, and there is not a need for highly trained personnel, nor significant time requirements for the necessary semiskilled employees. Annual costs range from $5,000 to $50,000 per year for small systems. However, ancillary research can greatly increase these expenditures. The estimate for the Incline Village system, made at the time of final conceptual design, was $163,000 per year (Table 23.12).

Permit-related sampling and reporting, combined with maintenance of upstream and downstream treatment processes, constitute most of the routine O&M associated with FWS wetlands. At Arcata, California, it has been estimated that O&M tasks directly associated with the wetlands require about $1600/ha·yr when adjusted to 2006 USD (U.S. EPA, 2000a). These types of routine checks do not include muskrat or nutria removal, or mosquito control, which will increase operating costs. Costs for these types of animal or vector control activities are highly site specific in nature, and are generally best estimated by assessing the amount of operator labor involved plus the cost of any control agents or equipment.

**TABLE 23.12**
**Estimated O&M Costs for the 135-ha Incline Village FWS Wetland System**

| Item | Annual Cost ($) |
|---|---|
| Personnel | 95,315 |
| Energy | 4,766 |
| Monitoring | 40,032 |
| Maintenance materials | 22,876 |
| **Total** | **162,988** |

*Note:* This information was developed after the conceptual design was finalized. It does not include research costs, nor profits derived from hunter use charges. Costs are in 2006 USD. O&M = operation and maintenance.

*Source:* Data from CWC (1983) *Draft Design Memorandum, Incline Village General Improvement District Wetlands*. Culp, Wesner, and Culp (CWC): 3461 Robin Lake, Cameron Park, California.

The limited amount of FWS O&M data from the literature is presented in Figure 23.16, and shows a median cost of about $2,000/ha.

### Subsurface Flow Wetlands

One primary operating cost of a single-home HSSF system is the cost associated with pumping the septic tank. Other costs are mainly determined by local permit requirements, which vary widely across the United States. In western Ohio, the annual maintenance and monitoring costs are approximately $250/yr when adjusted to 2006 costs (Steer *et al.*, 2003). Many systems are permitted at the state or provincial level, and are required to have licensed operators submit monthly discharge monitoring reports. An example of the cost breakdown for an HSSF system is shown in Table 23.13. Most of the O&M is associated with permit-related sampling and correspondence, followed by management of pumps, septic tanks, control

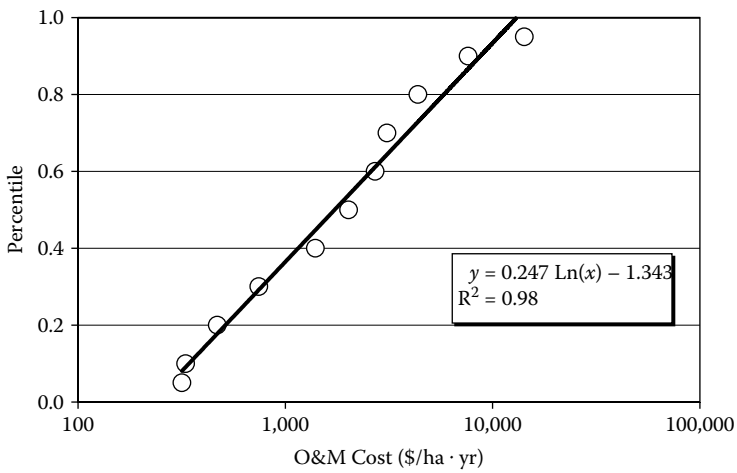



**FIGURE 23.16** Operation and maintenance (O&M) costs for FWS wetlands (N = 21).

**TABLE 23.13**
**Example Annual Operational Costs for a 46-Home Residential Development in Minnesota**

| Monitoring | Cost ($) |
|---|---|
| Sample collection | 4,320 |
| Lab coordination | 364 |
| Analytical | 1,405 |
| Regulatory correspondence | 573 |
| Permit renewal | 2,166 |
| *Subtotal* | *8,828* |
| **Operating** | |
| Visual monitoring | 1,353 |
| Electric | 3,596 |
| Telephone | 521 |
| Septic tank pumping | 1,249 |
| *Subtotal* | *6,719* |
| **Maintenance** | |
| Equipment replacement | 2,320 |
| Equipment maintenance | 521 |
| Emergency repairs | 521 |
| Sewer maintenance | 1,041 |
| Inlet zone bed maintenance | 3,698 |
| *Subtotal* | *8,101* |
| **Overhead** | |
| Billing | 1,437 |
| **Total** | **25,085** |

*Note:* Costs are in 2006 USD. Note that the inlet zone of the wetland bed line item accounts for nearly 50% of the maintenance cost.

panels, and other conventional treatment components. For small-scale systems, it is not meaningful to project per-hectare O&M costs because small HSSF systems have essentially the same monitoring and reporting requirements as large systems. For larger systems, it is possible to separate wetland O&M from total O&M required, including the rest of the facility. The annual O&M expenses for the HSSF wetland at Carville, Louisiana (568 $m^3$/d, 0.26 ha), were estimated at $780/yr or $3,000/ha.yr (U.S. EPA, 2000a) (2006 USD). This is approximately 6% of the O&M budget for the entire treatment system.

Inlet zone maintenance has the potential to be a large component of HSSF maintenance costs. Inlet zone maintenance will depend on the remedy method selected (replacement, washing, or chemical cleaning; none of these procedures have been used widely to fix clogged HSSF wetlands). Also, the time interval between bed maintenance is a factor. To date, no rational methods to predict bed clogging have been developed, although it is apparent that it is a widespread problem in full-scale HSSF wetlands (Cooper *et al.*, 2006a).

Here, we will consider the potential economic consequences of inlet zone maintenance. In the example of Table 23.13, it was assumed that inlet zone maintenance would be 15% of the construction cost, and inlet zone maintenance would occur every five years. These assumptions are based on the very limited data presented in Table 22.6. Based on these assumptions, and Equation 23.2, the cost for inlet zone maintenance every five years is predicted at $20,438 (2006 USD); at a 5% interest rate, the annual cost is $3,698. The assumption parameters (time to clog, percentage of construction costs) can, however, vary. It is apparent at this stage of technology development that inlet zone clogging is likely to be the largest maintenance cost component associated with HSSF wetlands. When evaluating the cost-effectiveness of any future projects, inlet zone maintenance costs must be considered. The continued viability of HSSF wetlands as an economically efficient technology is dependent on finding a cost-effective solution to the clogging problem.

## 23.3 PRESENT WORTH ANALYSES

The capital costs for a wetland project are "up front" needs, typically expended in the first year of the project. In contrast, the O&M costs are recurring, with a need for expenditure in every year of operation. Thus, when evaluating alternatives or planning for financing, it is necessary to combine these two types of costs to obtain a true picture of the overall project cost. Two basic choices are available. The first is to convert the future time stream of O&M costs to an amount of money that one would need to put in the bank to pay for the future O&M, and add it to the initial capital investment. This procedure produces the present worth of the project (Humphreys, 1991). The second choice is to convert the initial capital investment into a time series of payments that may be added to the time series of O&M costs, to create an annualized project cost. In essence, the capital cost is borrowed, and repaid over time. It is this latter approach that is often used for financing wastewater treatment projects. In fact, in the United States, a homeowner may well be paying the two bills separately each month: one payment to pay off the debt on building the system, and a second to pay for operating costs. Unfortunately, only a very few literature sources deal with the combination of both types of project costs for treatment wetlands.

Treatment wetlands are distinctively different from mechanical plants in a few very important ways. It is therefore necessary to examine the basic elements of the accumulation of costs, and for simplicity we here consider only the present worth analysis. However, it is a relatively simple matter to weave these concepts into an annualized cost analysis. Over the life of the project, in a simplified sense, it is expected that the following costs will be incurred:

- Capital costs, both direct and indirect, usually in the first year of the project life.
- Annual O&M costs. These include monitoring and other routine activities.
- Periodic, nonannual costs. These might include sediment cleanout, for instance.
- End of project costs and values, sometimes termed "salvage values." These may include not only demolition and disposal costs, but also the resale value of project assets.

The preceding sections have detailed the costs in the first two categories (capital and O&M).

## Present Worth Concepts

The total cost of a project at the time of inception is the total of capital costs, engineering services, and the present worth of O&M costs over the project life. This approach to economic estimating is required when the alternatives under consideration vary greatly in their life expectancy, and in their O&M costs. This is the case for wetlands. The overall project evaluation requires consideration of both capital and O&M costs, and the present worth technique is the appropriate vehicle for combining the two. The present worth of O&M costs, including equipment repairs and replacements, is the money that needs to be set aside now, at the prevailing interest rate, to pay for these future costs.

It is not possible to anticipate all the project-specific details of the economics of treatment wetlands. However, there are certain common features of many projects that are considered here.

### Discount Rate

The time value of money reflects the interest on borrowed sums and the inflation rate. The discount rate used in present worth calculations is the excess of the interest rate over inflation. In treatment wetland evaluations, discount rates of 4–8% are commonly used.

### Life Expectancy

Traditionally, the life expectancy of a "conventional" treatment alternative is 20 years. A treatment wetland has a longer life expectancy than concrete and steel equipment. Although there are no examples of engineered systems with long periods of operation, there are long-lived FWS wetland systems that have retained their effectiveness for up to 80 years, based on ex post facto monitoring. Both the Brillion Marsh, Wisconsin (Spangler *et al.*, 1976b), and Great Meadows Marsh, Massachusetts (Yonika *et al.*, 1979), operated for over 70 years, and in later years were shown to have retained treatment efficiency. As fully functional ecosystems, treatment wetlands may be expected to sustain their character for as long as appropriate hydrology is maintained. It is early in the history of constructed wetland facilities, so there is not a long track record on life expectancies for wetland components. However, in general terms, pumps and piping may last on the order of 40 years, and repair frequencies are known. Berms, dikes, and levees have lives in excess of 50 years. Therefore, it is plausible that treatment wetlands should be ascribed a basic life of 40–50 years, with mechanical components being replaced at greater frequency.

### Present Worth of Future One-Time Costs

The amount of money needed for a future expenditure reflects the discount rate in a simple, compound interest sense:

$$P = S\frac{1}{(1+i)^m} \tag{23.9}$$

where

$i$ = discount rate, fraction per year
$m$ = years before need
$P$ = present value, $
$S$ = future amount, $

Thus a $1,000 pump replacement ten years from now will require $614 today if the discount rate is 5%.

### Present Worth of Future O&M

Many O&M costs occur as a continuing stream. In analysis, it is usual to aggregate these into a yearly amount, and then to compute the sum of all such future payments. This may be done sequentially using Equation 23.9, or the sum can be directly evaluated from

$$P = Y\frac{(1+i)^N - 1}{i(1+i)^N} = Y \cdot F_{\text{pw}} \tag{23.10}$$

where

$i$ = discount rate, fraction per year
$F_{\text{pw}}$ = present value factor, dimensionless
$N$ = life expectancy, yr
$P$ = present value, $
$Y$ = yearly amount of O&M, $

Thus, the median FWS O&M of $2,000/ha·yr (Figure 23.16), over a 50-year lifetime, will require $36,512 today if the discount rate is 5%. Note that payment number 50 requires only $174 of today's money.

### Salvage Values and Decommissioning Costs

It is common practice to claim no salvage value at the end of project life in a feasibility study for mechanical plants; but this does not make sense in the context of a wetland project. Typically, the entire acquisition price is charged to the project up front, and there is no "salvage" value at the end of 20 years. In contrast to the crumbling concrete and rusted steel left after the mechanical process reaches the end of its useful life, the land associated with the wetland project will probably have greater (or equal) value than that at the time of acquisition. This principal component of the wetland project will probably have appreciated in value. It may be more accurate to delete land cost from the comparison for that reason. Investments in land are not subject to normal wear-out and replacement charges in a long-term view of the project economics. Therefore, the "salvage" value of the wetland property, which may have appreciated over the project life, is typically much higher than that of a mechanical plant, which may have depreciated. The present worth of the land after the nominal project life should be included as a credit in economic evaluation if comparisons are being made. These end-of-life costs and values are brought to present-day dollars by use of the present worth factor for the last year of the project.

## Total Present Worth

The various costs, consisting of capital plus the present worth of O&M, various replacement costs, and salvage values, are combined to yield the total present worth of the project. Thus, the example project in Table 23.8 (1 ha), with no replacements or equipment salvage, would yield at 5% discount rate and a 50-year life:

| | |
|---|---|
| Capital cost | $480,000 |
| Present worth of O&M | $36,512 |
| Present worth of final land value | $10,000 |
| Total present worth | $506,512 |

## An Illustration

An alternative comparison is illustrated in Table 23.14, evaluating wetland treatment and chemical treatment to remove phosphorus from agricultural runoff water in southern Florida. The dollar values in this example are large, because it is based on a system for the treatment of a very large flow (≈760,000 $m^3$/d). The estimates in this table were developed from information available at the time of final conceptual design, and are subject to change during final design and the accompanying modifications. The example is included here to illustrate the unique features of evaluating wetland alternatives.

The chemical treatment alternative is 17% cheaper than the treatment wetland on the basis of the capital expenditures needed to build the project. On the surface, this makes chemical treatment the more attractive alternative. This up-front comparison presumes a life span short enough that equipment does not need to be replaced, nominally 20 years. However, if the life span of the project is taken to be 50 years—which is characteristic of constructed wetlands, and has been demonstrated in the region—the analysis changes. It becomes necessary to consider the salvage value of worn out components, and their replacement costs. In Table 23.14, it is assumed that worn out equipment has zero value: it is unsalable, and there is no charge for disposal. On the other hand, land acquired for the project is assumed to maintain its value; no replacement purchases are necessary. It is therefore logical to exclude land costs from the analysis, because it can be sold at the conclusion of the project with no loss in value.

When these factors are taken into consideration, the treatment wetlands are 13% cheaper than chemical treatment. The conclusion of the capital cost analysis is reversed.

Next, the O&M costs are totaled and converted to their present worth. Chemical treatment, as the name implies, requires more energy, materials, labor, and supplies than wetland treatment. Monitoring costs would be the same. In this example, and in virtually all similar cases, O&M costs are higher for the equipment-oriented technology. The annual O&M for chemical treatment is twice as expensive. The present worth of O&M is a significant fraction of the project cost. Consequently, the total present worth of the wetlands project is only 70% of the total present worth of the chemical treatment alternative.

**TABLE 23.14**
**Estimated Cost Comparison for Phosphorus Reduction in Agricultural Runoff (in Thousands of Dollars)**

| Wetland Alternative | | Chemical Treatment Alternative | |
|---|---|---|---|
| **Total Capital Costs Excluding Replacement** | **193,028** | **Total Capital Costs Excluding Replacement** | **160,331** |
| Land | 51,228 | Land | 3,184 |
| Procurement premium | 15,368 | Procurement premium | 558 |
| Pump station capital cost | 21,256 | Replaceable equipment | |
| | | Mixing through thickening | 76,454 |
| Equipment replacement (present worth, 8% discount rate) | | Equipment replacement (present worth, 8% discount rate) | |
| Pump station replacement | | Pumps piping electrical (25% @ 25 years) | 1,015 |
| (25% pump station capital replaced @ 25 years) | 777 | Mixing through thickening (100% @ 20 and 40 years) | 2,898 |
| **Land-Free Capital Cost** | **142,577** | **Land-Free Capital Cost** | **161,060** |
| **Operating and Maintenance Costs** | | **Operating and Maintenance Costs** | |
| Labor | 881 | Labor | 1,577 |
| Materials | 184 | Materials | 372 |
| Chemicals | 0 | Chemicals | 833 |
| Energy | 339 | Energy | 339 |
| Monitoring | 223 | Monitoring | 223 |
| Total annual O&M | 1,628 | Total annual O&M | 3,704 |
| **Present Worth of O&M (50-year life span, 8%)** | **49,754** | **Present Worth of O&M (50-year life span, 8%)** | **113,294** |
| **Total Present Worth of Capital + O&M** | **192,330** | **Total Present Worth of Capital + O&M** | **276,175** |

*Source:* Base information from Brown and Caldwell (1993) *Phase II Evaluation of Alternative Treatment Technologies*. Report to the South Florida Water Management District, 18 February 1993.

The assumed factors in this example will not prevail in all circumstances, but it does serve to indicate that extra care should be taken in economic analysis of a wetland alternative.

### Annualized Cost

This procedure has all the same elements as present worth, but the total project costs are allocated to a uniform series of annual payments over the life of the project. In simplest terms, a loan for the initial capital investment is repaid over the life of the wetland, and added to the annual O&M charges. This total annualized cost becomes the basis for comparison. For example, the capital costs for the project in Table 23.13 were $513,515 (2006 USD). At a 5% discount rate and 20-year life, the annual repayment would be $41,206. The O&M is $21,386 per year, and thus the total annualized cost is $62,592.

Another situation in which an annualized cost approach may be warranted is in the calculation of sewer bills. In the United States, homes are typically billed for sewer service on a quarterly or monthly basis. Funds are accumulated and used to pay off debt associated with capital costs, routine maintenance, emergency repairs, and component replacement. An example of this type of rate calculation for a 53-home treatment system is shown in Table 23.15; the associated cash flow is shown in Figure 23.17.

Annualization has also been applied to urban stormwater wetlands in North Carolina (Wossink and Hunt, 2003). For watersheds of 6–20 ha, annual costs were found to be $22,000–$85,000/ha, depending upon the (irrecoverable) value ascribed to land. Those land values were taken to be $125,000/ha for undeveloped residential land, and $540,000/ha for land in commercial development zones. Wetland watershed area ratios of 1.0–6.5% were applied, on a regionally variable basis.

## Cost per Pollutant Load Removed

It is sometimes appropriate to measure the cost of a project, not in terms of the population served or the hectares required, but per-unit mass of pollutant removed. This may be done by dividing the cost by the removal. For instance, phosphorus removal in the Florida Everglades wetland projects has been assessed in terms of the present worth of dollars required divided by the lifetime total mass of phosphorus removed in kilograms. For a discount rate of 8% and a lifetime of 20 years, the present worth for removing an estimated 157,000 kg of phosphorus per year for 20 years was $81,000,000, or about $26/kg.

The concept of nutrient credit trading, coupled with the land needed for treatment wetlands, has spurred consideration of the use of off-site wetlands to remove nitrogen and phosphorus rather than on-plant-site mechanical equipment (Hey *et al.*, 2005). The loads that need to be removed at large wastewater treatment plants are easily identified, as can the costs of such nutrient removal at the plant. Therefore, the wetland alternative needs to be evaluated on the basis of cost per kilogram of nitrogen and phosphorus. FWS treatment wetlands are more economical than traditional removal (Hey *et al.*, 2005).

### Economics of Storage

Cold climate conditions may engender the need for winter storage of wastewater. Overland flow systems require storage whenever the mean daily temperature is below freezing (Crites *et al.*, 2006), but FWS constructed wetlands have a wider operating window. For example, storages for northern wetlands are 180 days at Minot, North Dakota; Houghton Lake, Michigan; and Estevan, Saskatchewan. However, storage may be as much as 320 days in the arctic (Dillon Consulting, Ltd., 2006).

The use of winter storage avoids the "bottleneck" winter period, when some wetland processes are operating at their slowest pace. As discussed in Chapter 17, this period may be avoided if a storage pond is available, or is constructed. There are several competing factors that must be considered in the evaluation of alternatives. First, if the wetland is to operate in winter, the area needed will be larger than that needed in warm conditions. The area factor is on the order of three, meaning that the winter size will be about triple the summer size to accomplish the same degree of treatment for the same flow. This corresponds to a temperature coefficient of about $\theta = 1.07$, and a temperature difference of 16°C (mean summer 18°C, mean winter 2°C). Second, the flow is not the same if storage is added; the entire year's flow must be disposed of during the warm season. For instance, if six months' storage is employed, then the warm season flow will be double the annual average. Therefore, the wetland size is reduced according to the warmer temperatures, but increased to account for the increased flow. Nevertheless, the size of the wetland will decrease if storage is added. Third, the storage pond, however, adds to the overall project footprint, thus increasing land and liner costs. In general, there is only a small difference in total area for storage periods of 90 days, but a significant increase (≈70%) in total project area for 180 days of storage, compared to year-round operation with a large wetland to deal with the winter bottleneck. Fourth, the amount of earthmoving increases, nearly in direct proportion to the volume of water being treated (for level sites).

### An Illustration

As an illustration, consider a system to be constructed on a level site, with a (square) storage pond and a square wetland outline, which may have internal compartments and parallel flow paths (Figure 23.18). The further assumptions are:

- The design hydraulic loading during the warm season is 5 cm/d. If storage is used, this loading rate prevails on average throughout the discharge (warm) season, and determines the required area.
- The design hydraulic loading during the cold season is 1.25 cm/d. If the wetland is to be used year-round, this determines the required area.

## TABLE 23.15

## Example of a Rate Structure Calculation Using the *AccuRate* Financial Planning Software (in Dollars)

**AccuRate Financial Planning Software**

| | |
|---|---|
| Including capital replacement | |
| Total revenue | 1,133,704 |
| Total expense | 1,108,704 |
| Net | 25,000.00 |

| **Year** | **2007** | **2008** | **2009** | **2010** | **2011** | **2012** | **2013** | **2014** | **2015** | **2016** | **2017** | **2018** | **2019** | **2020** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Revenue | | | | | | | | | | | | | | |
| Monthly rate per home | 42.91 | 43.76 | 44.64 | 45.53 | 46.44 | 47.37 | 48.32 | 49.29 | 50.27 | 51.28 | 52.30 | 53.35 | 54.42 | 55.50 |
| Occupied homes | 12 | 25 | 50 | 53 | 53 | 53 | 53 | 53 | 53 | 53 | 53 | 53 | 53 | 53 |
| SAC fee | 1,200 | 1,200 | 1,200 | 1,200 | 1,200 | 1,200 | 1,200 | 1,200 | 1,200 | 1,200 | 1,200 | 1,200 | 1,200 | 1,200 |
| SAC revenue | 14,400 | 15,600 | 30,000 | 3,600 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Subtotal Revenue** | **20,578** | **28,729** | **56,784** | **32,559** | **29,538** | **30,129** | **30,731** | **31,346** | **31,973** | **32,612** | **33,264** | **33,930** | **34,608** | **35,300** |
| Expenses | | | | | | | | | | | | | | |
| Monitoring (service provider cost) | 14,300 | 14,586 | 14,878 | 15,175 | 15,479 | 15,788 | 16,104 | 16,426 | 16,755 | 17,090 | 17,432 | 17,780 | 18,136 | 18,499 |
| Operating costs | 7,020 | 7,160 | 7,304 | 7,450 | 7,599 | 7,751 | 7,906 | 8,064 | 8,225 | 8,390 | 8,557 | 8,728 | 8,903 | 9,081 |
| Maintenance costs | 3,450 | 3,519 | 3,589 | 3,661 | 3,734 | 3,809 | 3,885 | 3,963 | 4,042 | 4,123 | 4,206 | 4,290 | 4,375 | 4,463 |
| Capital replacement | 0 | 801 | 520 | 1,455 | 3,240 | 1,414 | 5,992 | 1,565 | 580 | 4,466 | 0 | 2,284 | 0 | 7,730 |
| **Subtotal Expenses** | **24,770** | **26,066** | **26,291** | **27,741** | **30,052** | **28,762** | **33,887** | **30,018** | **29,602** | **34,069** | **30,194** | **33,083** | **31,414** | **39,773** |
| Net | (4,192) | 2,663 | 30,493 | 4,818 | (514) | 1,367 | (3,156) | 1,328 | 2,371 | (1,457) | 3,070 | 847 | 3,194 | (4,472) |
| Interest | (83.83) | 53.26 | 609.86 | 96.35 | (10.28) | 27.34 | (63.12) | 26.56 | 47.41 | (29.13) | 61.40 | 16.94 | 63.88 | (89.45) |
| **Cash in Bank** | **–4,275** | **–1,559** | **29,544** | **34,458** | **33,934** | **35,328** | **32,109** | **33,464** | **35,882** | **34,396** | **37,527** | **38,391** | **41,649** | **37,087** |

| | | | | Replacement | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Annual Asset Replacement** | **Manufacturer Cost** | **Installation Labor** | **Incidental Costs** | **Cost (today)** | **Schedule (years)** | **2007** | **2008** | **2009** | **2010** | **2011** | **2012** | **2013** | **2014** |
| **Collection** | | | | | | | | | | | | | |
| Lift station control panel parts | 400 | 100 | 0 | 500 | 3 | 0 | 0 | 520 | 0 | 0 | 550 | 0 | 0 |
| Lift station pump replacement | 2,000 | 650 | 25 | 5,350 | 7 | 0 | 0 | 0 | 0 | 0 | 0 | 5,992 | 0 |
| Sewer repair | 500 | 10,000 | 5,000 | 15,500 | 25 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Treatment Constructed Wetland** | | | | | | | | | | | | | |
| Pump replacement | 1,500 | 200 | 25 | 0 | 7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Piping repair | 100 | 350 | 0 | 450 | 15 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Blower replacement | 750 | 350 | 75 | 588 | 4 | 0 | 0 | 0 | 623 | 0 | 0 | 0 | 670 |
| **Disinfection UV Disinfection** | | | | | | | | | | | | | |
| Bulb replacement | 600 | 160 | 25 | 785 | 2 | 0 | 801 | 0 | 832 | 0 | 864 | 0 | 895 |
| **Permitting** | | | | | | | | | | | | | |
| Permitting improvements | 2,500 | 500 | 0 | 3,000 | 5 | 0 | 0 | 0 | 0 | 3,240 | 0 | 0 | 0 |
| **Subtotal** | | | | | | **0** | **801** | **520** | **1,455** | **3,240** | **1,414** | **5,992** | **1,565** |

*Note:* Revenue is received from monthly sewer bills and in the early years of the project, sewer connection fees. Funds are expended to operate the system and replace worn-out components.

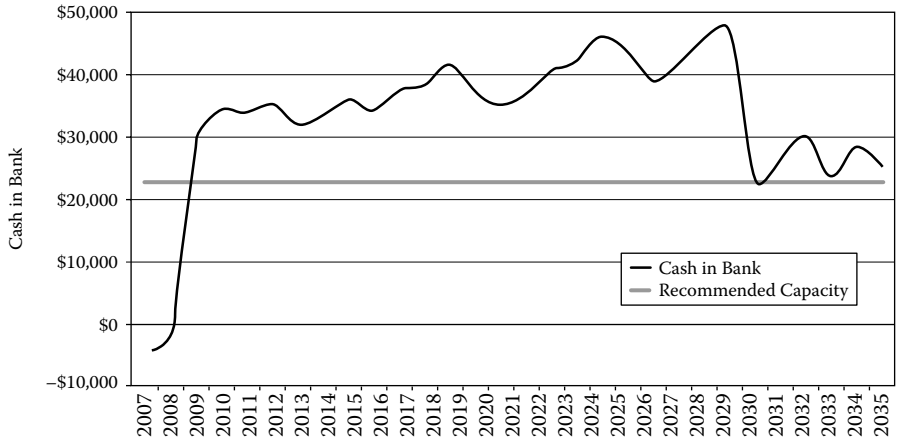



**FIGURE 23.17** Example of cash-flow projection for a 53-home treatment wetland system based on the user rate structure of Table 23.15, using AccuRate financial planning software.

- The soils on the site are acceptable for berms, and thus cut and fill may be balanced. This is presumed for pond and wetland individually.
- Setbacks around the project are 20% of the bermed areas.
- The berms have 3:1 side slopes, and a 3-m top width.
- The storage pond requires a 0.5-m freeboard and a 0.5-m minimum depth. The operating depth change is to be 2 m. Thus, the total depth of the basin is 3 m.
- The wetland requires a 1-m freeboard (to allow for solids accumulation and depth increases) and a 0.3-m average depth. Thus, the total depth of the basin is 1.3 m.
- Wetland internals, meaning inlet and outlet deep zones and cell divider berms, are not considered.

The results of cut-and-fill calculations are given in Table 23.16, for three flow rates (10, 100, and 1,000 $m^3/d$), and for 0, 90, and 180 days of storage. Even though this is a highly simplified illustration, several important issues can be identified. First, the increased summer flows partially balance the higher area requirements for winter operation. This results in a wetland size of one third to half the winter size, not a reduction to one fourth as the loading implies. The earth moved is in the range of 0.4–1.2 $m^3/m^2$ of the total bermed area. The total project area increases with the addition of storage, but only minimally for 90 days. In general, 90 days of storage means less earthmoving than for zero storage, and 180 days does not require much more. The nature of the cut and fill is such that the wetland water surface is near the original ground elevation (−0.29 to +0.09 cm). However, the maximum pond water elevation is above the original grade (0.43–2.09 m).

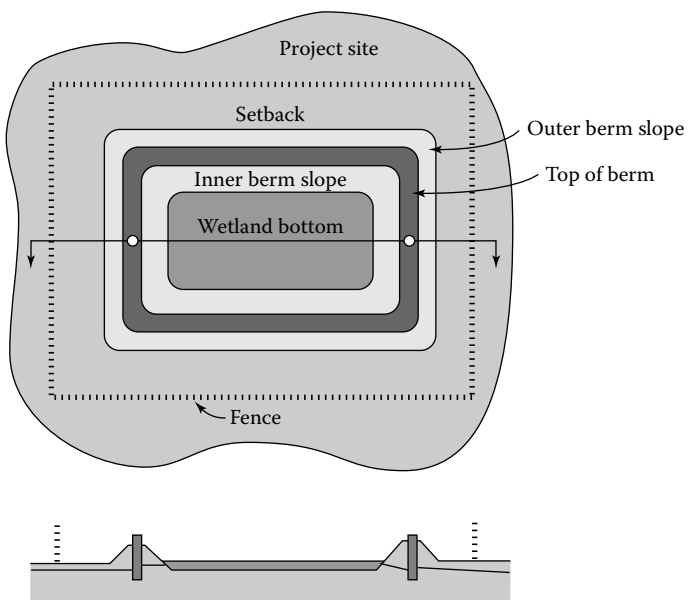



**FIGURE 23.18** General layout of a treatment wetland. For a small system, the overall project footprint is much larger than the wetted area.

**TABLE 23.16**
**Effects of Storage on Project Areas and Earthmoving for Hypothetical Site Conditions[a]**

| | Wetland Wetted Area ($m^2$) | Wetland Earth Moved ($m^3$) | Storage Volume ($m^3$) | Storage Top Area ($m^2$) | Storage Earth Moved ($m^3$) | Project Total Area ($m^2$) | Total Earth Moved ($m^3$) |
|---|---|---|---|---|---|---|---|
| **10 $m^3$/d** | | | | | | | |
| No storage | 800 | 1,207 | 0 | 0 | 0 | 2,332 | 1,207 |
| 90 days | 265 | 446 | 900 | 896 | 708 | 3,204 | 1,154 |
| 180 days | 395 | 634 | 1,800 | 1,505 | 1,280 | 4,681 | 1,914 |
| **100 $m^3$/d** | | | | | | | |
| No storage | 8,000 | 10,901 | 0 | 0 | 0 | 13,852 | 10,901 |
| 90 days | 4,705 | 3,745 | 9,000 | 5,775 | 4,666 | 14,459 | 8,411 |
| 180 days | 3,946 | 5,485 | 18,000 | 10,775 | 7,776 | 25,555 | 13,261 |
| **1,000 $m^3$/d** | | | | | | | |
| No storage | 80,000 | 105,557 | 0 | 0 | 0 | 109,428 | 105,557 |
| 90 days | 26,545 | 35,411 | 90,000 | 48,887 | 22,820 | 105,094 | 58,231 |
| 180 days | 39,459 | 52,394 | 180,000 | 95,469 | 34,852 | 187,883 | 87,246 |

[a] See text for conditions and assumptions.

This illustration is not intended to be used in design, because of the highly restrictive assumptions. However, modern computer-aided design tools are capable of easily dealing with irregular sites, more general basin geometries, and wetland internals.

Economics vary from location to location. The relative costs of land and earthmoving combine with the size and volume calculations to provide a basis for alternative selection. In the event that costs of storage or no storage are competitively close, the choice to avoid winter operation is likely to be more attractive. Additionally, controlled discharge strategies can modify the storage and wetland requirements as detailed in Chapter 17. The treatment potential of the combination is evaluated for the wetland (using the procedures in this book) and lagoon (from other sources) performance calculations.

## SUMMARY

Despite the inherent variability in costing a wetland treatment system, basic cost components of a wetland can be estimated using a line-item approach. Wetland treatment systems generally share a basic list of common components, which usually includes items such as land, earthwork, liner, rooting medium, wetland plants, and hydraulic control structures. The cost of these basic components can be estimated using pricing and availability information for a given regional economy. However, the line-item approach makes estimating economies of scale difficult, and has limited application in areas in which local pricing information is unavailable. Other components of wetland system design, such as site investigation, engineering design fees, and operation and maintenance (O&M) costs, should also be included when developing an overall cost estimate.

# 24 Modified and Combined Systems

Treatment wetlands can be designed to be "au natural" or complex. Examples of designs along a full spectrum from almost zero fossil-fuel input to highly engineered and managed systems can be found around the world. In this book we term these design and operation strategies as Type *A* and Type *B* wetlands. Type *A* systems are at the most natural, low-energy end of the scale. Type *B* wetlands are higher-energy and more highly engineered systems. There are good reasons to slide on this scale of design and management complexity, depending upon project goals. In this chapter we describe some of the rationale in selecting the appropriate level of management intensity.

Type *A* treatment wetlands clearly cost less to operate than Type *B* systems. Type *A* systems may also cost less to construct, even though they typically occupy a larger footprint area than Type *B* systems. They are the preferred solution when a number of factors are not limiting:

- Relatively inexpensive land area is abundant.
- Land is relatively level.
- There is insufficient treatment potential with a Type *A* approach.
- There is insignificant potential for unauthorized human interaction with the wastewater in the wetland.

Habitat and aesthetic benefits outweigh the potential for nuisance conditions related to mosquitoes or other wildlife.

Type *A* wetlands are typically constructed free water surface (FWS) marshes and relatively low-tech constant-flow horizontal subsurface flow (HSSF) wetlands.

The Type *B* treatment wetland-management strategy is preferred when one or more of the factors in the preceding list are limiting the design. Specific examples of Type *B* treatment wetlands include:

- HSSF wetlands augmented with reactive media intended for phosphorus or metals removal
- Addition of aerators for treatment improvement
- Operation of vertical flow (VF) wetlands in a fill-and-drain mode for improvement of nitrification and denitrification
- Combinations of VF, HSSF, or FWS wetlands to achieve specific process goals
- Recycling of treated effluent to improve denitrification and/or address influent toxicity issues
- Step-feeding of influent to reduce the impact of localized loading zones
- Wetlands and aquatic plant treatment systems with plant harvesting
- High-solids treatment wetlands with frequently cleaned forebays
- Operating wetlands in a load-and-rest operating regime to address bed clogging or improve treatment efficiency

There are situations where these wetland modifications make sense from a process design and economic standpoint, as discussed in this chapter. However, the system will invariably require more operator attention, and there is often a reduction in system reliability. For instance, if a treatment wetland requires an air blower to achieve regulatory compliance, system reliability is essentially governed by the mechanical blower, not the wetland ecosystem. The same is true for different operating regimes; if a treatment wetland requires alternating periods of loading and resting to avoid bed clogging, failure to follow the prescribed operating sequence will lead to system failure.

Constructed wetlands have limits of performance, which have been discussed in previous chapters. It is human nature to attempt to relieve some of these constraints by modifying the basic wetland construct or its operation. It is the purpose of this chapter to explore some of the innovative ideas that have been implemented to enhance wetland performance. These fall roughly into four categories:

- Ecological or environmental modifications
- Chemical additions
- Operational strategies
- Integrated sequences of natural systems, including one or more wetlands

These will typically move the wetland system from a passive mode (Type *A*) to a mode of active management (Type *B*). They will also usually involve more capital and operating cost, so the potential advantages should be weighed against other options for improving performance, such as alternative technologies or increased size.

## 24.1 ECOLOGICAL OR ENVIRONMENTAL MODIFICATIONS

Natural wetlands have been regarded as "self-designing" systems. They have also been ascribed certain functions that provide many kinds of benefits: habitat, flood control, biodiversity, and water quality improvement. Treatment wetlands target the water quality improvement functions. It is therefore logical—to humans—to try to nudge the wetland to a condition that fosters water quality. For instance, it has been

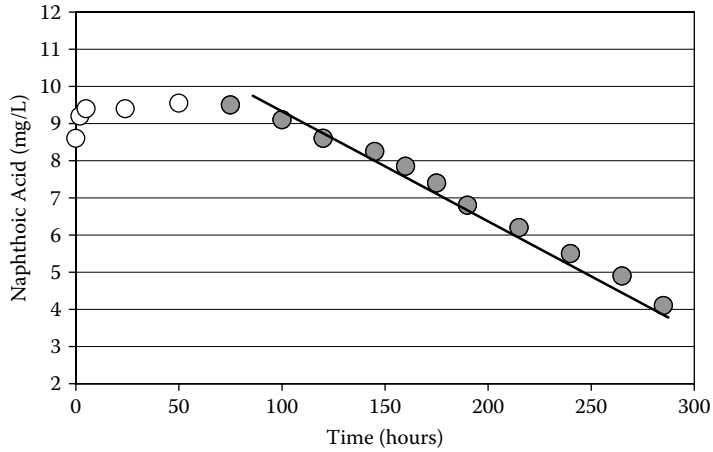



**FIGURE 24.1** Disappearance of naphthoic acid in cattail mesocosms. A zero-order fit is appropriate after an induction period. ($R^2$ = 0.99). (Data from Kadlec and Srinivasan (1995). *Wetland treatment of oil and gas well wastewaters*. Final Report DOE/MT/92010–10 (DE95000176), U.S. Department of Energy: Bartlesville, Oklahoma; graph from Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

suggested that magnetic fields can improve wetland performance (Rasit, 2006). Ecological and environmental modifications that have been implemented to enhance the treatment performance of wetlands are briefly discussed in the following text.

### Microbial Enhancement

Microbial processing plays a very important role in the reduction of many pollutants in treatment wetlands. In virtually every case, among the many thousands of treatment wetlands, the presence of microbial populations that will deal with the target pollutants is assumed in the design. For the major categories of contaminants, that presumption appears justified: if there is ammonia, nitrifiers are there; if there is nitrate, denitrifiers are there; if there is sulfate, sulfur bacteria are there. However, these populations cannot establish instantaneously, and a period of some weeks of grow-in may be necessary. In general, it does not seem necessary to inoculate a wetland with the desired microorganisms.

It has been found that specialty bacteria—those that focus on particular or unusual compounds—will also require a period of time to develop. This leads to the observation of an induction period, during which little or no pollutant reduction occurs. For instance, Kadlec and Srinivasan (1995) found no degradation of naphthoic acid for about 100 hours in batch cattail mesocosms, but thereafter a steady decline (Figure 24.1).

Runes *et al*. (2001) found that bioaugmentation of sediment with small quantities of an atrazine spill-site soil (1:100 w/w) resulted in the mineralization of 25–30% of atrazine under both unsaturated and water-saturated conditions. Atrazine and its daughter products were almost undetectable after 30 days. Unbioaugmented sediment supplemented with organic amendments (cellulose or cattail leaves) mineralized only 2–3% of atrazine. The population density of atrazine-degrading microorganisms in unbioaugmented sediment was increased from $10^2$ to $10^4$/g by bioaugmentation, and increased to $6 \times 10^5$/g) after incubation. A high population of atrazine degraders (approximately $10^6$ $g^{-1}$) and enhanced rates of atrazine mineralization also developed in bioaugmented sediment after incubation in flooded mesocosms planted with cattails (*Typha latifolia*) and with atrazine at 3.2 mg/L. Runes *et al*. (2001) concluded that bioaugmentation could be a way to enhance constructed wetlands. However, the persistence of degraders was not determined, and could be a constraint.

### Willow Wetlands with Zero Discharge

The growth habit of willows allows their use to provide zero-discharge, zero-percolation treatment for small households. There are approximately 1,400 willow systems in use in Denmark, and these have been described in Chapter 3. They rely upon an excess of evapotranspiration (*ET*) over precipitation, which would not provide acceptable water disposal for *regional* values of *ET*. However, willow wetlands can be configured to enhance *ET* losses by using linear layouts in a crosswind aspect (see Figure 3.27). Willow systems are small, serving 30 PE or less, and are essentially "wind rows" that multiply regional *ET* by a factor of 2.5 due to the clothesline effect described in Chapter 4. The gravel beds that support the willows serve as a reservoir to accumulate wastewater during the wet seasons and periods. During dry periods, the interstitial water is drawn down. Therefore, sizing is determined by the water balance, which in turn is specific to a particular climatic zone (Gregersen and Brix, 2001; Brix and Gregersen, 2002; Brix and Arias, 2005). Approximately 120–300 $m^2$ are required for a single household under Danish climatic conditions.

Raw sewage is pretreated in a sedimentation tank, and the supernatant is pumped onto or just under the surface of the plastic-lined bed. Treated effluent may be collected in a system of drainage pipes, and water may be recirculated back to the pumping well or the sedimentation tank. Zero-discharge

willow systems are constructed with a width of eight meters and a depth of 1.5 meters (Gregersen *et al*., 2003). A drainage pipe is placed at the bottom of the bed, which is used to purge salt water from the bed after salinity rises too much. One third or one half of the willows are harvested every year to keep them in a young and healthy state with high transpiration rates (Brix and Arias, 2005). Harvesting at such small scale can readily be accomplished by hand labor.

The willow systems will me*et al*l regulatory requirements because there is no outflow to any receiving water bodies. Willow systems may also be used in conjunction with soil infiltration. The willows will evaporate all wastewater during the growing season, but during winter there is partial discharge that is infiltrated into the soil.

## Engineered Plants

In previous chapters it has been shown that the particular species of plants used in treatment wetlands is not critical, provided they create and sustain the desired biogeochemical cycles. However, there are differences among plants for some pollutants, most notably for trace metals and trace chemicals. Thus, in addition to the possibility of selection of hyperaccumulators, there is the longer-term possibility of genetic engineering for the desired traits.

In recent years, much attention has been focused on the improvement of plants for the purpose of enhancing phytoremediation through genetic engineering. One approach is to influence the enzymatic pathways for assimilation and volatilization by overexpressing genes of rate-limiting enzymes in plants (Berken *et al*., 2002). Another approach is to introduce additional metabolic pathways for hyperaccumulators. In this way the capacity of plants to take up, accumulate, and volatilize compounds can be increased beyond that of any naturally occurring plant species.

An ideal plant for environmental cleanup can be envisioned as one with high biomass production, combined with superior capacity for pollutant tolerance, accumulation, and degradation. With genetic engineering, it may be feasible to manipulate a plant's capacity to tolerate, accumulate, and metabolize pollutants and thus create an improved plant for environmental cleanup. Pilon-Smits and Pilon (2002) found that mercury volatilization and tolerance was increased by genetic engineering approaches by two- to threefold more metal per plant. Nandakumar *et al*. (2005) found that *Typha latifolia* could be genetically transformed by *Agrobacterium tumefaciens* to enhance phytoremediation enzyme production.

Heaton *et al*. (1998) proposed the use of transgenic aquatic, salt marsh, and upland plants to remove available inorganic mercury and methylmercury from contaminated soils and sediments. They noted that plants engineered with a modified bacterial mercuric reductase gene were capable of converting Hg(II) taken up by roots to the much less toxic Hg(0), which is volatilized from the plant. Plants engineered to express the bacterial organomercurial lyase gene were capable of converting methylmercury taken up by plant roots into sulfhydryl-bound Hg(II).

Results from lab and greenhouse studies are beginning to show promise for transgenics, but field studies are needed to establish their treatment potential, competitiveness in the wetland environment, and risks associated with their use.

## Artificial Enclosures

One extreme example of environmental modification is to place the treatment wetland in an artificial environment. In fact, this is commonly done for bench- and pilot-scale treatment units because the convenience of working indoors (especially in cold climates) outweighs added operational costs such as artificial lighting (Figure 24.2).

In larger-scale applications, greenhouses have been used as artificial enclosures to maximize the use of available sunlight (Figure 24.3). Due to the high cost of the building envelope; greenhouse systems are designed as mechanical activated-sludge processes (U.S. EPA, 1997), although floating plant racks are used on top of the aeration tanks. Plants used in the system may contribute to more stable operation due to retention of biosolids, and lower waste-activated sludge production has been reported (Austin, 2001).



**FIGURE 24.2** Indoor bench-scale unit treating landfill leachate; White Bear Lake, Minnesota. Two 1,000-Watt lights are required to provide the necessary light intensity to support plant growth in these four mesocosms. Care must be taken to ensure that the lights do not cause the plants to dry out or burn. This setup was later modified by constructing a mini-greenhouse over the reactors using a PVC frame and a clear plastic sheet.

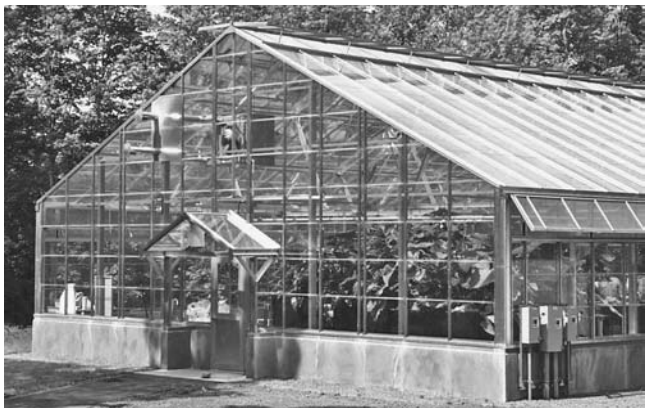


**FIGURE 24.3** Greenhouse-based treatment system near Land O' Lakes, Wisconsin. In the winter, considerable heating costs are incurred to minimize snow loads on the roof.

The energy input associated with the activated-sludge process is quite high compared to treatment wetlands (see Table 1.1), and if the system is located in a cold climate, heating demands can significantly increase energy costs (Brix, 1999).

Due to the high capital and operating costs associated with indoor systems, they have been used in applications where ancillary benefits such as aesthetics, visitor access, and education are at a premium.

## 24.2 CHEMICAL ADDITIONS

Wetlands do not necessarily supply sufficient reactants that may be desirable or required to foster a particular removal process. Those additional reactants may be supplied via the media for either FWS or SSF wetlands, or they may be supplied as additional chemical streams entering the wetland.

### Reactants via Media

The selection of media to support specific chemical reactions is a design issue. In the case of FWS units, it is common to ensure that adsorption capacity is present by the incorporation of organic material in the rooting medium. Peat, compost, or organic soils components also condition the soil and promote easier plant establishment and maintenance. In the case of HSSF systems, a much wider choice can be considered, but typically materials are in three categories:

- Sorbents for phosphorus
- Reactive media for metals, including alkalinity/sulfide generators to generate precipitation reactions
- Materials that are designed to decompose and generate a source of organic carbon for denitrification or sulfate reduction

Light expanded clay aggregates, zeolites, ferrous or calcitic substrates, or organic materials can all be considered. Their phosphorus binding potential has been discussed in Chapter 10. Metals typically will bind to organics, and therefore mine drainage applications incorporate some form of compost as a layer in their substrate. This zone of wetland bed also functions anaerobically to reduce sulfate to sulfide, which then precipitates metals (see Chapter 11). Mulch applied to the top of an SSF wetland bed can be used to supply organic carbon in excess of the plant biogeochemical cycle to fuel reactions such as denitrification if this is a process objective.

All of these techniques of bed solids used as a chemical addition have the implied need for periodic maintenance. Sorbents become saturated, and surface mulches become depleted in soluble materials unless the plant geochemical cycle can produce sufficient organic carbon after start-up. Therefore, these solid reactants must be replaced on some schedule, determined by the stoichiometry and loadings of the materials being treated. Because of the cost and difficulty of restructuring the bed, and reestablishing the wetland, the projected lifetime of a bed solid reactant typically needs to be at least five to ten years. Alternatively, reactant chemicals may be added as separate flows to the wetland.

### Reactants via Added Streams

Mechanical treatment plants often utilize chemical additions to sustain particular portions of their treatment processes. Aeration or oxygen addition fosters both biochemical oxygen demand (BOD) reduction and nitrification. In plants that include denitrification, some means of resupplying the carbon requirement is needed for conventional denitrification, such as utilization of an internal carbon-rich stream or the addition of an external carbon source (Metcalf and Eddy Inc., 1991). Removal of phosphorus in mechanical plants can involve the use of iron or aluminum salts to precipitate phosphorus. These process steps all have analogs in constructed wetland technology.

The processing of nitrogen in a wetland requires oxygen for nitrification and carbon for conventional denitrification (see Chapter 9). Supplies of these necessary reactants may not be sufficient to support the full requirements of required removals. Thus, HSSF wetlands are usually limited in their nitrification capabilities, and VF wetlands are usually limited in their denitrification capabilities. The capacity of a wetland to foster nitrogen reduction may be increased by inclusion of stream additions of carbon or oxygen, often at some considerable operational expense. The cost of such an enhancement should be weighed against other options, including the capital cost of increasing the size of the system if necessary to be closer to the Type *A* end of the spectrum.

### Aeration of FWS Wetlands

Although the entire water surface of a FWS wetland is exposed to air, reaeration is still subject to mass transfer resistances as discussed in Chapter 5. Oxygen supply may therefore be a constraint on BOD reduction and nitrification. It has been supposed that open-water areas may contribute to enhanced oxygen supply (U.S. EPA, 2000a), but

that presumption is in question based on operating data (Kadlec, 2003d; 2005e). Therefore, forced-air diffusers have been considered for augmenting FWS performance, although such attempts have been few. At West Jackson County, Mississippi, air injection was attempted in the deep-zone cross trenches. No conclusive data was obtained because nutria ate the plastic air diffusers.

### *Ouray, Colorado*

Situated at 2,365 meters above sea level, this municipal constructed wetland is surrounded by the San Juan Range, the youngest and steepest range of the Rocky Mountains (Campbell and Ogden, 1999; HDR/ERO, 2001). There is an average annual flow of about 1,500 $m^3$/d, higher in summer during the tourist season. The wetlands are arranged in two parallel paths of three cells each. Each path is 21 × 143 m, for a total system area of 6,100 $m^2$. Vegetation in all six cells is 80 to 90% cattail (*Typha* spp.), with the exception of Cell 3, which is 80% duckweed (*Lemna minor*) and 20% cattail (*Typha latifolia*). Other species present include common reed (*Phragmites australis*), curly dock (*Rumex crispus*), and lady's thumb (*Polygonum persicaria*). Because of concerns of sulfate reduction in the wetland inlet, the inlet deep zone of the wetlands is aerated with submerged perforated air distributors.

The water quality from the aerated lagoon pretreatment is approximately secondary. The wetlands reduced the concentrations of BOD to 4 mg/L, and total suspended solids (TSS) to 6 mg/L, over the period 1996–2000. These are clearly near background, and thus it is not possible to assign any advantage or disadvantage to aeration in these reductions. Ammonia in the wetland influent was not monitored, but the wetland effluent ammonia was 9 mg/L over the same four-year period. This is a high concentration, which exceeds that expected for an aerated lagoon alone, and therefore no large advantage could have been achieved in the wetland aeration for ammonia reduction.

## Carbon Additions to FWS Wetlands

Under many circumstances, denitrification in FWS wetlands is not carbon limited. The decomposing dead biomass is a moderately large carbon supply, although only a fraction of that material is useable (see Chapters 9 and 17). However, some waters contain large enough concentrations of nitrate that removal is constrained by the carbon supply. For instance, Stengel *et al*. (1987) worked with tap water at 30 mg/L nitrate nitrogen and augmented that to 110 mg/L to emulate groundwaters in northern Germany. It was concluded that, under some circumstances, insufficient carbon would be present to support full denitrification.

As noted in Chapter 9, a C:N ratio of 5 or 10 to 1 has been deemed to be required for denitrification. This ratio determines the potential need for an external carbon supply. For instance, for a HLR = 10 cm/d and a nitrate concentration of 50 mg/L, the $NO_x$-N load is 5 g/$m^2$·d. Decomposition of 1,000 g/$m^2$·yr (dry weight) of biomass would yield about 500 gC/$m^2$·yr, or 1.4 gC/$m^2$·d. Thus, the C:N ratio of 1.4:5 is far less than needed to support full denitrification.

### *Lamb–Weston, Connell, Washington*

The Lamb–Weston company built an integrated natural system for treating and reusing wastewater from a potato processing facility near Connell, Washington (Kadlec *et al*., 1997; Burgoon *et al*., 1999). Two 1-ha free water surface wetlands were constructed in series for denitrification of nitrified effluent from VF beds (intermittent sand filters) (see Figure 24.18). The wetlands were designed to remove $NO_3$-N from wastewater prior to land application. During a five-month period of intensive study, the flow was 4,950 $m^3$/d, producing a direct hydraulic loading of 25 cm/d. When coupled with an inlet nitrate nitrogen of 45 mg/L, the denitrification wetlands received a very high nitrate load of 110 kg/ha·d. The chemical oxygen demand (COD) in the nitrified water was insufficient to support denitrification, even with the carbon generated by the wetland plants. Therefore, a small flow of untreated potato wastewater (COD ≈ 3,000 mg/L) was directed to the wetlands as a supplemental carbon supply. This flow raised the hydraulic loading rate (HLR) to 28 cm/d. Addition of this primary effluent resulted in a COD:$NO_3$-N mass load ratio that ranged from 10 to 25. The $NO_3$-N mass removal rate ranged from 50 to 99% of the influent load. During the five months January to May 1998, average monthly effluent $NO_3$-N was 1.6 mg/L, and an average of 85% of the nitrate load was removed (Burgoon, 2001). Thus the feed-forward of untreated water provided the necessary carbon to allow nearly complete denitrification at high loading rates.

Sources of carbon are not always conveniently at hand. Some wetlands have utilized methanol or molasses, but these are expensive. Multiple wastewater sources can potentially provide synergism. For example, the remediation of a high-nitrate source water could be enhanced in a wetland that also received high BOD municipal or food processing wastewaters. At the time of this writing, such multisource systems have been proposed but not built.

## Aluminum and Iron Additions to FWS Wetlands

The phosphorus-binding capabilities of iron and aluminum, and their role in the wetland biogeochemistry of phosphorus, are very well known (Reddy and DeLaune, 2007). Very early in the history of FWS technology development, it was observed that naturally occurring, blending streams of water containing iron were very effective in phosphorus removal. At Waitangi, New Zealand, phosphorus deposition occurred predominantly by reaction between treatment wetland waters (high in phosphorus, alkaline pH) with natural wetland waters (high in Fe and Al, acid pH) immediately below the confluence of their flow streams (Cooper, 1992; 1994).

This study, coupled with the knowledge of mechanical plant phosphorus precipitation technology, led to investigation of the direct addition of these metal salts to FWS wetlands for the purpose of enhancing phosphorus removal (Bachand *et al*., 1999; Bachand and Richardson, 1999). Preliminary jar

test results suggested that treatment with ferric chloride and alum could effectively remove phosphorus to between 8 and 30 µg/L (Metcalf and Eddy Inc., 1997). It was hoped that chemical amendments directly within the treatment wetlands would create a synergistic effect in which phosphorus is more effectively and efficiently removed than by either of these two methods, chemical amendments and wetlands, alone. It was also hoped that addition of a polymeric coagulant might not be necessary because the wetland would effectively serve as a settling basin with very long HRT. However, small flocs formed, which did not quickly settle. Total phosphorus concentrations in the water column remained elevated. This approach was therefore abandoned as a means of enhancing wetland operational effectiveness.

### Aeration of SSF Wetlands

Here for the first time we encounter a significant influence of patents on SSF treatment wetland technology. Aeration of HSSF systems has developed both in the open literature and in the proprietary sector. Aerated subsurface flow wetlands are available as a patented technology in North America (Wallace, 1998; Dufay, 2000; Wallace, 2001b; Wallace, 2002a; Flowers, 2003; Wallace and Lambrecht, 2003). We cannot access all the proprietary data, and hence cannot independently evaluate it. For most patented systems, only anecdotal, qualitative information has been presented. However, we can summarize the results from the open literature and provide interpretation of it. Nothing here should be taken as supporting or refuting the claims in existing patents.

The growing realization in North America during the mid-1990s that HSSF wetlands were inherently oxygen-transfer-limited systems led to considerable interest in artificial aeration through the application of a mechanical blower and diffuser tubing as a means to boost treatment performance (especially for ammonia) in these systems. One of the most basic applications of the technology is shown schematically in Figure 24.4:

In the most simplistic sense, this can be thought of as "blowing bubbles" inside a saturated-flow HSSF wetland. However, implementation of the technology requires an understanding of the frictional loss of compressible fluids, hydrodynamics, and oxygen transfer theory. Nevertheless, a variety of successful application areas have been developed, including petroleum hydrocarbons (Wallace and Kadlec, 2005), ammonia (Wallace *et al*., 2006a), domestic wastewater (Wallace *et al*., 2006b), and airport de-icing runoff (Wallace *et al*., 2007a).

The decision to aerate a SSF wetland comes at a heavy operational cost, as operation and maintenance (O&M) of the facility is greatly increased. From an economic viewpoint, aeration is only justified when the lifecycle cost of aeration is sufficiently offset by the reduction in capital cost, as the net savings of reduced wetland size less the cost of aeration equipment.

Aerated wetlands typically occupy a much smaller footprint than a nonaerated wetland, which translates into lower construction cost. Or, sufficient land may not be available for a more passive Type *A* wetland, or not available at an economically feasible price.

At the time of this writing, performance information on aerated HSSF wetlands is a mixture of forensic analysis of full-scale systems, as well as analysis of performance data from pilot-scale systems or mesocosms.

In the first generation of aerated wetland design, assumptions from other treatment technologies, such as lagoons, were often "borrowed" to support blower sizing and diffuser-tube placement in HSSF wetland systems. As a result, these systems were not optimally designed. In recent years, there has been an increased focus on pilot-scale systems, especially for industrial effluents. A well-designed and implemented pilot study can produce data on rate coefficients (essential for first-order modeling), but questions about the hydrodynamic aspects of the pilot reactor versus the proposed full-scale system often remain (Noorvee *et al*., 2005b).

Apart from reaction rates, there is the issue of oxygen transfer rates. These depend upon bed depth, interstitial

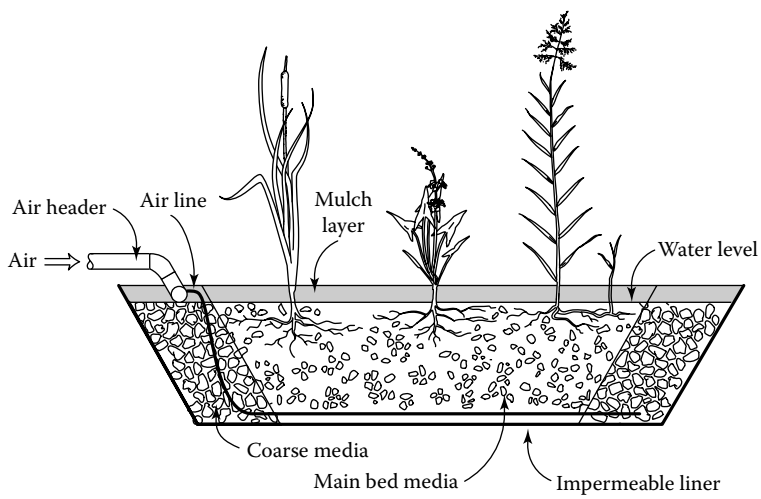



**FIGURE 24.4** Schematic of an aerated HSSF wetland.

oxygen concentrations, diffuser geometry, and other factors. However, relatively gentle aeration can deliver something on the order of 50 to 100 gO/m$^2$·d to the HSSF bed. This rate of oxygen delivery is significantly greater than rates that can be inferred from Type *A* nonaerated HSSF wetlands (Tables 9.26, 9.27, Chapter 9), which are usually in the range of 1 to 6 gO/m$^2$·d.

The energy requirement for such gentle aeration can nevertheless be an important factor in overall energy usage (see Table 1.1, Chapter 1), especially compared to passive Type *A* wetlands, but is generally much lower than conventional mechanical treatment processes.

### *Domestic Wastewater Treatment*

In a recent study, Wallace *et al*. (2006b) compared the treatment efficiency of 17 full-scale aerated HSSF wetlands and 22 full-scale nonaerated wetlands, using the loading chart and statistical methods presented in Wallace and Knight, 2006. The conclusion of this study was that aeration resulted in significant improvements in the reduction of both $BOD_5$ and TSS. With aeration, the equivalent wetland size needed for $BOD_5$ removal could be reduced by 67%; for TSS, the size reduction was approximately 36%.

### *Ammonia Oxidation*

Aeration of SSF wetlands for ammonia removal has been studied in recent pilot-scale systems. For an effluent containing a mixture of cyanide/iron/ammonia from a gold mine in South America, aeration was found to increase the ammonia oxidation rate over tenfold, from a volumetric rate (2 TIS) of 0.52 $d^{-1}$ to 5.7 $d^{-1}$ (Wallace *et al*., 2006a). A study of ammonia oxidation at different temperatures for domestic wastewater at a pilot-scale wetland in Canada (Wallace *et al*., 2006a) yielded a temperature correction factor (θ) of 1.02, indicating that temperature sensitivity is of some importance in aerated wetlands (compare with Table 9.17, Chapter 9). This would be consistent with a view that heavily loaded aerated HSSF wetlands are microbially dominated ecosystems.

### *De-icing Runoff*

Treatment of de-icing compounds (discussed in Chapter 13) has been studied under aerated and nonaerated conditions in HSSF mesocosms. For removals of $CBOD_5$ associated with de-icing runoff, the volumetric rate coefficient was measured at 0.6 $d^{-1}$ (2 TIS); aeration at 0.85 m$^3$ air per hour (per m$^3$ of wetland bed) improved the rate to 5.5 $d^{-1}$. Additional aeration at a rate of 2.0 m$^3$ air per hour (per m$^3$ of wetland bed) further increased the degradation rate coefficient to 15.9 $d^{-1}$. Based on the range of temperatures studied (22 to 4°C), the temperature correction factor was 1.03 (Wallace *et al*., 2007a).

### *Petroleum Hydrocarbons*

Aeration has been demonstrated to improve removals of petroleum hydrocarbons through aerobic degradation and volatilization. Early work was conducted at the Gulf Strachan Gas Plant in Alberta, where an aeration system initially designed for freeze resistance during winter operations resulted in improved removals of total petroleum hydrocarbons (TPH) and benzene, toluene, ethylbenzene, and xylene (BTEX) compounds (Moore *et al*., 2000a). A full-scale aerated wetland system was constructed by Williams Pipeline for a terminal facility in Watertown, South Dakota, and has been operating successfully (Wallace, 2002b).

Pilot-aerated SSF wetland studies were conducted by Ferro *et al*. (2002) for the purpose of reducing petroleum hydrocarbons in extracted groundwater. Both aerated and nonaerated degradation rates were measured, as summarized in Table 13.2, Chapter 13 (Wallace and Kadlec, 2005). These findings are consistent with the growing body of knowledge of biodegradation and volatilization rates for hydrocarbons under aerobic and anaerobic conditions (Suarez and Rifai, 1999).

### *Aquaculture Waste*

A study at the Botanical Garden of Montreal investigated the effects of aeration in mesocosms for the treatment of aquaculture waste (Ouellet-Plamondon *et al*., 2006). Aeration improved the removal of TSS, COD, and total Kjeldahl nitrogen (TKN). For instance, the first-order COD rate coefficient (plug flow) improved from 0.15 to 0.25 m/d for unplanted mesocosms, and from 0.19 to 0.24 m/d for *Typha*-planted mesocosms. Aeration was only used in the inlet region of the HSSF bed. The researchers concluded that aeration was a promising approach for improving the performance of HSSF wetlands.

## Oxygen Transfer

A recent study was conducted to measure the standard oxygen transfer efficiency of aerated SSF beds using off-gas measurements (Wallace *et al*., 2007b). Oxygen transfer is a function of the contact time between the bubble and the water column, which depends on the water depth. It also depends on constituents within the wastewater that affect oxygen mass transfer at the bubble/water interface (Metcalf and Eddy Inc., 1998). The average oxygen transfer efficiency in a HSSF wetland mesocosm using a standard sodium sulfite solution was 4.7% per meter of water depth. As a frame of reference, coarse bubble diffusers have a transfer efficiency of approximately 2.6% per meter, and fine bubble diffusers have efficiencies of approximately 6.6% per meter (U.S. EPA, 1989). Because most HSSF wetlands have a bed depth of 0.3 to 0.6 m, this would put the oxygen transfer efficiency in the range of 1.4 to 2.8%, depending on the bed depth.

The linear diffuser tubing used in this study was a fine bubble aeration device. In an open-water tank, the expected oxygen transfer would be approximately 6.6% per meter. The lower-than-expected oxygen transfer rate could speculatively be from the influence of the granular media, which may serve to coalesce bubbles and increase their size, reducing the oxygen transfer efficiency.

Air-induced mixing of the wetland was greatly reduced by the presence of the granular bed media; based on visual



**FIGURE 24.5** Testing of HSSF wetland aeration system for BTEX degradation; Casper, Wyoming. Rows of bubbles represent the location of linear air diffuser tubes, which were placed in radial arcs within this circular wetland. The wetland basin was artificially flooded to allow visual observation of the bubble pattern.

observations, the influence of aeration was generally restricted to a radius of approximately 15 cm from the point of bubble introduction (Wallace *et al*., 2007b). Aerated wetlands are hydraulically inefficient, as shown by tests yielding number of tanks in series (*N*TIS) of 1 to 3. It is apparent that a very uniform aeration grid is required to maintain oxygenated conditions in the wetland bed. Figure 24.5 shows the test of such an aeration system.

Aeration of SSF wetlands is not a maintenance-free activity. Without aeration, the system will fall back into the operating envelope of nonaerated wetlands. However, because regulatory compliance is dependent on the aeration system, scheduled, proactive maintenance of blowers and other aeration components need to be carried out by the system operator. Wetland designers should also consider the fouling of air diffusers within SSF wetlands, and provisions to replace or chemically clean diffuser assemblies should be addressed.

### Carbon Additions to HSSF Wetlands

Early work on denitrification in HSSF wetlands was conducted by Gersberg *et al*., 1984. HSSF wetlands were fed with secondary effluent low in BOD (3.3 mg/L) and high in nitrate (18.8 mg/L). In initial trials, the removal of total nitrogen was limited by the availability of organic carbon, and ranged between 9 and 19% at hydraulic loading rates of 8.4 to 20 cm/d. Utilization of methanol as a supplemental organic carbon supply dramatically increased denitrification efficiency, up to 97% at a hydraulic loading rate of 16.8 cm/d (Gersberg *et al*., 1984). As a lower-cost carbon supply, plant biomass was mulched and placed on top of the beds. At a hydraulic loading of 8.4–12.5 cm/d, the mean removal rate was 89% for total nitrogen, although the carbon loading (0.09 kg/m$^3$) was higher than that required for methanol (0.03 kg/m$^3$). This finding is consistent with more recent studies on the bioavailability of plant detritus to fuel denitrification (see, for instance, Ingersoll and Baker, 1998; Baker, 1998; Hume *et al*., 2002a).

Use of organic materials as a thermal insulator has the drawback of producing leachates with BOD that adds to the loading for that contaminant in domestic and municipal systems (Wallace *et al*., 2001). However, in systems treating low BOD sources, such as many runoff and groundwater remediation projects, such an additional source of organics is desirable and may be essential to the proper functioning of the wetland. Surface mulching of HSSF wetlands is also an option for supplying carbon for denitrification of nitrate-rich influents, as demonstrated by Gersberg *et al*., 1984. Mass-balance considerations to fuel denitrification processes are discussed in Chapter 9.

VF wetlands are effective in oxidizing ammonia to nitrate, but typically are not efficient in denitrification processes (Cooper, 2001). One of the limitations of denitrification is availability or organic carbon; and VF wetlands are often combined with HSSF wetland stages for this purpose (Seidel, 1973; Brix, 1994d; Cooper, 2001).

## 24.3 OPERATIONAL STRATEGIES

The flow of water need not be continuous, nor need it be unidirectional from inlet to outlet of a wetland or integrated natural system. There are other methods of operation, involving spatial or temporal sequencing, and the potential of recycle flows. Some of these concepts are explored in the following text.

### Step Feed

Step feeding refers to the introduction of water to a system at several points along the flow path. In activated-sludge mechanical plants, it refers to the introduction of settled wastewater at several points along the flow in a plug-flow aeration tank (Metcalf and Eddy Inc., 1991; Crites and Tchobanoglous, 1998). In water hyacinth systems, as many as eight points of introduction have been used (Water Environment Federation, 2001). The Gustine, California, FWS system was



**FIGURE 24.6** The step feed distribution pipe in the Gustine, California, system.

built with provisions for step feeding (U.S. EPA, 1988b). The reason for step feeding is, in all cases, stated to be the avoidance of "overloading" of the inlet zone of the system. Because the concept of loading to an interior zone of the wetland is fuzzy, a hypothetical situation is considered.

The Gustine example is selected as the framework, for which a step feed at the one-third point of the length is possible (Figure 24.6). The original design called for either one third or two thirds of the flow to be introduced at this interior point (U.S. EPA, 1988b). It is known that the decline of BOD through this system is exponential (see Figure 8.7, Chapter 8). Because of the high length-to-width ratio (337 m:11.6 m = 29:1), the plug-flow version of the $k$-$C^*$ model will be used. Calibration to the Gustine data of Walker and Walker (1990) yields $k = 48$ m/yr and $C^* = 4$ mg/L. The average BOD entering the system was about 200 mg/L, and the hydraulic loading was 3.3 cm/d. Figure 24.7 shows the effect of the step feed of one third of the influent to the one-third point. The front end of the wetland receives a lesser flow and does a more effective job of reduction of concentration. Upon blending with the step feed, the concentration goes up above that which would occur without step feed and stays above for the rest of the wetland travel distance. The outlet concentration is 29% higher for the step-feed option. It may be shown that for any step-feed operation and first-order kinetics, step feed gives lesser performance.

The overall BOD load to the wetland is not changed by step feeding because the area is fixed, and the inlet flow and concentration are fixed. The question then becomes: what has been accomplished? If all that is considered is BOD, the answer is, nothing. However, if it is presumed that part of BOD removal has to do with the sedimentation of solids, then TSS accumulation might factor into our thinking. The Gustine system reduced TSS from 75 mg/L to about 30 mg/L. Suppose that the entire accretion is to an inlet zone of 37 m length (1/9 of the length), at a bulk density of 0.5. Then the buildup would be about 1 cm per year. By step feeding, one-third of that accretion would be transferred to the step-feed zone, and the inlet buildup would be only two thirds of a centimeter per year. Therefore, the evaluation of step feeding as

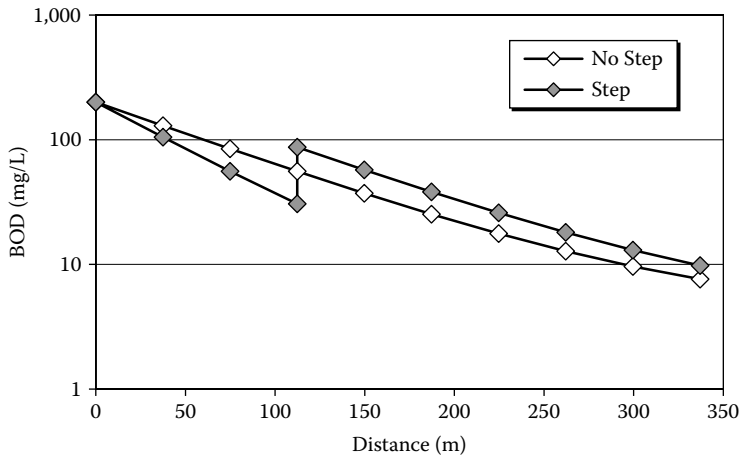



**FIGURE 24.7** Effect of step feed on BOD concentrations. These are model-based calculations, based on calibrations to the Gustine, California, system.



**FIGURE 24.8** A leachate seepage outbreak at the Saginaw, Michigan, closed landfill. This water contained ammonia at concentrations sometimes exceeding 1,000 mg/L, and was severely toxic to all vegetation.

an alternative depends upon the necessity for spreading out the solids and the corresponding penalty in performance.

## Recycle

Recycle can serve two important purposes in treatment wetlands: (1) it brings products of reactions in the wetlands from the exit back to the inlet, and (2) it dilutes reactants entering the system. Both of these functions can be used to advantage in some wetland situations.

### Dilution of Influents

Wetland vegetation can be sensitive to high concentrations of some pollutants in the water. Therefore, strong influents can damage or destroy vegetation, and the problem will be most severe in the inlet of the wetland before any treatment has occurred. For example, landfill leachates can be inimical to wetland vegetation (Figure 24.8). Recycling treated water from the wetland system outlet can serve to dilute the incoming water, but at some price in terms of cost and level of treatment.

To illustrate the concept, a hypothetical wetland is considered, with a recycle of water back to the inlet. A fresh-feed Damköhler number $k/q = 2.0$, $P = 4$ TIS, and $C^* = 0$ are chosen, to be representative of a moderate degree of removal of the contaminant (80%). Figure 24.9 shows the calculated concentration profiles through the system for differing amounts of recycle up to 200% of the incoming raw water. Note that as the recycle ratio goes to very high levels, the effect is to create a single well-mixed unit because of the intense recirculation. The limiting concentration reduction for this extreme is 67%. Typical recycle rates are 200% or less, for which the reduction is 72% for this example. Thus, a price in performance has been incurred by recycle. However, the concentration in the inlet region is cut in half (52%) by 200% recycle.

The economic consequences of a recycle pump and piping may not be large for very small systems, but for large FWS wetlands they may be a significant contribution to cost. Recycle is being used on the large Everglades-constructed wetlands to process seepage collected around their perimeters.

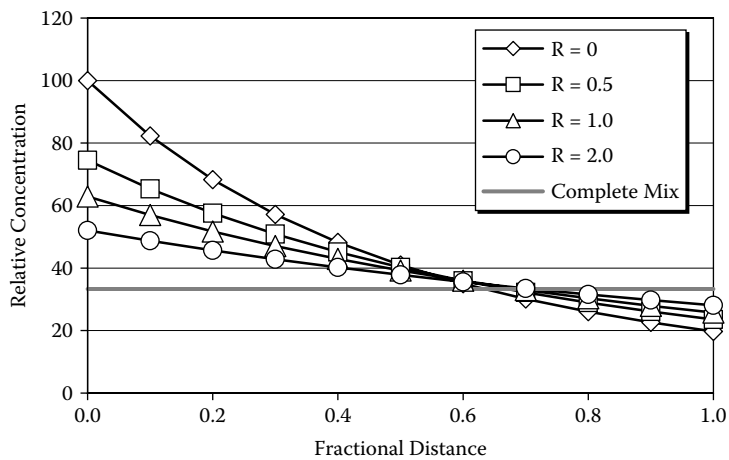



**FIGURE 24.9** The effect of recycle on a hypothetical wetland. The fresh-feed Damköhler number is $k/q = 2.0$, $P = 4$, and $C^* = 0$.

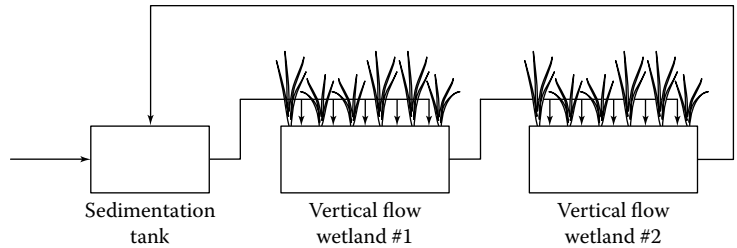



**FIGURE 24.10** Basic components of the experimental recycle VF system in Denmark. (Adapted from Brix *et al*. (2002a) *BOD and Nitrogen Removal from Municipal Wastewater in an Experimental Two-Stage Vertical Flow Constructed Wetland System with Recycling*. Mbwette (Ed.). Proceedings of the 8th International Conference on Wetland Systems for Water Pollution Control, 16–19 September 2002; Comprint International Limited: University of Dar Es Salaam, Tanzania.)

### Recycled Nitrate

One of the dilemmas of once-through treatment wetlands is that carbon (BOD or COD) is utilized in the front end of the wetland, prior to the nitrification of a significant fraction of any ammonia that may be present. This is due to the more rapid rate of elimination of BOD, compared to ammonia. Thus, although both processes occur more or less simultaneously in wetlands, the carbon is gone before a large share of the nitrate is formed. Therefore, denitrification is hampered by a lack of carbon in the exit section of the wetland. This phenomenon is present in all types of treatment wetlands but is especially problematic in VF systems. One "cure" is to add another wetland component with the capability of providing the carbon needed, which would usually be a FWS system with its litter decomposition as the carbon supply. Another possibility is to recycle the nitrate back to the inlet of the wetland, where there is a supply of carbon associated with the incoming fresh feed to the system.

For example, Brix *et al*. (2002a) performed comparison studies on a two-stage VF system treating municipal wastewater, with and without recycle (Figure 24.10; see also Figure 16.11, Chapter 16). Table 24.1 shows data from one pair of two-month campaigns, out of four that were conducted. Several conclusions can be drawn from these results. First, the wetlands are very effective in reducing $BOD_5$ and TSS, whether or not there is recycle. The sedimentation tank effectively reduces organic nitrogen, and the wetlands reduce it further to levels less than 1 mg/L, whether or not there is recycle. The wetlands are very effective in reducing ammonia, whether or not there is recycle. The major effect of recycle is in the reduction of oxidized nitrogen and consequently total nitrogen leaving the system. The effluent oxidized nitrogen was 38 mg/L without recycle, and 18 mg/L with recycle.

Because of the success of this strategy, the Danish guidelines for small on-site treatment wetlands require recycle at a recycle ratio of 50% (Brix and Arias, 2005).

## FWS Timed Operational Sequences

The flows to FWS wetlands need not be steady and continuous. Event-driven systems constitute one group of systems that receive nonsteady flows, and these have been discussed

**TABLE 24.1**
**Comparison of Recycled and Once-Through Operation of a Two-Stage Experimental System in Trige, Denmark**

| | Raw Input | | Combined | Sedimentation Tank Out | | Wetlands Out | |
|---|---|---|---|---|---|---|---|
| | No Recycle | Recycle | Input | No Recycle | Recycle | No Recycle | Recycle |
| Raw flow ($m^3$/d) | 2.14 | 1.88 | — | 2.14 | — | 2.14 | — |
| Recycle flow ($m^3$/d) | 0 | 2.37 | 4.25 | 2.14 | 4.25 | 2.14 | 4.25 |
| Recycle ratio (%) | 0% | 126% | — | — | — | — | — |
| Wetland HLR (cm/d) | — | — | — | — | — | 14.3 | 28.3 |
| $BOD_5$ (mg/L) | 322 | 251 | 113 | 241 | 138 | 3.7 | 3.3 |
| TSS (mg/L) | 400 | 279 | 125 | 100 | 62 | 1.4 | 2.0 |
| $NH_4$-N (mg/L) | 34 | 32 | 14 | 56 | 17 | <0.5 | <0.5 |
| $NO_x$-N (mg/L) | <0.5 | <0.5 | 10 | <0.5 | <0.5 | 38 | 18 |
| Organic N (mg/L) | 25 | 19 | 19 | 3 | 6 | 0.8 | 0.8 |
| Total N (mg/L) | 59 | 51 | 33 | 59 | 23 | 39 | 19 |

*Source:* Data from Brix *et al*. (2002a) *BOD and Nitrogen Removal from Municipal Wastewater in an Experimental Two-Stage Vertical Flow Constructed Wetland System with Recycling*. Mbwette (Ed.). Proceedings of the 8th International Conference on Wetland Systems for Water Pollution Control, September 16–19, 2002; Comprint International Limited: University of Dar Es Salaam, Tanzania.

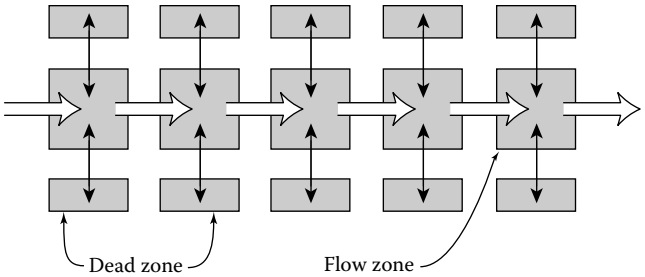



**FIGURE 24.11** Conceptual model of wetlands with dead zones under a fill-and-draw operation. This is the finite stage model with five stages (see Appendix B). (Adapted from Helsel (1992) *Fill and Draw Wetlands*. M.S. Thesis, University of Michigan, Ann Arbor.)

in Chapter 14. Another set of systems operate in the batch mode, and these also have been covered in Chapter 14. There remains another strategy that targets hydraulic inefficiency in what are otherwise continuous flow systems: a fill-and-drain operation.

Helsel (1992) explored the potential advantages of a fill-and-draw operation for a hypothetical FWS system describable by a finite stage model with five units (Figure 24.11; see Appendix B). Under steady operation, it was envisioned that some fraction of the wetland existed out of the main flow path, with essentially no flushing by the wastewater flow, termed a dead-zone fraction. In the long term, under steady operation, this reduces the effective wetland area. Under fill and draw, these dead zones are emptied and refilled to an extent determined by the fraction of the wetland water that is drained. Therefore, more of the wetland area becomes involved in treatment. The effectiveness then depends upon how the wetland performance depends upon depth. Helsel (1992) assumed a constant volumetric first-order rate coefficient, thus taking a penalty for decreased detention time at smaller water depths. A sample of results is given in Figure 24.12. In steady operation, percent removal drops off rapidly as the dead zone fraction increases. A fill-and-draw operation lessens the penalty but does not forecast improvement over full utilization of the wetland area. Therefore, proper design should eliminate the need to access the dead zones by raising and lowering water levels.

The Oxelösund FWS system in Sweden is operated in a partial fill-and-draw mode, in that the flow is alternated between two parallel inlet cells (Wittgren and Tobiason, 1994; Kallner and Wittgren, 2001). This system has highly irregular shape, with possibilities of dead zones (see Figure 1.19, Chapter 1). However, operational data reports do not include any comparative evaluation of performance.

## SSF Timed Operational Sequences

Most SSF wetlands operate with pulsed flow because of the use of pump-induced dosing in HSSF systems and the loading and resting operating regime deliberately used in many VF systems as a means of reducing bed clogging.

Although these can be considered Type *B* approaches to wetland design and operation, they are fairly standard in the wetland field, and numerous examples of these systems have been presented previously in this book.

### Fill-and-Drain SSF Wetlands

It has been demonstrated that the treatment performance of HSSF wetlands can be improved by fluctuating the water level. The basic operational sequence is best explained in terms of nitrogen and BOD removal (Austin *et al*., 2003; Maciolek and Austin, 2006):

1. The wetland basin is filled with water; ammonia is absorbed onto the bed media (McBride and Tanner, 2000); organic matter is degraded by microbial biofilms.
2. The wetland basin is drained; bulk water is routed to the next wetland basin. As water is drained, air enters the bed. Microbial biofilms are rapidly oxidized due to atmospheric diffusion, ammonia is converted to nitrate, and degradation of sorbed and particulate organic matter continues.

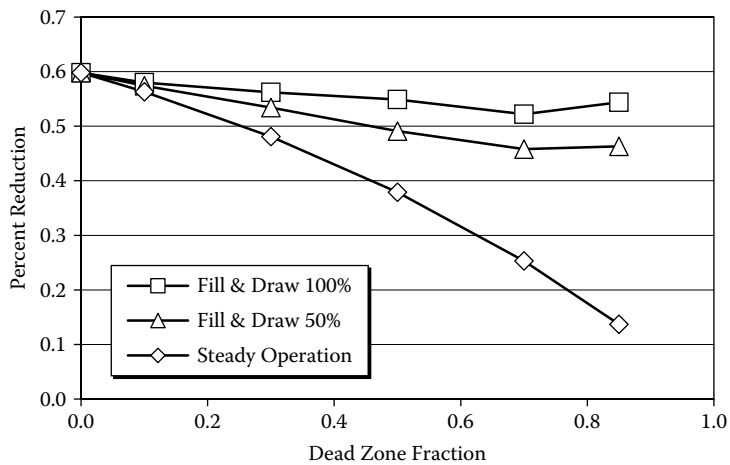



**FIGURE 24.12** Behavior of wetlands with dead zones under a fill-and-draw operation. (Data from Helsel (1992) *Fill and Draw Wetlands*. M.S. Thesis, University of Michigan, Ann Arbor.)

3. The wetland basin is filled a second time; due to the charge reversal, nitrate and nitrite are released from the bed media; influent organic matter provides a carbon supply for denitrification processes to drive total nitrogen reduction.

The filling and draining sequence can be done in two ways. The simplest approach is to use the influent flow to fill the wetland basin to the target water level. The wetland basin can then be immediately drained or allowed to operate as a continuous-flow, constant-water-level system for some percentage of the operational sequence. The treatment performance of a full-scale HSSF wetland in Minoa, New York, significantly improved after the facility was converted from a constant-flow to a fill-and-drain mode of operation (Liebowitz *et al*., 2000). The presence of sulfide precipitates in the wetland effluent was used as an indicator that the bed should be drained. The rate of cycling was about four days for the second-stage wetland basin, and about every two weeks for the first-stage wetland basin. A two-stage HSSF wetland system at Dolni Mesto in the Czech Republic also demonstrated improved treatment for BOD, COD, and ammonia when it was operated in a fill-and-drain mode instead of having a constant water level (Vymazal and Masa, 2003). The wetland basin was drained whenever it had filled to a target water level; fluctuations of 8 cm and 15 cm were evaluated, with a 15-cm fluctuation providing greater improvement. Influent flow rates were not measured, so the cycle frequency could not be determined.

The second operational strategy is to accelerate the fill-and-drain cycle frequency by pumping or recirculating the flow between several wetland beds. These types of wetlands are essentially vertical flow systems. A fill-and-drain pilot-scale wetland operating on these principles has been developed in Israel (Green *et al*., 1998). This system utilizes surface flooding of the bed to ensure uniformity in influent distribution, unsaturated flow through a deep bed (2.4 m) (Green *et al*., 2004), and a siphon or timer-controlled outlet. The rapid release of the effluent is desirable as air is drawn into the wetland bed through passive ventilation pipes, increasing the oxygen transfer (Green *et al*., 1997a). Such a system has shown successful treatment of dairy wastewater, although the each bed must be rested approximately 70% of the time to avoid clogging (Green *et al*., 2002). This implies that 3 or 4 beds in parallel would be needed to treat such a type of wastewater.

Fill-and-drain wetlands operating under an accelerated cycle frequency are commonly called "reciprocating" or "tidal flow" wetlands. The fill-and-drain operational sequence helps BOD removal and nitrogen removal to occur simultaneously, and some nitrate-bound oxygen may be recovered. To achieve the necessary degree of oxygen transfer to drive the treatment process, cells must be filled and drained multiple times per day; greater than 6 cycles per day is often needed to meet treatment goals (McBride and Tanner, 2000). The rate of fill-and-drain cycles is dependent on the overall oxygen transfer requirements, and will generally require an internal pumping system to fill and drain wetland cells independent of the influent flow rate (Cooper, 1999; Sun *et al*., 1999).

Fill-and-drain wetlands that rely on pumps to accelerate the cycle frequency are available as a patented technology in North America (Zoeller and Byers, 1999; Behrends, 1999a; Austin *et al*., 2002; Austin *et al*., 2004). In one variation of this technology, flow is pumped back and forth (reciprocated) between two wetland beds; this is known as the ReCip® process in the United States (Behrends *et al*., 1996b; Behrends, 1999a; Behrends *et al*., 2000).

Here, for the second time, we encounter a significant influence of patents on SSF treatment wetland technology. We cannot access the proprietary data, and hence cannot independently evaluate it. For most patented systems, only anecdotal, qualitative information has been presented. However, we can summarize the results from the open literature and provide interpretation of it. Nothing here should be taken as supporting or refuting the claims in existing patents.

There is a growing understanding that the cation exchange capacity (CEC) plays a fundamental role in the effective performance of fill-and-drain SSF wetlands. A recent laboratory study by Austin (2006) indicated that fill-and-drain columns filled with an expanded shale aggregate (CEC of approximately 4 meq/100 g) significantly outperformed columns filled with high-density polyethylene (HDPE) media (zero surface charge) despite the fact that the HDPE media provided more surface area for microbial biofilms. Expanded-shale columns outperformed HDPE in terms of TSS, COD, TKN, and total nitrogen (TN). Improved performance for ammonia (TKN) can be explained in terms of the CEC (Johns *et al*., 1998; Gisvold *et al*., 2000). The reasons for improvement for other parameters are less clear.

Fill-and-drain wetland reactors are also susceptible to bed plugging despite the reactors being flushed several times per day, and there is the apparent potential to "purge" fine mineral sediments. Behrends *et al*. (2006) found reciprocating beds were "severely" clogged after seven years.

One approach to predicting bed clogging has been advanced by Austin *et al*. (2006a), which utilizes a dimensionless ratio between the rate of reaction and the rate of mass transport to assess the potential for bed clogging based on Monod kinetics. The analysis is dependent on empirical methods to estimate the effective surface area available for microbial biofilms but has been successfully calibrated to mesocosm tidal-flow reactors, based on observed increases and decreases in microbial biomass as a function of organic loading. It is interesting that the short-term clogging results of Figure 21.5, Chapter 21, can be predicted using this method, indicating that Monod-based predictions of microbial biomass will likely be a part of the more advanced design methods that will inevitably be presented as treatment wetland technology continues to evolve.

## 24.4 INTEGRATED NATURAL SYSTEMS

Increasing demands for improved treatment, together with the limitations of each individual wetland type, makes the selection of multiple-unit systems inevitable in many situations. The alternative wetland categories, FWS, HSSF, and

**TABLE 24.2**
**Comparison of Strengths and Weaknesses of Various Unit Ecosystem Alternatives**

| | | Relative Pollutant Reduction Potential | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Unit Ecosystem** | **Relative Cost per Area** | **$BOD_5$** | **TSS** | **$NH_4$-N** | **$NO_x$-N** | **TP** | **Metals** | **Pathogens** | **Organics** |
| **Wetland Alternatives** | | | | | | | | | |
| FWS marsh | L | H | H | M | H | M | H | H | H |
| FWS floating aquatic plant | M | H | H | L | M | M | M | H | M |
| FWS submerged aquatic plant | L | M | H | H | M | H | H | H | H |
| FWS algal-dominated | L | M | M | H | M | H | H | M | M |
| HSSF | H | H | H | L | H | M | H | H | H |
| VF | H | H | H | H | L | L | L | L | L |
| **Related Natural Technologies** | | | | | | | | | |
| Slow-rate land application | H | H | H | H | L | H | H | H | H |
| Overland flow | H | H | M | H | L | M | M | M | M |
| Rapid infiltration | H | H | H | H | L | M | M | M | M |
| On-site filtration | H | H | H | H | L | M | M | M | M |
| Facultative ponds | L | H | L | M | M | M | M | H | M |
| Aerated ponds | H | H | M | H | L | M | M | H | M |
| High-rate algal ponds | H | H | L | H | L | M | M | H | M |

*Note:* L = Low, M = Moderate, H = High.

VF, are all being used as "stand-alone" wastewater treatment technologies. They each have the potential to provide effective treatment of a variety of pollutants as long as loading rates are appropriate and the desired plant community is maintained. Although these technologies work alone, it is often beneficial to combine one or more of these options into integrated natural-treatment systems.

In many sectors of industry, including wastewater treatment, there exists a set of processing units that may be combined to develop the integrated plant that performs the required tasks. These may include filters, reactors, clarifiers, and biological treatment units of several kinds (Metcalf and Eddy Inc., 1991). There are also several kinds of land-based units that comprise the suite of "natural" treatment systems (Water Environment Federation, 2001). These include wetlands and ponds of various types, which are here termed unit ecosystems.

Table 24.2 summarizes the major advantages of several unit ecosystem technologies described in this book or elsewhere. Each of the various wetland alternatives described previously provides numerous treatment benefits. However, they each have certain water quality treatment properties that outshine the companion technologies. For example, constructed FWS treatment wetlands provide excellent reduction of TSS and denitrification of nitrate and nitrite nitrogen, but they are not extremely effective at nitrifying ammonium nitrogen. HSSF treatment wetlands are also effective at TSS and nitrate reduction but at a much higher unit cost. However, these systems clearly excel at avoiding potential impacts of poorly treated wastes on wildlife and humans by keeping the wastewater belowground. VF treatment wetlands have excellent nitrification potential, in addition to their benefits for reduction of BOD and TSS. However, they are not very effective for denitrification.

Unit ecosystems can be combined in series or in parallel to provide flexibility and to take advantage of higher pollutant transformation rates characteristic of each ecosystem type. This section provides suggestions for integrating these diverse ecosystem units into optimal treatment systems. Several examples are presented of existing integrated systems.

## Ponds and Wetlands

### High-Sediment Waters

Many urban stormwater treatment systems include sedimentation basin forebays (pond elements) followed by emergent SF wetlands (Strecker *et al*., 1992; Schueler, 1992). Figure 18.31, Chapter 18, provides an example of a stormwater treatment system that incorporates this idea. This series combination has the advantage of providing for easy removal of accumulated sediment, together with further processing and reduction of concentrations of dissolved pollutants in the marsh. Many stormwater wet-detention systems combine these ecosystem units in a single basin.

In the agricultural landscape, more complex arrangements of ecosystem units in series have been utilized. The nutrient–sediment control system (NSCS) developed by the U.S. Natural Resources Conservation Service (NRCS) is composed of five units in series: sedimentation basin, overland flow, FWS emergent marsh wetland, pond, and overland flow (Figure 24.13) (Wengrzynek and Terrell, 1990; U.S. Department of Agriculture, 1991; Higgins *et al*., 1993). This system is intended to intercept the TSS from row crops (i.e., potatoes), and the inlet basin will fill. For an example operation in Maine, accretion

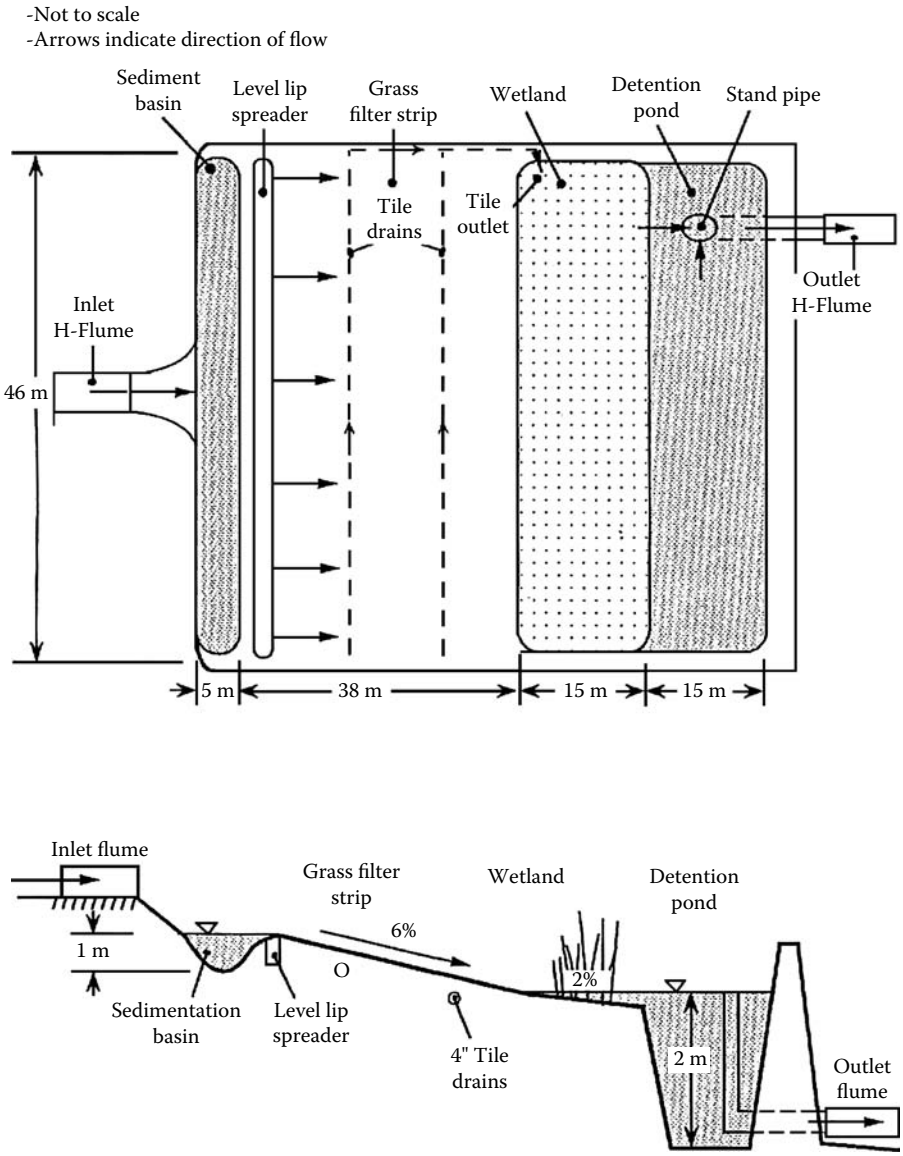



**FIGURE 24.13** Plan and profile views of an example of the NRCS nutrient–sediment control system. (From Higgins *et al.* (1993) In *Constructed Wetlands for Water Quality Improvement*. Moshiri (Ed.), Lewis Publishers, Boca Raton, Florida, pp. 359–367. Reprinted with permission.)

was about 3 cm/yr, and a 30-cm depth would require removal on about a ten-year cycle (Higgins *et al*., 1993).

Some continuous-flow applications may also entail high TSS in the entering water. For instance, the Tucson Sweetwater wetland receives filter backwash with high TSS, and this material must be removed from the inlet deep zones (see Figure 7.13, Chapter 7).

## Pond and Wetland Combinations

Wetlands possess the potential to be a valuable adjunct to treatment lagoons. They need not replace lagoons because those aquatic systems provide reasonable treatment for many water quality parameters such as BOD and TSS. A relatively common combination of natural technologies for municipal wastewater treatment found in the United States includes a facultative lagoon or pond for primary and secondary treatment, followed by a constructed FWS wetland. Kadlec (2003d) lists 21 such systems, and there are more being developed around the world (for example, see Browne and Jenssen, 2005; Polprasert *et al*., 2005; Herrera-Melian *et al*., 2006). This integrated system utilizes a minimum amount of fossil-fuel energies, generates minimal solids requiring disposal (infrequent pond-sediment removal), and is relatively inexpensive to construct. Almost any advanced wastewater treatment effluent standards are achievable as long as adequate land is available.

Facultative and aerated ponds are time-tested systems for treating wastewater to approximately secondary standards (Water Environment Federation, 2001). Modern improvements call for multiple cells of varying design, designated as advanced integrated wastewater ponds (Shilton, 2005). Table 24.3 provides a sense of the effluent concentrations that may be expected from anaerobic, facultative, and aerated ponds treating domestic wastewater. The general sense is that

**TABLE 24.3**
**Typical Effluent Concentrations from Lagoons Treating Domestic Sewage**

| Lagoon Type | TSS (mg/L) | BOD (mg/L) | Ammonia (mg/L) | Phosphorus (mg/L) | Fecal Coliforms (MPN/100 mL) |
|---|---|---|---|---|---|
| Facultative | 40–100 | 30–40 | 10–20 | 3–5 | 200–10,000 |
| Aerated | 20–40 | 30–60 | 8–15 | 3–5 | 30–100 |
| Partial-mix aerated | 20–40 | 30–60 | — | — | — |
| Anaerobic | — | 90–180 | — | — | — |

*Source:* Data from Crites and Tchobanoglous (1998) *Small and Decentralized Wastewater Management Systems.* McGraw-Hill, New York; Water Environment Federation (2001) *Natural Systems for Wastewater Treatment (WEF Manual of Practice FD-16).* 2nd edition, Water Environment Federation, Alexandria, Virginia; Mara (2005) *Pond Treatment Technology*. Shilton (Ed.), IWA Publishing, London, United Kingdom.

lagoons may not meet secondary (30 TSS, 30 BOD) standards in all cases. Pathogens, as measured by fecal coliforms, may be too abundant to meet human health standards (typically 200/100 CFU/100 mL in North America). Additionally, there is likely to be too much ammonia to meet ammonia toxicity standards for fish, which may require no more than 2–4 mg/L ammonia in the discharge. Finally, eutrophication of receiving waters may dictate phosphorus removal to 1.0 mg/L or below, which cannot be met with lagoon treatment alone.

Wetlands offer an alternative way to improve the quality of lagoon discharges. They are in use for reduction of TSS and BOD, nutrients nitrogen (N) and phosphorus (P), and pathogens (fecal coliforms, FC). Table 24.4 represents a collection of summary data from North American wetlands that are used in conjunction with lagoons. There are very large differences in size, loadings, and design purposes. A summary of removal percentages based on Table 24.4 shows that some of these wetlands are very successful in improving water quality, primarily those with lower loadings.

Facultative ponds utilize algae for the predominant removal functions. There is a strong connection between the phytoplankton (algae) produced in a lagoon and the TSS leaving that lagoon. These algal solids can cause lagoon TSS in the range 60–140 mg/L, which is well above secondary treatment standards. A FWS treatment wetland is one option for reducing lagoon TSS.

An interesting application of pond plus wetland technology is a stormwater/snowmelt system in Anchorage, Alaska. Snow removal from streets is accumulated in a snow dump, which contains the northern version of urban runoff constituents. In the summer, urban runoff is routed directly to the pond. In spring and early summer, the meltwater from the storage pile is conducted to the constructed pond (Figure 24.14). Water then overflows a low bank into the constructed wetland, from which it is discharged to a receiving stream. The wetland was constructed by excavating frozen blocks of peat and associated vegetation, and placing them in cross-rows in the constructed wetland basin. This herringbone structure sagged and collapsed during warm conditions, creating an array of interconnected peat berms and open water zones. These subsequently colonized with *Typha* and other local wetland plants.

### Confined Animal Feeding Operation Wastewater Management Systems

Wastes from confined animal feeding operations (CAFOs) are biodegradable, but they tend to have very high concentrations of oxygen-demanding substances. Many CAFOs presently discharge partially treated wastes to nearby lakes, creeks, streams, and rivers. There is a growing awareness of the need to control discharges from many of these sites to protect downstream water quality resources. Treatment of CAFO wastes has often been limited to ponds meant to trap and degrade manure solids and animal urine. These ponds do not typically treat the wastes to levels that can be safely discharged to receiving waters and often do not even provide enough treatment to safely irrigate croplands while protecting groundwater resources (Table 24.5). As discussed in Chapter 25, there has been a growing interest in building integrated natural systems to improve the quality of waters leaving these sites. A typical integrated system includes one or more pretreatment ponds and a constructed FWS wetland (see Figure 24.13).

## FWS and Infiltration

### FWS to Infiltration

Water from a FWS treatment wetland may be directed to an infiltration system for final effluent polishing or disposal. This alternative is used when there is no acceptable surface water body to accept treated effluent, and there is adequate water quality to protect the surficial groundwater aquifer for its designated uses. The unit ecosystem is a rapid-infiltration bed (Water Environment Federation, 2001), also known as a soil aquifer treatment system (Crites *et al*., 2006). Water is treated as it percolates downward through a soil matrix. Physical, chemical, and biological processes operate during this unsaturated flow to groundwater. Application is usually intermittent, and vegetation plays no role in the process. Desirable soil characteristics include high permeability, and that usually means sand. The depth to groundwater would be several meters, unless the percolate is collected in an underdrain system (Water Environment Federation, 2001).

## TABLE 24.4
## Characteristics of Pond–Wetland Systems by Latitude

| System and Data Source | | Years | Area (ha) | Latitude | Annual Op | | HLR (cm/d) | TSS (mg/L) | BOD (mg/L) | $NH_3$ (mg/L) | TP (mg/L) | FC (MPN/100 mL) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| West Jackson Co., Mississippi | P-W | 11 | 23 | 30 | Y | in | 3.1 | 40 | 29 | 3.9 | — | — |
| U.S. EPA (1999) | Con | | | | | out | | 14 | 8 | 1.2 | — | — |
| Fort Deposit, Alabama | AP-W | 24 | 6 | 31 | Y | in | 0.8 | 92 | 34 | 3.1 | — | — |
| Bays and Knight (2002) | Con | | | | | out | | 9 | 5 | 1.0 | — | — |
| Pontotoc, Mississippi | P-W | 2 | 0.04 | 34 | Y | in | 1.27 | 96 | 49 | 112 | 29 | — |
| U.S. EPA (1999) | Con | | | | | out | | 31 | 23 | 36 | 18 | — |
| Duplin, North Carolina | P-W | 6 | 0.024 | 35 | Y | in | 4 | — | — | 120 | — | — |
| Unpublished Data | Con | | | | | out | | — | — | 21 | — | — |
| Kingman, Arizona | P-W | 6 | 9 | 35 | Y | in | 4.1 | 46 | 50 | 22 | — | — |
| Gerke *et al.* (2001) | Con | | | | | out | | 3 | 7 | 11 | — | — |
| Benton, Kentucky 1 | P-W | — | 1.5 | 36 | Y | in | 4.7 | 57 | 26 | 7.7 | 4.54 | 9 |
| U.S. EPA (1999) | Con | | | | | out | | 11 | 10 | 7.9 | 4.22 | 14 |
| Benton, Kentucky 2 | P-W | 2 | 1.5 | 36 | Y | in | 4.7 | 57 | 26 | 7.7 | 4.54 | 9 |
| U.S. EPA (1999) | Con | | | | | out | | 12 | 16 | 6.43 | 3.98 | 20 |
| Listowel, Ontario 2 | P-W | 4 | 0.91 | 43 | Y | in | 2.8 | 23 | 20 | 7.2 | 1.1 | 1,736 |
| Herskowitz (1986) | Con | | | | | out | | 9 | 15 | 5.1 | 0.8 | 458 |
| Listowel, Ontario 5 | AP-W | 4 | 0.94 | 43 | Y | in | 2.6 | 111 | 56 | 8.6 | 3.2 | 222,990 |
| Herskowitz (1986) | Con | | | | | out | | 12 | 15 | 7.9 | 1.0 | 2,317 |
| Bellaire, Michigan | P-W | 12 | 7 | 44 | Y | in | 1.0 | — | — | 8.54 | 2.78 | — |
| Kadlec and Bevis (1990) | Nat | | | | | out | | — | — | 0.73 | 0.30 | — |
| Houghton Lake, Michigan | P-W | 24 | 70 | 44 | N | in | 0.3 | 30 | 17 | 6.1 | 3.15 | 89 |
| Unpublished Data | Nat | | | | (warm) | out | | — | 4 | 0.08 | 0.04 | — |
| Brighton, Ontario | P-W | 2 | 6 | 44 | Y | in | 4.6 | 21 | 19 | 11 | 0.52 | 1,180* |
| Unpublished Data | Con | | | | | out | | 9 | 9 | 8 | 0.27 | 357* |
| Cannon Beach, Oregon | P-W | 17 | 7 | 45 | N | in | 25 | 39 | 31 | — | 12.2 | 19 |
| Unpublished Data | Nat | | | | (dry) | out | | 9 | 8 | — | 4.6 | 53 |
| River Hebert, Nova Scotia 1 | P-W | 2 | 1.9 | 45 | Y | in | 0.91 | 22 | 21 | — | 3.21 | 10,800 |
| Hanson (2002) | Con | | | | | out | | 12 | 5 | — | 0.55 | 48 |
| River Hebert, Nova Scotia 2 | P-W | 2 | 6.5 | 45 | Y | in | 0.25 | 29 | 27 | — | 3.09 | 11,550 |
| Hanson (2002) | Con | | | | | out | | 15 | 8 | — | 0.31 | 26 |

(*Continued*)

**TABLE 24.4 (CONTINUED)**
**Characteristics of Pond–Wetland Systems by Latitude**

| System and Data Source | | Years | Area (ha) | Latitude | Annual Op | | HLR (cm/d) | TSS (mg/L) | BOD (mg/L) | $NH_3$ (mg/L) | TP (mg/L) | FC (MPN/100 mL) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Mt. Angel, Oregon | P-W | 11 | 3.6 | 45 | N | in | 13.3 | 11 | 25 | 2.6 | 1.5 | — |
| Unpublished Data | Con | | | | (wet) | out | | 8 | 14 | 2.2 | 1.4 | — |
| Lake Nebagamon, Wisconsin | P-W-I | 1 | 0.8 | 46 | N | in | 2.7 | 30 | 10 | 4 | — | — |
| Unpublished Data | Con | | | | (warm) | out | | 10 | 4 | 0.05 | — | — |
| Drummond, Wisconsin | P-W | 24 | 6 | 46 | N | in | 0.14 | — | — | 0.13 | 2.36 | — |
| Unpublished Data | Nat | | | | (warm) | out | | — | — | 0.17 | 0.94 | — |
| Minot, North Dakota | P-W | 12 | 51 | 48 | N | in | 3 | 32 | 16 | 4.5 | — | 381 |
| Hammer and Burckhard (2000) | Con | | | | (warm) | out | | 14 | 10 | 2.1 | — | 219 |
| Estevan, Saskatchewan | P-W | 4 | — | 49 | N | in | 3 | 33 | 9 | 2.82 | 2.49 | — |
| Unpublished Data | Con | | | | (warm) | out | | 24 | 5 | 0.35 | 2.18 | — |
| Musselwhite, Ontario | P-W | 4 | — | 54 | N | in | 55.1 | — | 6 | 12.8 | 0.018 | — |
| Bishay and Kadlec (2005) | Nat | | | | (warm) | out | | — | 5 | 3.9 | 0.012 | — |
| **Removals** | | | | | | | | | | | | |
| **Maximum** | | | | | | | | 93% | 87% | 99% | 99% | 99.8% |
| **Minimum** | | | | | | | | 27% | 17% | –33% | 7% | –179% |
| **Mean** | | | | | | | | 65% | 58% | 50% | 49% | 20% |
| **Median** | | | | | | | | 67% | 61% | 61% | 48% | 72% |

*Note:* Years = operational period (op). Operation is yearlong (Y), warm, wet, or dry season. P = pond, AP = aerated pond, W = wetland, I = infiltration, Con = constructed, Nat = natural, * = *E. coli*.

*Source:* Updated from Kadlec (2003d) *Water Science and Technology* 48(5): 1–8.



(a) (b)

**FIGURE 24.14** An interesting application of pond plus wetland technology is this system in Anchorage, Alaska. Snow removal from streets is accumulated in a snow dump (a), which contains the northern version of urban runoff constituents, leading to the black appearance. The meltwater is conducted to a constructed pond (b, foreground), and then overflows a low bank into the constructed wetland (a, background).

Hydraulic loading rates are commonly in the range of 1–100 m/yr, with application periods of 1–9 days and drying periods of 5–20 days (Crites and Tchobanoglous, 1998). Passage through the soil results in BOD, TSS, and pathogen removal. Depending on soil sorption characteristics, there can be considerable reductions of metals and phosphorus. Nitrogen is removed by soil sorption of ammonia on the application part of the cycle and subsequent nitrification during the drying portion of the cycle. Design procedures are detailed in (Crites and Tchobanoglous, 1998; Water Environment Federation, 2001; Crites *et al*., 2006).

The sequence of pond–wetland–infiltration has been implemented at Lake Nebagamon, Wisconsin (Figure 24.15), and at Isanti-Chisago, Minnesota, for example (see Figure 25.17, Chapter 25). In both cases, the goal was disposal of treated water but with the expectation that further polishing would take place in the soil profile.

### Infiltration to FWS

There are also systems that reverse the order, using the sequence pond–infiltration–wetland. For instance, the Onaway, Michigan, system uses this sequence so that phosphorus control can be exerted prior to the wetland, which then serves as a polishing unit ecosystem (Figure 24.16). When this sequence is used, the sand beds provide nitrification, but

**TABLE 24.5**
**Typical Effluent Concentrations (mg/L) from Lagoons Treating Animal Wastes**

| | Dairy Cow | Beef | Pig | Poultry |
|---|---|---|---|---|
| **Anaerobic Ponds** | | | | |
| $BOD_5$ | 200–1,200 | 200–2,500 | 230–3,600 | 600–3,800 |
| TSS | 1,100–47,000 | 500–1,300 | 460–9,200 | 2,000–8,000 |
| Ammonia | 50–720 | 180–265 | 200–2,000 | 330–1,500 |
| Total N | 75–1,750 | 180–530 | 300–3,000 | 800–3,600 |
| Total P | 25–200 | 150–480 | 75–300 | 200–360 |
| **Anaerobic/Facultative Ponds** | | | | |
| $BOD_5$ | 65–240 | — | 100–1,700 | 500–1,500 |
| TSS | 150–300 | 300 | 100–450 | — |
| Ammonia | 40–100 | — | 220–300 | 200–900 |
| Total N | 69–480 | — | 300–480 | 200–900 |
| Total P | 57–240 | — | 75–240 | 80–360 |

*Source:* Based on Sukias and Tanner (2005) In *Pond Treatment Technology*. Shilton (Ed.), IWA Publishing, London, United Kingdom.



**FIGURE 24.15** The Lake Nebagamon, Wisconsin, infiltration beds developed into a constructed wetland that infiltrates the treated water from a lined constructed wetland.

there is not enough BOD to fuel denitrification because BOD has been removed in the aerated pond. However, the FWS unit ecosystem generates its own carbon supply and therefore provides denitrification for the sand-bed effluent.

### Overland Flow to FWS

Overland flow is a unit ecosystem that incorporates shallow sheet flow over a grassed slope. It is essentially a fixed-film biological treatment system in which grass and litter form the matrix (Water Environment Federation, 2001; Crites *et al*., 2006). The soils should be relatively impermeable, and the site slope should be in the range of 2–8%. Slope lengths vary from 20 to 80 m, and water delivery rates are on the order of 2–10 $m^3$/d·m of cross-width. Drying periods are part of the operation, with ammonia removal improving as the wet/dry ratio shrinks to about 0.25 (Crites *et al*., 2006). Design procedures are detailed in (Crites and Tchobanoglous, 1998; Water Environment Federation, 2001; Crites *et al*., 2006). Depending upon the carbon availability in the wastewater, denitrification may or may not be complete. Overland flow is one method of distributing water into a FWS wetland while providing pretreatment. The treatment wetland at the Orange County Eastern Service Area near Orlando, Florida, effectively uses overland flow as the inlet distributor (Schwartz *et al*., 1994) (Figure 24.17).

### Vertical Flow to FWS

Although infiltration and overland flow are sometimes acceptable alternatives to provide nitrification, the VF wetland (sand filter) with intermittent flow is often a more



**FIGURE 24.16** The Onaway, Michigan, infiltration beds. See Figure 18.29, Chapter 18, for a view of the constructed wetlands that follow. Aerated lagoons precede these beds.



**FIGURE 24.17** The inlet to the Orange County Eastern Service Area constructed wetlands near Orlando, Florida. The area in the foreground is a grassed overland flow zone. The shrub area in the background is the first unit of the constructed wetland.

compact unit ecosystem that can also provide that nitrification. However, the VF process also removes BOD and in fact nitrifies more effectively at low inlet BOD levels. Therefore, a follow-on FWS wetland, which can generate the carbon needed for denitrification, completes a logical sequence that both nitrifies and denitrifies.

One example of such an integrated process operates for the Lamb–Weston potato-processing facility in Connell, Washington (Kadlec *et al*., 1997; Burgoon *et al*., 1999; Burgoon, 2001). The first element of this treatment train is a parallel, 10-ha pair of FWS emergent wetland cells that function primarily as a trap for organic solids to decrease chemical oxygen demand (COD), and to foster the conversion of organic N to ammonium (Figure 24.18) (see also Figure 25.8, Chapter 25). This unit ecosystem is necessary because the incoming TSS is about 300 mg/L and would clog a VF unit quite quickly. The incoming COD is about 3,000 mg/L and would impede nitrification. The incoming nitrogen is largely organic, which must be mineralized before nitrification can occur. A 4-ha set of parallel VF wetland cells then nitrify this ammonium N to nitrate N. The VF wetland cells are followed by two 1-ha FWS wetlands to denitrify this nitrate N and reduce the ultimate mass load of N prior to effluent disposal. This treated water is stored in a 17-ha pond prior to ultimate disposal via slow-rate land application on 213 ha of hayfields. The irrigation water rate to the fields is 57 cm/yr, and a remaining nitrogen load to the hay is 360 kg/ha·yr, which are the desired rates for hay production. A pilot version of this system was operated with two inlet FWSs, one VF, and one FWS in series. The nitrogen species transformations are shown in Figure 24.19.

A similar combination VF–FWS is used at the Saginaw, Michigan, site to treat landfill leachate (see Figure 25.14, Chapter 25).

### SSF and Infiltration

Early implementations of SSF wetland technology assumed that the wetland system could arbitrarily be divided 50/50 between treatment and infiltration (TVA, 1993), although later studies have indicated this is not necessarily a good assumption (White and Shirk, 1998), and current design

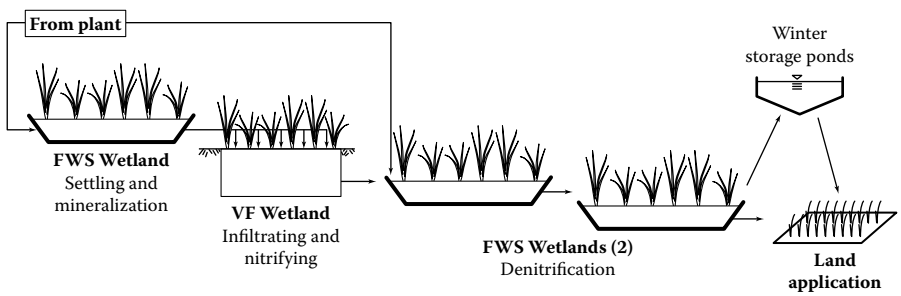



**FIGURE 24.18** Layout of the Lamb–Weston integrated natural system.

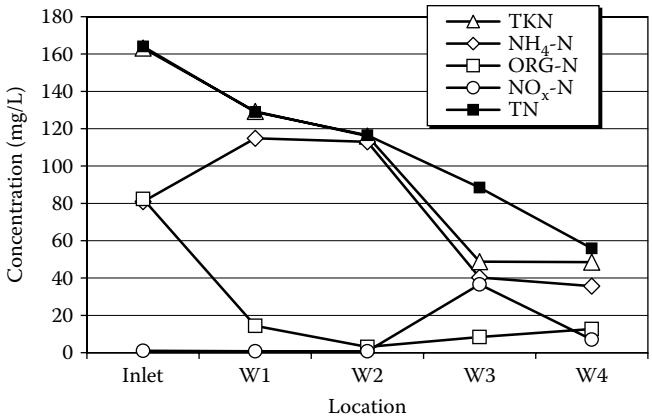



**FIGURE 24.19** Performance of the Lamb–Weston pilot integrated natural system.

guidelines recommend that infiltration systems be designed on soil morphology instead of prescriptive standards (U.S. EPA, 2002c).

Interactions between applied wastewater and the soil matrix are complex because a clogging layer develops at the soil/water interface (U.S. EPA, 2002c) in the soil infiltration system. This clogging layer both restricts flow and can aid in soil filtration processes such as pathogen removal. However, the cumulative organic loading to the soil matrix eventually leads to clogging of the soil and hydraulic failure of the infiltration system (Tyler and Converse, 1994).

The longevity of soil infiltration systems can be dramatically increased if the organic loading to the infiltration system is reduced (Tyler and Converse, 1994); if water is pretreated, the life of infiltration systems can be extended almost indefinitely because soil clogging develops at a much slower rate. Some regulatory agencies in the United States allow for reductions in infiltration area on the order of 25 to 50% if the water is pretreated to secondary standards prior to application to the soil infiltration system (Wisconsin Department of Commerce, 2000). Treatment wetlands have been used in this regard as a simple, low O&M treatment alternative (Axler *et al*., 2001). This is most likely to be a benefit where available land for replacement infiltration systems is limited or nonexistent.

### VF and HSSF Combinations

Although long regarded as separate technologies, combinations of VF and HSSF wetlands have in fact existed since the development of SSF wetland technology (Seidel, 1966). Both VF–HSSF and HSSF–VF combinations have been developed (EC/EWPCA Emergent Hydrophyte Treatment Systems Expert Contact Group and Water Research Centre, 1990; Brix, 1998; Cooper, 1999; Noorvee *et al*., 2005a) based on different treatment process objectives, and are considered standard process configurations. These combinations have been previously discussed in detail in Chapters 15, 20, and 21.

At the present time, two process configurations have been commonly employed:

*VF–HSSF combinations.* This approach dates back to the original work of Seidel (Seidel, 1966). VF wetlands (often two or more stages in combination) are used for TSS filtration, BOD removal, and nitrification; subsequent HSSF stages are used for effluent polishing (see Figure 15.9, Chapter 15).

*HSSF–VF combinations.* The design strategy with this approach is to use a large HSSF wetland as the initial stage to remove BOD and TSS, both of which can be removed in an anaerobic environment. This reduces the overall oxygen demand on the subsequent VF stage, which is designed primarily for nitrification (Brix, 1998).

Either process configuration is capable of producing a nitrified effluent. If total nitrogen reduction is a design goal, then recycle of the nitrified effluent back to the primary treatment tank is commonly employed (Laber *et al*., 1999). Individual unit processes may be designed as discrete elements using the procedures previously outlined in this book.

## SUMMARY

At the present stage of technology development, the performance envelope for Type *A* FWS and HSSF treatment wetlands can be fairly well defined based on the large number of operating, full-scale systems. Performance of these systems has been defined (based on available data) in Part I of this book. System modifications have been sought to increase that performance envelope.

Free water surface wetlands are rarely modified internally, partly because of their size and partly because of a desire to emulate natural wetlands. Their operational strategy may involve recycle or other steps to improve performance.

Type *B* engineered SSF wetlands remain an area of technology development. At the research scale, Type *B* approaches have been explored primarily to increase oxygen supply. Modified full-scale HSSF applications are much less common. The current spectrum of HSSF treatment wetlands encompasses both the Type *A* (passive) and Type *B* (engineered) design approaches. Any decision to use SSF wetland treatment technology must balance the requirements of land availability and simplicity versus the need for operator intervention and complexity, to meet regulatory goals.

Combinations of various wetland technologies have been presented throughout this book, and more specific applications have been provided in this chapter.

# 25 Wetlands by Application Group

The strategy of this book up to here has considered individual wastewater constituents by analysis of the available data and in terms of the types of wetlands that may be employed to reduce concentrations of those individual chemicals. That might be termed a horizontal approach because removal of, say, biochemical oxygen demand (BOD), is a crosscutting issue that affects many types of effluents. Primary emphasis has been on the reduction of specific contaminants, including BOD, total suspended solids (TSS), nutrients, metals, and hydrocarbons.

In this chapter, a "vertical" approach is taken, in which particular treatment wetland applications are considered as a group. This is done because some technology growth areas are very application specific. It is not necessary to repeat the many details already presented in two key application areas—metals and hydrocarbons—because each has had an entire chapter already (Chapters 11 and 13). Likewise, the general application area of stormwater runoff has been singled out for an entire chapter (Chapter 14). There remain several applications that deserve additional attention because they present unique features that can influence design. These include wetland treatment of animal waste, food wastes, landfill leachate, and chemical industry effluents.

## 25.1 ANIMAL WASTES

### Confined Animal Operations

Liquid animal wastes consist of manure, flushwater, and rainfall. The *Agricultural Waste Management Field Handbook* (USDA-NRCS, 1992) provides information on volumes of manure (feces and urine) and mass excretion rates of nutrients for different types of livestock. Average quantities of nitrogen, phosphorus, and $BOD_5$ in manure are given in Table 25.1. Solid wastes consist of manure and sometimes bedding material such as straw. These amounts are then diluted with water, and the amount of water added to the system usually exceeds the amount of manure. For cattle and dairy facilities in the United States, 160–200 L/d per cow are used for flushing requirements for freestall alleys (CH2M Hill and Payne Engineering, 1997). Swine flushing quantities are typically 60–95 L/d per adult animal. However, Tanner and Kloosterman (1997) use a lower water rate of 50 L/d per cow, and the French use an even lower range, from 5–15 L/d per cow (Liénard *et al.*, 2002).

Operators use a variety of methods to manage wastewater, including lagoons, ponds, storage structures, compost areas, filters, and sediment basins. Liquid wastes are typically collected and treated in one or more lagoons before being land applied. The organic matter and nutrients in manure are important resources that are recycled to the land, if possible, to foster crop production. Solid and slurry wastes usually have very high nutrient concentrations and are good soil amendments, but, in many cases, it is not feasible to collect and contain all animal waste. Some portions are leached from loafing lots, and spills occur. In other cases, the availability of cropped land does not accommodate the entire waste production.

A constructed wetland offers another option for polishing wastewater before it is recycled as flushwater, land applied, or ultimately discharged. As for other wastewater types, the goal is to prevent manure from being discharged to receiving waters. In natural recipients, the organic matter and nutrients promote algal growth and deplete dissolved oxygen, or in other words, promote eutrophication.

Wetland treatment can be implemented on farms in cold climates. Systems have been evaluated across Canada, from Alberta (Reimersma, 2001) to the Atlantic Provinces (Smith, 2003; Smith *et al.*, 2006). Other cold-climate applications have been reported in Germany (Kern and Carlow, 2000; Kern, 2003). These low-technology, solar-driven systems are passive and user-friendly, and thus farmers do not need to acquire the skills of wastewater treatment plant operators. The operation of a wetland system is closely related to current farming practices. The wetland system requires sunlight, nutrients, and water, but does not need to be harvested. The treatment wetland plants can tolerate relatively high concentrations of nutrients, but full-strength wastes should not be applied directly to wetlands because many wetland plants cannot tolerate the high nutrient levels.

The use of wetlands for treating concentrated animal wastes gained a measure of popularity in the early 1990s. The Brenton cattle operation in Iowa, which started in 1930, was the oldest system in North America found by CH2M HILL (1997). Other early sites include Sand Mountain, Alabama, started in 1989, and the Newton and Hattiesburg, Mississippi, systems, also started in 1989. In the mid-1990s, data collection efforts catalogued the performance of 68 wetland systems in North America (DuBowy and Reaves, 1994; CH2M Hill and Payne Engineering, 1997; Knight *et al.*, 2000). Design procedures are offered, which generally fall in the two categories of loading methods and rate constant methods (CH2M Hill and Payne Engineering, 1997).

Examples are included here to illustrate the breadth of applicability for several types of animal operations.

**TABLE 25.1**
**Approximate Waste Quantities for Animals on the Basis of 100 kg Weight**

| | | Nitrogen (g/d·100 kg) | Phosphorus (g/d·100 kg) | $BOD_5$ (g/d·100 kg) |
|---|---|---|---|---|
| **Swine** | Growers | 42 | 15 | 207 |
| | Replacement gilts | 24 | 9 | 108 |
| | Gestating sows | 20 | 7 | 84 |
| | Lactating sows | 46 | 15 | 200 |
| | Boars | 15 | 51 | 66 |
| | Nursery pigs | 59 | 24 | 339 |
| **Dairy cattle** | Lactating cow | 44 | 7 | 161 |
| | Dry cow | 35 | 5 | 119 |
| | Heifer | 31 | 4 | 130 |
| **Beef cattle** | Forage feeder | 31 | 11 | 137 |
| | Energy diet feeder | 29 | 9 | 137 |
| | Yearling | 31 | 10 | 130 |
| | Cow | 33 | 12 | 119 |
| **Poultry** | Laying hens | 84 | 31 | 370 |
| **Human** | 70 kg | 19 | 2.3 | 26 |

*Source:* Adapted from data in CH2M Hill, Payne Engineering (1997) *Constructed wetlands for animal waste treatment*. Report to the Gulf of Mexico Program Nutrient Enrichment Committee, U.S. EPA: Stennis Space Center, Mississippi.

## Poultry

### Example: Auburn Poultry

Three systems of free water surface (FWS) constructed wetlands were prepared at the Alabama Agricultural Experiment Station's (AAES) Poultry Science Unit at Auburn University in Auburn, Alabama, in June 1992 (Rogers *et al.*, 1995; Hill and Payton, 1998). Each system consisted of two cells in series, one planted with *Phragmites* and *Scirpus*, a second with *Sagittaria lancifolia*, while the third series acted as an unvegetated control. Each wetland cell was 5.5 m wide, 30.5 m long with an average depth of 30 cm, and lined with 10 cm of bentonite clay. Subsystems of wooden dowels were tested within the open water system. The wetlands influent was taken from the first cell of the lagoon system fed by the caged layer poultry operation, and the wetland effluent flowed to a holding pond nearby. A hydraulic loading rate of 3.1 cm/d was applied to each system, which resulted in chemical oxygen demand (COD) and total Kjeldahl nitrogen (TKN) loading rates of 145 and 30 kg/ha·d, respectively. In order to determine plant porosity blockage and thus also to assist in calculation of detention time, water column displacement tests were performed. The measured values were 10.7% and 7.0% for *Phragmites* and *Sagittaria*, respectively. The detention time was calculated to range from 11.1 to 11.6 days. The concentration reductions are shown in Table 25.2, and ranged from 40–60% for the principal contaminants for the vegetated systems. It was expected that temperature would have positively

**TABLE 25.2**
**Removal Efficiencies (%) during 1993–94 for the Auburn Poultry FWS Treatment Wetlands**

| | $BOD_5$ | COD | TKN | $NH_4$-N |
|---|---|---|---|---|
| Inlet concentration (mg/L) | 233 | 535 | 135 | 121 |
| *Sagittaria* | 50 | 61 | 43 | 38 |
| *Scirpus* and *Phragmites* | 46 | 62 | 57 | 53 |
| Water | 49 | 54 | 46 | 44 |
| Dowels 10% | 30 | 28 | 20 | 20 |
| Dowels 5% | 35 | 40 | 27 | 26 |

*Source:* Adapted from Rogers *et al*. (1995) In *Versatility of Wetlands in the Agricultural Landscape*. Campbell (Ed.), American Society of Agricultural Engineers, St. Joseph, Michigan; and Hill and Payton (1998) *Transactions of the American Society of Agricultural Engineers* 41(2): 393–396.

affected treatment of several of the parameters under study, but treatment showed little correlation to water temperature (Hill and Payton, 1998).

## Cattle

Wetlands are used to manage feedlot runoff, dairy cattle waters, and pasture waters. Two fairly distinct situations may be found in the use of treatment wetlands for cow operations: pastures and feedlots versus dairies. Pastures and feedlots are primarily event-driven, runoff control systems, whereas dairies are concerned with water originating in milking parlors, comprised of cleaning waters. The two are considered separately here.

Pasture runoff has only been recently considered for treatment. The studies of Tanner and coworkers in New Zealand have utilized modern design concepts and made extensive measurements, which form the beginnings of a performance knowledge base (Tanner *et al.*, 2003; Tanner *et al.*, 2005b). Because some pastures are tile-drained, there is sometimes the possibility of directing tile drainage to constructed wetlands (see Figure 14.8, Chapter 14). The density of animals on the contributing watershed is typically low, and therefore the concentrations of nutrients and other contaminants is relatively low (e.g., 10–20 mg/L nitrogen). Further, speciation of nitrogen favors nitrate as the dissolved form, rather than ammonia, because the infiltration through the vadose zone provides for nitrification.

Feedlots present a different situation because the density on the contributing watershed is high. Nitrogen is almost all in the form of organic and ammonia (TKN), and concentrations of all pollutants are high in the direct runoff. One of the earliest constructed wetlands was a two-cell FWS system built in 1930 at the Brenton Brothers feedlot in Dallas County, Iowa, which serves a 26-ha feedlot that typically has 7,000 head of cattle. The first wetland cell, of extremely irregular shape, comprises 25 ha and receives rain-driven runoff from the feedlot. Over a two-year study period, there were reductions of 87% BOD, 60% TSS, and 58% TKN. This system was one of the two included in the North American Livestock Database (CH2M Hill and Payne Engineering, 1997). Subsequently, other wetland systems have been tested, with good results. For instance, runoff from Iowa State University's 380 head beef cattle concrete feedlot was treated via solids settling, soil infiltration, and a small constructed wetland, in series (Lorimor *et al.*, 2003). Concentrations and nutrient mass flows were reduced by over 97% for TKN, 94% for total phosphorus, and 93% for TSS. No buildups in the system were observed over five years. The authors concluded that the passive treatment system could very effectively protect surface-water quality below open beef feedlots. In their review of vegetated systems for cattle runoff control, Koelsch *et al*. (2006) found that virtually all of the existing information was from non-CAFO (Confined Animal Feeding Operations) situations, i.e., dairies. These authors concluded that critical design factors included pretreatment, sheet flow, discharge control, siting, and sizing, together with maintenance of a dense vegetation stand.

### Example: Cattle

The Canadian livestock farming community is generally not required to meet the strict surface-water discharge regulations imposed on municipalities and industries. However, farming operations generate high levels of nutrients and bacteria that originate from manure storage and application areas, manure storage tank overflows, and feedlot runoff, and there is a need to intercept and reduce these contaminant loads. Demonstration of treatment wetland technology was the prime objective in constructing demonstration wetland systems in Manitoba in 1996. Full-scale wetland projects were constructed at two cattle feedlot operations in the Interlake Region of Manitoba (Pries and McGarry, 2001). The second constructed wetland, described here, was at an 1,800-head feedlot bordering Lake Manitoba in the Interlake area, commissioned in 1998. The treatment wetland was constructed to reduce the nutrient and solids loading to Lake Manitoba.

The project captures the feedlot runoff, stores it in a holding pond, and then treats the wastewater in constructed wetland cells operated during approximately 150 days each year from mid-May through mid-October. The 0.25-ha holding pond collects and stores contaminated runoff from the feedlot area during the late fall, winter, and early spring, and has a working volume of approximately 3,300 $m^3$. The average hydraulic retention time (HRT) in the pond during the wetland operating period is approximately two months. The stored water is pumped to the wetland cells; the topography does not allow for gravity flow.

The treatment wetland consists of two 0.5-ha cells operating in parallel with an average depth of 0.3 m and a working volume of 3,000 $m^3$. Flow to each of the wetland cells is controlled by gate valves. For normal operation, the gate valves at the weir box are opened to allow a maximum flow of about 100 L/min to each cell. The flow enters the wetland in the center of the width of an influent distribution deep zone, which distributes the water across the width of each cell. Treated water is discharged at the effluent level control structure with a weir for flow measurement. In dry years there may be no outlet discharge due to water removal through high rates of evapotranspiration.

The holding pond provided considerable treatment prior to discharging the contaminated stormwater to the wetland. Table 25.3 presents the average annual data from monthly samples for two years of operation. The wetland effluent $BOD_5$ was 15 mg/L or less throughout both years, and the TSS was also consistently less than 15 mg/L. Inflow ammonia averaged 2.1 mg/L, with excursions to 4.8 mg/L, but the effluent ammonia-nitrogen was less than 0.5 mg/L. Phosphorus reduction was consistent each year with lower concentrations in the effluent than was measured in the influent.

## Dairy Operations

Dairy operations can be major contributors of nutrients to the watersheds in which they are located. For instance, in a nutrition study, it was found that cows excreted 88.2% of phosphorus

**TABLE 25.3**
**Manitoba Interlake Site 2: Average Annual Concentrations in the Wetland Inflows and Outflows**

| | Average Value 1999 | | Average Value 2000 | |
|---|---|---|---|---|
| **Parameter** | **Inflow** | **Outflow** | **Inflow** | **Outflow** |
| $BOD_5$ (mg/L) | 20 | 4 | 27 | 7 |
| $NH_3$-N (mg/L) | 2.1 | 0.4 | 1.4 | 0.1 |
| TKN (mg/L) | 25 | 16 | 32 | 15 |
| TP (mg/L) | 6 | 4.3 | 8.5 | 5 |
| TSS (mg/L) | 21 | 6 | 73 | 31 |
| pH | 8.1 | 8.1 | 8.3 | 8.2 |
| Conductivity (μs/cm) | 3,377 | 3,139 | 3,629 | 29,54 |
| COD (mg/L) | 247 | 280 | 303 | 222 |
| Fecal coliform (col/100 mL) | 420 | 201 | 1,082 | 324 |
| Total coliform (col/100 mL) | 11,848 | 1,496 | 14,044 | 25,58 |
| DO (mg/L) | 1.2 | 2.2 | 3.1 | 4.8 |

*Source:* From Pries and McGarry (2001) *Constructed Wetlands for Feedlot Runoff Treatment in Manitoba*. Proceedings of the 2001 Water Environment Association of Ontario Technical Conference; Toronto, Ontario. Reprinted with permission.

consumed daily: 68.6% in feces, 1.0% in urine, and 30.3% in milk (Morse *et al.*, 1992). Current and abandoned dairy systems in South Florida's Lake Okeechobee watershed have a demonstrated phosphorus pollution potential (Nair *et al.*, 1995). In Ireland, about 25% of the land area is devoted to dairy farming, and it is estimated that the percentage of pollution attributed to agriculture is approximately 32% in rivers and streams that are slightly to moderately polluted.

Most dairies produce two types of waste streams: "white" water, which is defined as the wastewater produced during cleaning and sterilization of the milking equipment, and "green" water, which is wastewater resulting from the washdown of the manure-spattered walls and floors of the milking parlor and the associated holding pen or staging areas. The combination of these two waste streams produces the "typical" milkhouse wastewater. Because each dairy is different, the ratio of white water to green water will vary. Some farms may have existing manure treatment lagoons, which may be used as pretreatment device for the wetland treatment system. A typical dairy waste treatment schematic is shown in Figure 25.1.

Constructed wetlands have been demonstrated to be effective add-ons to supplement the treatment afforded by farm dairy pond systems. Farm dairy ponds provide for the return of a significant proportion of the nutrient loading to the pasture via periodic desludging, but the water discharges are not sufficiently good to protect receiving water quality (Table 25.4; see also Table 25.3). This application has spurred considerable research over the past decade (Geary and Moore, 1999; Karpiscak *et al*., 1999; Newman *et al.*, 2000; Hunt and Poach, 2001; Shamir *et al.*, 2001; Kowalk, 2002; Ibekwe *et al.*, 2003; Forbes *et al.*, 2004a; Dunne *et al.*, 2005c; Smith *et al.*, 2006; Healy *et al.*, 2007). Cronk (1996) identified 40 dairy treatment wetlands in North America, established between 1989 and 1993. A database compilation in 1996 identified 60 constructed wetlands at 38 sites in North America (CH2M Hill and Payne Engineering, 1997; Knight *et al.*, 2000). However, a large number of systems have been built, starting in the late 1990s. For instance, by 2004 there were 12 FWS wetlands in the Anne valley of County Wexford in Ireland (Figure 25.2) (Carroll *et al.*, 2005). In general, these systems have large surface areas, averaging 120 $m^2$/cow, and provide excellent water quality improvement (Table 25.4).

Although dairy wetlands have primarily been FWS, horizontal subsurface flow have also been implemented (Tanner *et al.*, 1995a; Tanner *et al.*, 1995b; Mantovi *et al.*, 2003; Kern, 2003). Tanner and Kloosterman (1997) recommend HSSF as an add-on to FWS, presumably because of the potential for clogging of a gravel bed with direct lagoon discharges. The New Zealand wetland design guidelines suggest about 1.0 $m^2$/cow for this post-FWS application (Tanner and Kloosterman, 1997).

However, if a HSSF system is to be used as a "standalone" device, more area is required to achieve good pollutant reductions. In the United Kingdom, a HSSF wetland was used to treat influent with an average $BOD_5$ of 1,192 mg/L on a dairy farm in Drointon (Cooper *et al.*, 1996). An area of 250 $m^2$ for 160 cows (1.6 $m^2$/cow) proved unacceptable, and vertical flow wetlands were subsequently installed ahead of the HSSF system. A 95% reduction in $BOD_5$ was achieved (Figure 25.3). In Germany, an area of about 10 $m^2$/cow produced reductions of 91% COD, 86% total nitrogen (TN), and 99% fecal coliforms (FC), with better nutrient reductions in winter (Kern, 2003). A HSSF system in Italy dimensioned at 1.84 $m^2$/cow (at a measured 80 L/d per cow) was successful in reducing BOD from 451 mg/L to 28 mg/L, for a settled influent from Imhoff tank pretreatment (Mantovi *et al.*, 2003). Total nitrogen was reduced from 65 to 33 mg/L, due to mineralization of organic nitrogen, because ammonia removal was nonexistent. It should be noted that European dairy wetlands

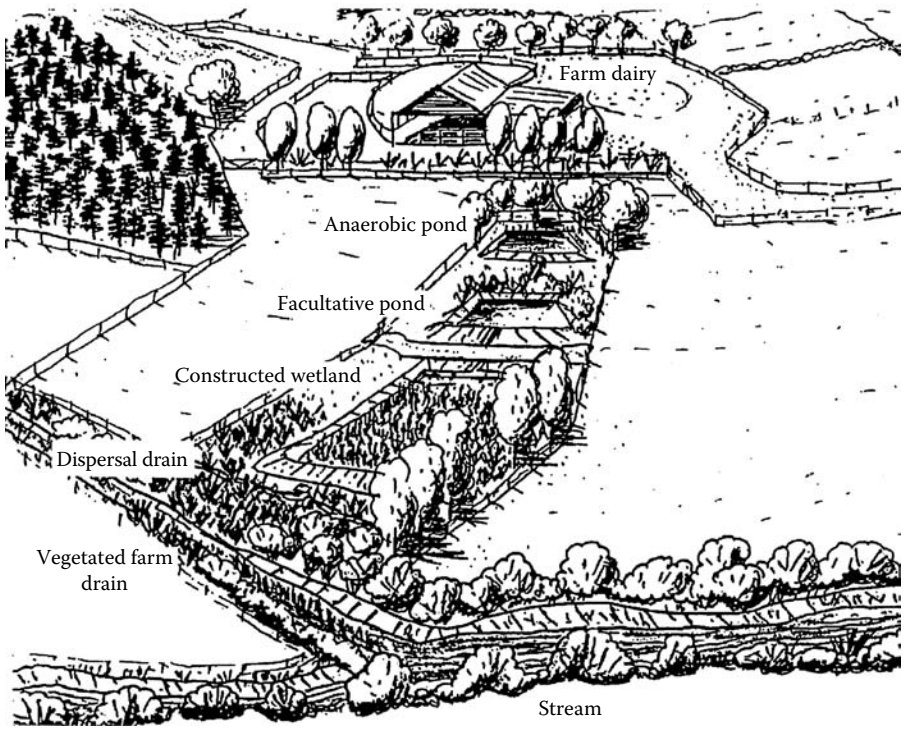



**FIGURE 25.1** Concept for a farm dairy treatment wetland system. (From Tanner and Kloosterman (1997) *Guidelines for Constructed Wetland Treatment of Farm Dairy Wastewaters in New Zealand*. NIWA Science and Technology Series No. 48, National Institute of Water and Atmospheric Research (NIWA): Hamilton, New Zealand. Reprinted with permission.)

do not usually have a lagoon for pretreatment, although that is the norm in North America and New Zealand. Therefore, direct comparisons are difficult.

Research is being directed to more highly oxygenated systems, such as forced aeration or vertical flow wetlands (Whitney, 2003). Vertical flow wetlands have high rates of oxygen transfer and are capable of handling high oxygen demands of milk-house wastes (Green *et al.*, 2002). Recommended vertical flow wetland criteria for dairy waters in France total 0.22–0.44 $m^2$/cow in two stages in each of two

**TABLE 25.4**
**Constructed Wetlands for Farm Runoff in the Anne Valley near Waterford, Ireland**

| | | | | $BOD_5$ | | TSS | | $NH_4$-N | |
|---|---|---|---|---|---|---|---|---|---|
| **Runoff Type** | **Area ($m^2$)** | **Cows** | **$m^2$/Cow** | **In (mg/L)** | **Out (mg/L)** | **In (mg/L)** | **Out (mg/L)** | **In (mg/L)** | **Out (mg/L)** |
| Dairy | 6,081 | 60 | 101 | 7,194 | 12 | 1,141 | 13 | 184 | 0.3 |
| Dairy | 22,025 | 60 | 367 | 510 | 31 | 532 | 24 | 64 | 0.5 |
| Dairy, beef | 10,290 | 50 | 206 | 447 | 26 | 136 | 20 | 56 | 0.8 |
| Dairy | 10,340 | 100 | 103 | 521 | 34 | 1,119 | 52 | 119 | 3.1 |
| Dairy, tillage | 3,950 | 35 | 113 | 313 | 19 | 136 | 14 | 51 | 0.6 |
| Dairy | 12,710 | 80 | 159 | 213 | 19 | 210 | 22 | 39 | 0.3 |
| Beef | 3,942 | — | — | 59 | 3 | 43 | 9 | 22 | 0.2 |
| Mixed | 7,958 | 55 | 145 | 494 | 12 | 459 | 15 | 41 | 0.6 |
| Mixed | 4,380 | 50 | 88 | 171 | 14 | 165 | 20 | 40 | 0.4 |
| Dairy | 7,690 | 77 | 100 | 694 | 21 | 693 | 18 | 40 | 0.4 |
| Mixed | 10,726 | 85 | 126 | 2,245 | 17 | 3,089 | 24 | 266 | 0.7 |
| Sheep, tillage | 3,621 | — | — | 50 | 16 | 200 | 20 | 13 | 0.1 |
| **Median** | | | **120** | **471** | **18** | **335** | **20** | **46.0** | **0.5** |

*Source:* Data from Carroll *et al*. (2005) In *Nutrient Management in Agricultural Watersheds: A Wetland Solution*. Dunne *et al*. (Eds.), Wageningen Academic Publishers, Wageningen, The Netherlands.



**FIGURE 25.2** Farm wetland in the Anne valley near Waterford, Ireland. This FWS system takes advantage of a downhill location from the barns and yards to employ gravity flow.

trains (Liénard *et al.*, 2002). As for other VF systems, dairy VF wetlands are pulse loaded to induce hydrodynamic movement of air through the wetland bed.

The practitioner can find several design guidelines documents (CH2M Hill and Payne Engineering, 1997; Tanner and Kloosterman, 1997; Forbes *et al.*, 2004a). However, the supporting data is in the process of being strengthened at this time, as the various authors note, and improvements are to be expected as more operating data become available. Rate constants for dairy FWS can be found in the literature, in Payne Engineering and CH2M HILL (1997). In general, these are within the distributions presented in earlier chapters of this book, and indeed the dairy data were included in those determinations. McGechan *et al.* (2005) give rate constants for HSSF and VF dairy systems for nitrogen processing for mesocosms for dilute farm influents. However, loading specifications are more typically used to interpret data and for design. As an example, one set of median-loading recommendations for FWS systems is shown in Table 25.5. It should be noted that those loadings correspond to fairly large wetland areas, comparable to those used in the Irish systems in Table 25.4. Those dairy loading recommendations are comparable to, but somewhat less than, those recommended by U.S. EPA for municipal wastewaters.

Nitrogen reductions are in general less than for BOD and TSS. Dairy wastewaters differ from most municipal and domestic wastewaters in that they contain a relatively high amount of organic nitrogen. For instance, the database used in this book shows a median inlet organic nitrogen of 37 mg/L for animal treatment wetlands, whereas the median

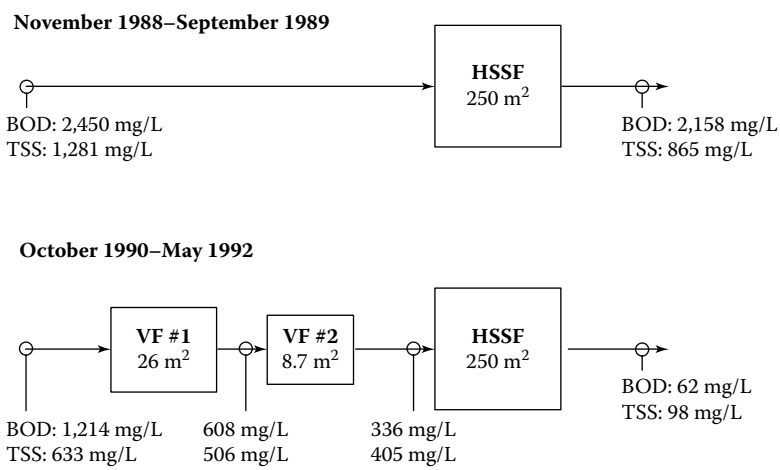



**FIGURE 25.3** Performance of the first and second iterations of the Drointon, U.K., reedbeds for farm waste treatment. In the first case, the BOD loading to the HSSF wetland was 255 kg/ha·d, and in the second case it was 35 kg/ha·d. (Based on Cooper *et al*. (1996) *Reed Beds and Constructed Wetlands for Wastewater Treatment*. WRc Publications, Swindon, United Kingdom.)

**TABLE 25.5**
**Median-Loading Recommendations for FWS Animal Wastewater Wetlands (kg/ha·d)**

| Effluent Target | 20 mg/L | 30 mg/L | 40 mg/L | 100 mg/L |
|---|---|---|---|---|
| $BOD_5$ | 20 | 33 | 51 | 100 |
| TSS | 15 | 30 | 61 | 135 |
| TN | 8 | 10 | 27 | 61 |
| TP | 11 | 18 | 28 | 56 |
| U.S. EPA $BOD_5$ | 45 | 40–60 | — | — |

*Source:* Adapted from data in CH2M Hill, Payne Engineering (1997) *Constructed wetlands for animal waste treatment*. Report to the Gulf of Mexico Program Nutrient Enrichment Committee, U.S. EPA: Stennis Space Center, Mississippi; and U.S. EPA (2000b) *Constructed wetlands: Agricultural wastewater*. EPA 843/F-00/002, U.S. EPA Office of Water.

for municipal systems is 7 mg/L. As a consequence, there is a need to mineralize this organic nitrogen before it can be reduced by oxidative and other processes. This additional processing step slows overall nitrogen removal and is partly to blame for low removal rates. Mineralization of the organic nitrogen can increase the ammonia content of the water prior to its nitrification. Ultimately, the loss of total nitrogen suggests an implied oxygen supply that is limiting for nitrogen processing. The internal ammonia load created by mineralization of organic nitrogen adds to the incoming ammonia. The rates of nitrogen removal in animal wetlands are strongly suggestive of mechanisms other than traditional nitrification–denitrification. Oxidized nitrogen is rarely a significant component of influents or effluents from dairy water wetlands. This is speculatively due to an adequate carbon supply and rapid denitrification, and perhaps also due to Anammox or similar alternative processing mechanisms (see Chapter 9).

### Example: Dairy Farm Operations

A study on a three-cell integrated FWS system in County Wexford, Ireland, was conducted over a 2.5-year period (Dunne *et al.*, 2005a; Dunne *et al.*, 2005b). Water from farmyard management and dairy operations for a 42-cow facility were collected in a central storage tank before discharge to the wetlands. The three cells had a combined area of 4,265 $m^2$ (102 $m^2$/cow) and were maintained at a 30–40-cm depth. A 250-$m^2$ sedimentation basin preceded the treatment marshes. The vegetative cover was 80–90% during the growing season, and comprised of a mixture of *Carex riparia, Typha latifolia, Phragmites australis, Sparganium erectum*, and other emergent species. The flow rate to the wetlands averaged 7.6 $m^3$/d (180 L/cow·d), and was greatly augmented by rainfall (1,142 mm/yr) to produce an outflow rate of 31.1 $m^3$/d. As a result, there were considerable differences between mass reductions and concentration reductions (Table 25.6). Part of the observed concentration reduction was attributable to dilution. Nonetheless, this lightly loaded system produced very good removals, but with lesser performance in winter for ammonia and phosphorus. It is noteworthy that the implied maximum oxygen supply was about 4.8 gO/$m^2$·d, which is a feasible reaeration rate. It is also the case that the nitrogen supply places these wetlands in the group with agronomic control. BOD and TSS are reduced to what

**TABLE 25.6**
**Summary Performance of the Teagasc FWS Dairy Wetlands, Located at the Teagasc Research Center, Johnstown Castle, Wexford, Ireland**

| | Flow ($m^3$/d) | SRP (mg/L) | $NH_4$-N (mg/L) | $BOD_5$ (mg/L) | TSS (mg/L) |
|---|---|---|---|---|---|
| Inlet | 7.6 | 19 | 48 | 2,629 | 980 |
| Outlet | 31.1 | 1.7 | 1.9 | 20 | 11 |
| **Concentration reduction** | | **91%** | **96%** | **99%** | **99%** |
| | | **SRP ($g/m^2$·yr)** | **$NH_4$-N ($g/m^2$·yr)** | **$BOD_5$ (kg/ha·d)** | **TSS (kg/ha·d)** |
| Inlet | | 12.0 | 31.2 | 46.8 | 17.5 |
| Outlet | | 4.5 | 5.1 | 1.5 | 0.8 |
| **Mass removal** | | **62%** | **84%** | **97%** | **95%** |

*Source:* Data from Dunne *et al.* (2005c) *Ecological Engineering* 24(3): 221–234.

appear to be background concentrations because there was little change in concentrations after the first two wetlands.

### Swine

Swine wastewaters are strong effluents that received only minimal treatment in past years. A very popular system in the southeastern United States is an anaerobic lagoon followed by a spray irrigation system. In North Carolina, there are 4,500 active and 1,700 inactive swine waste lagoons (Humenik *et al.*, 2004), but these are no longer permitted because of problems with overflow and leakage. Constructed wetlands are a logical add-on, and therefore investigations on their effectiveness were initiated, approximately in 1990, at Auburn University's Sand Mountain Agricultural Experiment Station in DeKalb County, Alabama (Hammer *et al.*, 1993a; McCaskey *et al.*, 1994). Another facility followed soon after (1991) at the Pontotoc, Mississippi, Experiment Station (Cathcart *et al.*, 1994). Based on early swine wetland data and the performance of municipal treatment wetlands, design recommendations quickly proliferated (USDA-NRCS, 1992; Hammer, 1994). Loading numbers of approximately 60–70 kg/ha·d for BOD were suggested. In fact, FWS swine wetlands are at the poor performance edge of the municipal concentration-loading band (Figure 25.4), and this loading now appears likely to produce 60–80 mg/L BOD in the wetland outflow. This may be due to the much higher concentrations in swine wastewater (Cronk, 1996). Research intensified in the mid-1990s, with projects in North Carolina and Indiana. Investigations were conducted from 1992 to the present at a continuous marsh site in the eastern coastal plain in Duplin County, and at a marsh–pond–marsh site in Greensboro at North Carolina A&T State University. The site descriptions and operational procedures are reported in a number of publications (Hunt *et al.*, 1994; Reddy *et al.*, 2001; Stone *et al.*, 2002; Poach *et al.*, 2003; Stone *et al.*, 2004). The Purdue University system at West Lafayette involved 16 FWS wetland cells operated in a designed experimental array (Reaves *et al.*, 1994). These studies all showed significant reductions in BOD, TSS, and total nitrogen, but the removal percentages were not outstanding (Table 25.7).

Because of the high ammonia levels in swine lagoons and wetlands, there has been concern that ammonia volatilization could be a problem. A study was conducted to address the concern that nitrogen removal was due to high ammonia volatilization. Poach *et al*. (2002; 2004) documented that less than 10% of the input nitrogen was lost by ammonia volatilization (see Figure 9.7, Chapter 9). However, as for other animal wastes, the organic nitrogen content is high, and a mineralization step must precede ammonia conversions. The process of denitrification is greatly assisted by the availability of organics and carbon, with the result that oxidized nitrogen is not present in the waters, and by the fact that redox potentials are generally quite low (Hunt *et al*., 2003; Szögi *et al.*, 2004). Nitrogen processing in swine wetlands is quite temperature dependent, as for other FWS systems. However, that does not preclude their use in cold climates during the unfrozen season, even as far north as Alaska (Maddux, 2002).

Subsurface flow wetlands have also been used to treat swine effluents in North America (Sievers, 1997) and elsewhere around the world (Lee *et al.*, 2004b). However, the high TKN concentrations also cause problems with respect to oxygen supply in HSSF systems. Therefore, vertical flow systems have also been piloted (Kantawanichkul *et al.*, 1999; Sezerino *et al.*, 2003). As an alternative to vertical flow, a fill-and-drain mode has been employed with good success (Behrends *et al.*, 2003; Rice *et al.*, 2005). This concept involves continuously pumping wastewater back and forth

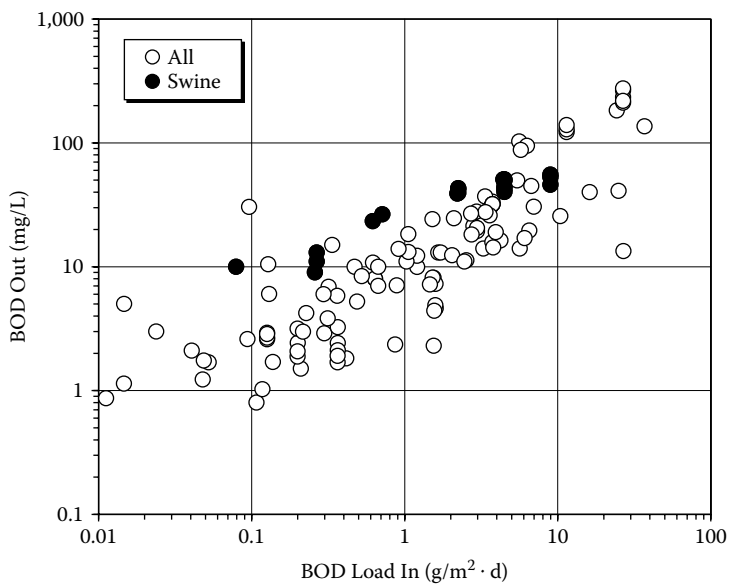



**FIGURE 25.4** The position of swine treatment wetlands compared to all other FWS systems.

**TABLE 25.7**
**Examples of Performance of FWS Wetlands for Swine Wastewater**

| | | TSS | | | BOD | | | TN | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Site | HLR (cm/d) | Inlet (mg/L) | Load (kg/ha·d) | Outlet (mg/L) | Inlet (mg/L) | Load (kg/ha·d) | Outlet (mg/L) | Inlet (mg/L) | Load (kg/ha·d) | Outlet (mg/L) |
| Greensboro, North Carolina | 1.1 | 522 | 57 | 164 | 821 | 90 | 383 | 146 | 16 | 71 |
| Greensboro, North Carolina | 2.2 | 522 | 114 | 175 | 821 | 180 | 471 | 146 | 32 | 93 |
| Pontotoc, Alabama | 1.54 | 93 | 14 | 35 | 47 | 7 | 27 | 130 | 20 | 40 |
| Pontotoc, Alabama | 1.27 | 96 | 12 | 31 | 49 | 6 | 23 | 129 | 16 | 47 |
| Purdue, Indiana | 1.96 | 137 | 27 | 52 | 114 | 22 | 41 | 398 | 78 | 212 |
| Purdue, Indiana | 3.91 | 136 | 53 | 70 | 114 | 45 | 47 | 407 | 159 | 233 |
| Purdue, Indiana | 7.83 | 134 | 105 | 82 | 115 | 90 | 51 | 417 | 326 | 265 |

*Note:* The Greensboro values are COD, not BOD.

between adjacent cells on a two-hour cycle. This vertical flow design provides aeration of the gravel substrate and exposes the internal biofilms to atmospheric oxygen. During the "drain phase" of the cycle, atmospheric oxygen causes enhanced oxidation of ammonia and organic matter. This mode of operation is described further in Chapter 24.

### Example: Swine

In 1995, six wetland systems to treat swine lagoon wastewater were constructed at the North Carolina A&T State University farm near Greensboro, North Carolina (Reddy *et al.*, 2001; Stone *et al.*, 2004). The wetland systems were configured into a marsh section, a central pond section, and another marsh section (marsh–pond–marsh). The marsh sections were approximately 10 m × 10 m, and the pond section was 10 m × 20 m. The marsh sections were planted with *Typha latifolia* and *Schoenoplectus americanus* in March 1996. Beginning in September 2000, the wetlands were loaded with specific total nitrogen at rates of 5, 14, 23, 32, 41, and 50 kg/ha·d. For one year, the wetland hydraulic loading rates were held approximately equal, with only the TN loading rate varying. The operating depths of the constructed wetlands were 15 cm for marsh sections and 75 cm for pond sections.

Mean TN and $NH_4$-N concentration reductions were 35 and 25%, respectively. However, the nitrogen performance was on the poor side of the scatter of performances for other FWS wetlands (Figure 25.5). The calculated first-order plug-flow rate constants ($k_{20}$) for TN and $NH_4$-N were 3.7–4.5 m/d and 4.2–4.5 m/d, respectively, which are considerably lower than the central tendencies given in Chapter 9. These are at about the 10th percentile for FWS wetlands. However, no allowance was made for the internal mineralization of organic nitrogen in the Stone *et al*. (2004) analysis.

## Zoos

The Wuhan Zoo, Wuhan City, China, is investigating the use of sedimentation systems and wetlands to reduce nutrient losses from the site to a lake. The project is conducted by the Research Centre for Eco-Environmental Sciences, Chinese Academy of Sciences, Beijing (D'Arcy, 2005). A reed bed has been operating for some years at Chester Zoo in the U.K., treating runoff from its Asian elephant enclosure (D'Arcy, 2005). A vertical flow treatment wetland is used at the Saint-Felicien zoo in Quebec (Pries, 1994). The seals of the Rheine zoo in Westphalia, Germany, have their water recycled back to their basin to cut costs that otherwise would arise by replacing used water with fresh drinking water (Dartmann *et al.*, 2000). The seals produce large amounts of residue that cloud the water, capable of impairing the visitors' view, especially in deep-water zones. A horizontal flow wetland system is used to filter and clarify the water.

## Aquaculture

Fish farming is an increasing activity worldwide, as wild supplies dwindle and dietary preferences change. Constructed wetlands have been explored as a means of controlling the associated pollution, notably for catfish, trout, and shrimp operations. However, other fish species have also been the object of treatment wetlands, such as milkfish (*Chanos chanos*) (Lin *et al.*, 2002a; Lin *et al.*, 2002b).

### Catfish

Channel catfish (*Ictalurus punctatus*) farming is the largest segment of aquaculture in the United States. Most of the nation's catfish is raised in Alabama, Arkansas, Louisiana, and Mississippi on 70,000 ha of catfish farms (Tucker, 1999). More farm-raised catfish is produced and sold in these states than all other U.S. aquacultured species combined. In fact, the farm-raised catfish industry enjoyed phenomenal growth over the past 37 years, from 2.6 million kilograms produced in 1970 to more than 275 million kilograms in 2006 (Harvey, 2006). Nearly all commercial aquaculture in the southeastern United States is conducted in earthen ponds. Good production from ponds is encouraged by using manufactured feeds

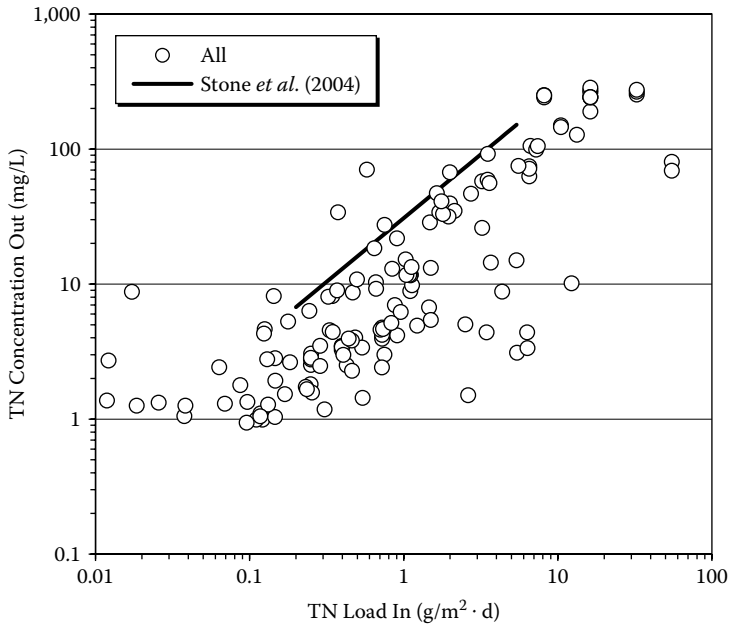



**FIGURE 25.5** Total nitrogen loading graph for the marsh–pond–marsh system at North Carolina A&T University Farm FWS swine wetlands at Greensboro, North Carolina, compared to the general FWS loading data. The line that represents their swine data (from Stone *et al.* (2004) *Ecological Engineering* 23(2): 127–133) is on the upper edge of the cluster of data.

or increasing the availability of natural foods by adding fertilizers. Less than 30% of the nitrogen and phosphorus added in feed or fertilizer is recovered at harvest. The remainder of the nutrient load is left in the pond and may be discharged when it rains or when ponds are drained between fish crops.

In the United States, aquaculture effluents are regulated under the National Pollutant Discharge Elimination System (NPDES) as part of the Federal Water Pollution Control Act of 1972. A study by the Southern Regional Aquaculture Center characterized catfish pond effluents from 45 ponds over a two-year period (Tucker, 1999). Effluent quality was poorest in the summer when fish-feeding rates and water temperatures are highest. Catfish pond effluents generally have higher concentrations of nutrients and organic matter than natural stream waters, but much lower concentrations than municipal and industrial wastewater.

As a result of successful demonstrations, the use of constructed wetlands for finfish production was thoroughly examined by Posadas and LaSalle (1997; 1998). Their report concluded that the use of constructed wetlands was not economically viable because of the added costs. Kouka and Engle (1994) estimated that wetland treatment would add $0.18/kg to the price of the fish, which is on the order of $1.75/kg (2006 USD) (Harvey, 2006).

### *Example: Catfish*

In 1992, a two-cell, 2,350-m$^2$ test wetland was implemented near Greensboro, Alabama (Schwarz and Boyd, 1995). Water from a production pond was passed through the constructed wetlands at HRTs of 1, 2, 3, and 4 days, with hydraulic loading rates of 77–91 L/d·m$^2$. Cell one was planted with California bulrush (*Scirpus californicus*) and giant cutgrass (*Zizaniopsis miliacea*), and cell two with maidencane (*Panicum hemitomon*). Concentrations of potential pollutants were much lower in effluent from the wetland than in influent from the channel catfish ponds. The mean mass removals were total nitrogen, 61%; biochemical oxygen demand, 62%; suspended solids, 87%; and total phosphorus, 75%. Overall performance of the wetland was not correlated with HRT, but that was clearly because hydraulic loading was held constant while depth was varied. Thus, this data supports the notion that removal is area specific rather than volume specific. The researchers observed that although pond overflow and water exchanges could be treated with reasonably sized wetlands, complete pond draining usually proceeds rather rapidly and would create unacceptably high hydraulic loads.

## Tilapia

"Tilapia" is the generic name of a group of cichlids endemic to Africa, consisting of three aquaculturally important genera (*Oreochromis, Sarotherodon,* and *Tilapia*). Worldwide harvest of farmed tilapia has now surpassed 800,000 metric tons, and tilapia are second only to carp as the most widely farmed freshwater fish in the world (Pompa and Masser, 1999). Positive aquacultural characteristics of tilapia are their tolerance to poor water quality and the fact that they eat a wide range of natural food organisms. Indeed, tilapia have been considered as part of a system to improve swine water quality, used in conjunction with wetlands (Dontje and Clanton, 1999). Biological constraints to the development of commercial tilapia farming are their inability to withstand sustained water temperatures below 10 to 15°C. The U.S. imports most of its

**TABLE 25.8**
**Examples of Performance of HSSF Wetlands for the Improvement of Salmonid Fish Farm Wastes**

| | | | | TSS | | BOD | | TN | | TP | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Site | Reference | Time | HRT (d) | Inlet (mg/L) | Outlet (mg/L) | Inlet (mg/L) | Outlet (mg/L) | Inlet (mg/L) | Outlet (mg/L) | Inlet (mg/L) | Outlet (mg/L) |
| Chisholm, Minnesota | Axler *et al*. (1996) | Autumn 1994 | 9.70 | 41.6 | 2.0 | 30.6 | 3.7 | 7.32 | 1.56 | 2.28 | 0.04 |
| Chisholm, Minnesota | Axler *et al*. (1996) | Summer 1995 | 10.80 | 18.6 | 1.6 | 13.4 | 1.0 | 4.43 | 1.42 | 1.04 | 0.05 |
| Chisholm, Minnesota | Axler *et al*. (1996) | Autumn 1995 | 5.60 | 15.1 | 1.8 | 10.1 | 0.9 | 31.60 | 5.91 | 13.30 | 0.07 |
| Dungeness, Washington | Michael (2003) | — | 5.80 | 58.0 | 5.0 | 18.9 | 3.5 | 0.43 | 0.11 | 0.76 | 0.13 |
| Brandenburg, Germany | Schulz *et al*. (2003a) | — | 0.15 | 9.9 | 3.2 | 17.6 | 12.1 | 1.18 | 0.96 | 0.19 | 0.11 |
| Brandenburg, Germany | Schulz *et al*. (2003a) | — | 0.23 | 9.9 | 3.1 | 17.6 | 12.0 | 1.18 | 0.89 | 0.19 | 0.09 |
| Brandenburg, Germany | Schulz *et al*. (2003a) | — | 0.46 | 9.9 | 2.5 | 17.6 | 12.3 | 1.18 | 0.81 | 0.19 | 0.09 |

*Note:* COD for Germany, and $NH_4$-N for Washington.

demand, but tilapia farms are beginning to appear, and with them the need to improve water quality.

In the mid-1990s, constructed wetlands were tested for treatment of tilapia wastewater at New Mexico State University, Las Cruces, New Mexico (Zachritz and Jacquez, 1993). Brown *et al*. (1999) used salt-tolerant plants (*Suaeda esteroa, Salicornia bigelovii,* and saltbush *Atriplex barclayana*) to remove nutrients from saline aquaculture wastewater in Tucson, Arizona. These were grown in sand in draining lysimeters and were irrigated to meet evapotranspiration and produce a 0.3 leaching fraction using aquaculture effluent generated from an intensive tilapia culture system. The system removed 98% and 94% of total and inorganic nitrogen, respectively. Behrends *et al*. (1999; 2002) utilized fill-and-drain SSF wetlands to treat wastewater from an experimental tilapia-rearing (*Oreochromis niloticus*) facility in Muscle Shoals, Alabama, with good success.

## Salmonids

Salmon and trout (family *Salmonidae*) are also farmed intensively in more northerly climates and have the same general water quality issues as catfish. In the Province of Quebec, for example, trout farm production increased from 40 tons/yr in 1976 to 1,900 tons/yr in 1996 (Comeau *et al.*, 2001). Trout farms effluents are typically 20 times more diluted than medium-strength municipal wastewaters and even below municipal secondary treatment criteria (Comeau *et al.*, 2001; Schulz *et al.*, 2003a). With respect to receiving-water-quality objectives, the most constraining element to remove from freshwater fish farms is phosphorus.

Evaluation of treatment wetlands to treat effluents began in the mid-1990s, with work at the Freshwater Institute in West Virginia (Summerfelt *et al*., 1999), and at Chisholm, Minnesota (Axler *et al.*, 1996). Subsequently, projects proliferated, involving both FWS systems (Michael, 2003) and HSSF systems (Comeau *et al.*, 2001; Naylor *et al.*, 2003; Schulz *et al.*, 2003a; Ouellet-Plamondon *et al.*, 2004; Schulz *et al.*, 2004; Lymbery *et al.*, 2006). In general, findings indicate that constructed wetlands are an ecologically attractive and economical method for treating salmon and trout farm effluents to reduce solids and phosphorus. Examples of wetland performance are given in Table 25.8.

## Shrimp

The production of crustaceans in pond culture is a major agricultural enterprise. In the United States, this is a fledgling industry, with 120 farms growing only about 3,800 metric tons of saltwater shrimp and freshwater prawns (USDA-NASS, 2006). In contrast, 647 farms grow 16,000 metric tons of crawfish. U.S. imports of shrimp totaled 560,000 metric tons in 2006, because shrimp farming is primarily a tropical activity, with major centers in China, India, Thailand, Indonesia, and South America (Harvey, 2006). Effluent waters from shrimp aquaculture can contain elevated levels of phosphorus, ammonia, nitrate, and organics. Regulation of the water quality of shrimp pond discharges has begun in many locations. Constructed wetlands offer an ecologically beneficial, low-cost treatment alternative, and, as a result, exploratory studies began in the mid-1990s (Sansanayuth *et al.*, 1996). It was observed by Gautier *et al.* (2001) that an engineered mangrove wetland produced good water-quality improvement for a shrimp farm effluent in Colombia. However, the use of natural wetlands is not always allowed, and thus research mostly focuses on constructed wetlands.

A 7.7-ha mesohaline (3–8 ppt) FWS-constructed wetland was used to treat 13,600 $m^3$/d of effluent from 8.1 ha of intensively farmed shrimp ponds near Port Mansfield, Texas, on the coast of the Gulf of Mexico (Badrinarayanan, 2001; Tilley *et al.*, 2002; Staff, 2006) The depth of water was maintained between 0.15 and 0.45 m, which provided a hydraulic retention time (HRT) in the wetland on the order of one day. Originally, the wetland was planted with ten species of salt-tolerant wetland plants, but by the second year of operation, the wetland had undergone self design. *Typha latifolia*

became the overwhelmingly dominant plant species, occupying nearly two thirds of the wetland area. Concentrations of total phosphorus (TP), total suspended solids (TSS) and inorganic suspended solids (ISS) were reduced by 31, 65, and 76%, respectively, during recirculation, and maintained satisfactory water quality. Parameterization of the *k–C** model for TSS or TP showed that mean target levels could be achieved when the ratio of pond surface to wetland surface was 12:1. Tilley *et al*. (2002) concluded that constructed wetlands can perform satisfactorily as recirculation filters in large-scale shrimp farms.

Similar results were obtained in a combination system in Taiwan (Lin *et al.*, 2003b; 2005). A FWS unit was followed by a HSSF unit, and integrated into a commercial-scale recirculating aquaculture system for intensive shrimp culture. This study investigated performance of the treatment wetlands for controlling water quality. Removals were TSS (55–66%), $BOD_5$ (37–54%), total ammonia (64–66%), and nitrite (83–94%) for recirculating water under high hydraulic loading rates (1.57–1.95 m/d). The area ratio of the demonstration wetlands to culture tank was 0.43. It was concluded that a constructed wetland was technically and economically feasible for managing water quality of an intensive aquaculture system.

## 25.2 FOOD-PROCESSING WASTEWATERS

### Sugar Refining

Wastewaters from sugar-processing plants have been treated in natural wetlands (Gambrell *et al*., 1987). The form of the BOD in these wastewaters is sucrose and its precursors and derivatives. It is therefore not surprising that removal follows the pattern established for BOD in municipal wastewaters. The *k-C** model does a good job of fitting the batch disappearance of soluble TOC (Figure 25.6). The rate constant is similar to that obtained from BOD data from numerous domestic wastewater wetlands (see Chapter 8).

American Crystal Sugar's Hillsboro, North Dakota, sugar beet refinery, built in 1973, generates about 260,000 $m^3$ of wastewater during the eight-month processing season. The plant uses a 64-ha FWS treatment wetland for advanced secondary since 1993 (Anderson, 1996). The major emergent macrophtye species include cattails (*Typha latifolia, Typha angustifolia*), bulrushes (*Scirpus acutus*), and common reed (*Phragmites australis*). American Crystal Sugar subsequently built a second 64.8-ha FWS wetland at their Drayton, North Dakota, plant in 1993.

Vertical subsurface flow wetlands were tested in the British Isles (Morris and Herbert, 1997). Three first-stage beds were followed by four second-stage beds, totaling 400 $m^2$ for the whole pilot system. Warm wastewaters during the beet processing season showed good removals of 87, 88, and 80% for COD, TSS, and ammonia nitrogen, respectively, at 2.5–7.4 cm/d hydraulic loading. Cooler temperatures in the nonprocessing season gave removals of 74, 88, and 93%, at 17.4–18.6 cm/d hydraulic loading.

Constructed vertical upflow SSF wetlands are used by Usina Costa Pinto S.A. in Piracicaba, Brazil, to treat wastewater from a sugar mill. Rice is planted in the several cells in series, situated in terraces on a modestly steep slope (Figure 25.7). The strong influent has BOD up to 10,000 mg/L, which is reduced by about 50%.

### Potato Processing

Potato processing produces wastewaters with high starch content. This material is highly biodegradable, and has been the target of constructed wetlands treatment since 1990 (de Zeeuw *et al.*, 1990).

Lamb-Weston's Connell, Washington, facility uses free water surface treatment wetlands as part of its wastewater

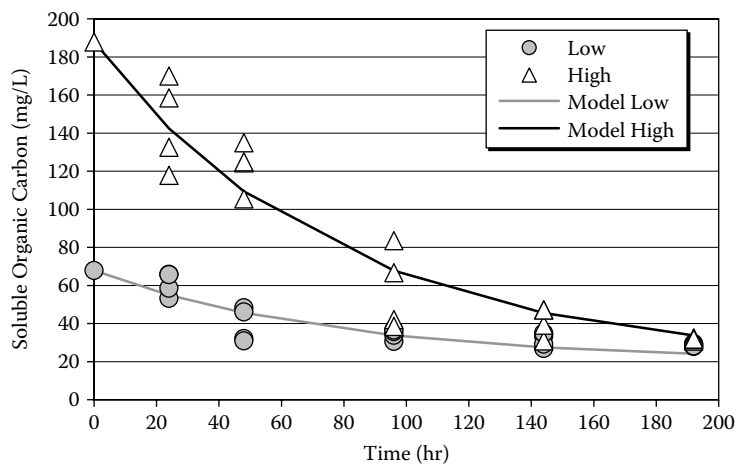



**FIGURE 25.6** Disappearance of soluble carbon in wetland microcosms dosed with sucrose. Data represent two levels of starting concentration, two levels of stirring, and two levels of supplementary nutrient addition. Model lines are for $k$ = 36 m/yr and $C^*$ = 20 mg/L. (Data from Gambrell *et al*. (1987) *Journal of the Water Pollution Control Federation* 59(10): 897–904; graph from Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)




**FIGURE 25.7** Vertical flow rice wetlands at Piracicaba, Brazil. Very strong sugarmill effluents (up to 10,000 mg/L BOD) flow upward through the root zone.

treatment system to handle the approximately 5,300 $m^3$ of wastewater it produces each day (Burgoon *et al.*, 1999). The goal of the project is to remove TSS and COD and to reduce nitrogen to the amount needed for growing an irrigated fodder crop. Two pilot projects were conducted (Kadlec *et al.*, 1997). The first investigated FWS wetlands only and produced 60–70% reduction in COD with two to four days' detention. The second was a prototype for the full-scale system, consisting of a FWS–VSSF–FWS combination.

In the full-scale project, clarified wastewater is pumped to a 10-ha lined FWS treatment wetland (Figure 25.8), followed by a 4-ha vertical SSF flow system, then to a 2-ha denitrification FWS treatment wetland, and finally ends in a 500-mL storage pond (Burgoon *et al.*, 1999). Treated water is then used for seasonal irrigation on 250 to 500 ha of fodder crops. The first FWS wetland provides TSS and COD removal, whereas the VSSF component is highly effective at nitrification. The final FWS wetland provides denitrification utilizing a small amount of raw influent as a carbon source (Burgoon, 2001). The holding pond provides storage so that the water may be irrigated only during the growing season. The design hydraulic retention time is 6.6 days (summer) and ten days (winter).

### Milk Products

The addition of milk-processing water to domestic treatment lagoons has generated de facto data for milk BOD, which is high in colloids. In Gustine, California, FWS wetlands received lagoon wastewater that was dominated by milk-processing effluent (Walker and Walker, 1990). Wetland cells 2B and 6D were 11.6 m × 337 m, and operated at 10.5–12.2 days detention for several months, during which time influent BOD was 280–440 mg/L. These cattail-bulrush cells achieved 86–89% reduction, corresponding to $k$ = 48 m/yr and $C^*$ = 4.2 mg/L for an assumed $P$TIS = 5 (see Chapter 8). A relatively smooth reduction occurred along the flow direction (see Figure 8.7, Chapter 8).

The Eichten Cheese, Center City, Minnesota, system consists of a septic tank, subsurface flow constructed wetland,



**FIGURE 25.8** Potato wastewater wetland at Connell, Washington.



**FIGURE 25.9** Treatment wetland at the Kilmeadan cheese factory in Ireland.

and an infiltration bed (Wallace, 2002c). The wetland and infiltration bed are insulated with mulch, and the wetland has operated without freezing problems. An aeration system was placed in the wetland cell to increase the oxygen transfer rate, to deal with the extremely high BOD. $CBOD_5$ removal was 94%, whereas total nitrogen removal was 74%. The Kilmeadan cheese factory near Waterford, Ireland, employs a multicell FWS wetland to improve discharge water quality (Figure 25.9).

A pilot surface flow wetland system was constructed in 1996 near Quebec City (Serodes and Normand, 1999). This system consisted of four similar cells with vertical walls, each 12 m long and 1.2 m wide. The cells were planted with indigenous broad-leaved cattail (*Typha latifolia*) harvested from a local natural marsh. Water depths of 4, 6, 10, and 10 cm, allowed nominal hydraulic residence times of 3.3, 5, 10, and 10 days, respectively. The influent was produced by adding 1,250 mL of raw milk, 55 g of detergent, and 35 mL of commercial phosphoric acid to 600 L of fresh water. Immediately following its production, this effluent had a COD of 425 mg/L and levels of *ortho*-P and total P of 12 and 19 mg/L, respectively. The results showed that COD elimination was rapid because, at mid-course in the 3.3-day wetland, COD reduction was already in excess of 85%. The presence of a residual concentration (27–57 mg/L) was noted, which could not be lowered further by more detention.

Perdomo *et al*. (2000) examined the performance of two mesocosm cells in series, containing *Eichhornia crassipes* and *Typha* spp. in all three permutations. The influent was wastewater from a milk products processing plant and contained high concentrations of BOD, TSS, TN, and TP. At several weeks detention time, 99, 97, and 96% removals of COD, TSS, and ammonia were found for the cattail–cattail combination.

A combination system of two vertical stages and a final HSSF stage was used at Loubressac, France, to treat cheese effluents, including milk serum (whey). This acid water was partially neutralized and the COD reduced significantly (Reeb *et al.*, 2004).

### Meat Processing

Wetlands are in use for treating abattoir wastewaters in a number of locations. These waters contain significant quantities of blood and other sources of BOD and are well treated in the wetland environment.

Pioneering work was done in the 1980s by Van Oostrom and coworkers in New Zealand (Van Oostrom and Cooper, 1990), and Finlayson and coworkers in Australia (Finlayson *et al.*, 1990). (Figure 25.10).

The New Zealand tests began with multiple wetlands, including three SSF systems (*Schoenoplectus validus*, *Glyceria maxima*, and unplanted) and three FWS trenches, planted with mixtures of *Typha, Glyceria*, and *Schoenoplectus* (Van Oostrom and Cooper, 1990). The target was removal of ammonia, and less than 22% removals were found at 4–7 cm/d hydraulic loading. The low removals were attributed to lack of nitrification. Subsequently, these researchers targeted nitrate removal in FWS with floating mats of *Glyceria* (Van Oostrom and Russell, 1994). Their pilot system reduced oxidized nitrogen by 37% at 4.6 cm/d hydraulic loading. Further work with floating *Glyceria* improved oxidized nitrogen removal to 46% at 5.7 cm/d hydraulic loading (Van Oostrom, 1995). Additionally, there was 66% COD removal and 96% TSS reduction.

Subsurface flow wetlands were tested for improving anaerobically digested abattoir effluents in Pachuca, Mexico (Rivera *et al.*, 1997). Two beds in series were used, vegetated with *Phragmites australis* and *Typha latifolia*. The nominal detention time in the two 12 m × 50 m beds was 1.7 days. Removals were 89, 87, and 89% for $BOD_5$, COD, and TSS, respectively, but much less for inorganic nitrogen species and



**FIGURE 25.10** FWS wetlands at Rangiuru, New Zealand. Strong meat plant effluents from the Affiliated Freezing Farmers Cooperative (AFFCO) are treated here.

phosphorus. Gutierrez-Srabia *et al.* (2004) also operated SSF wetlands in the State of Hidalgo, Mexico, to improve slaughterhouse wastewater quality after sedimentation and lagoon pretreatment. Good reductions were achieved in a 10.6-day detention in the wetland, 74% for COD and 44% for TSS, but only 9% for ammonia. Weedon (2004) investigated VF wetlands for abattoir effluent and found excellent initial removals of various contaminants (>95%). However, clogging ensued, and removals dropped considerably (60–90%).

A combination of meat-plant lagoon effluent from Cargill Foods Ltd. and municipal lagoon effluent from the town of High River, Alberta, has formed the major source of input water for the 1,246-ha Frank Lake wetland since 1989 (White, 1997). An essentially dry lake bed was thereby restored to a hemi-marsh condition. Approximately 60% of the incoming phosphorus load has been removed and stored in sediments, thus reducing incoming TP from 8.2 to 2.5 mg/L. Additional constructed treatment wetlands are in design to improve the quality of the incoming water to Frank Lake.

### Beer, Wine, and Spirits

The Coors Brewery in Golden, Colorado, operates a FWS treatment wetland for unspecified purposes. Billore *et al.* (2001) described a four-cell SSF wetland to treat secondary effluent from Associated Alcohols and Breweries, Ltd., in Baraha, India. The pilot project operated at 6.6 days nominal detention time and was vegetated by *Typha latifolia* and *Phragmites karka*. The system achieved 48, 49, and 84% reductions in TSS, COD, and ammonia, respectively.

Both FWS and SSF wetlands are in use to improve the quality of winery effluents (Kerner and Rochard, 2004) (Figures 25.11 and 25.12). As of 2002, there were seven wineries using treatment wetlands to treat winery wastewater in



**FIGURE 25.11** Full-scale vertical SSF wetland for treating winery wastewater at Chateau Mont-Redon, Chateauneuf-du-Pape, France.



**FIGURE 25.12** The Grove Mill winery FWS wetland in the Waihopai Valley near Blenheim, Marlborough, New Zealand.

California (O'Brien *et al.*, 2002). Two have free water surface wetlands, four have subsurface flow wetlands, and one uses both free water surface and subsurface wetlands technology. Two additional subsurface wetlands are either under construction or in permitting. Targets for treatment are typically COD, TSS, ammonia, and nitrate. Performance of these constructed wetlands has been found to depend on the management at each winery. Results also show that wineries with a pretreatment system, prior to the wetlands, perform better than those without a pretreatment system.

Pilot work on SSF treatment of winery wastewater was performed in 1996–1997 (Grismer *et al.*, 2001; Shepherd *et al.*, 2001a), using wastewaters from the Fetzer Vineyards, Hopland, California. Results indicated excellent reductions in TSS and COD. COD in the range 2,000–5,000 mg/L was reduced to 20–60 mg/L in about ten days' detention, and TSS was reduced from 300–600 mg/L to 12–32 mg/L. A full-scale system was placed in operation in late 1998 and has had some success (Grismer *et al.*, 2003) (Table 25.9).

In Italy there are at least three systems involving combinations of SSF, VSSF, and FWS components (Masi *et al.*, 2002). Very good removals have been reported at low hydraulic loading rates of 1.3–3.7 cm/d (Table 25.10). Tests have also given good results in Germany (Müller *et al.*, 2002), and in France (Rochard *et al.*, 2002).

Attempts at parameterizing removal models for HSSF systems have met with mixed results (Sheridan *et al.*, 2006).

### Vegetables

Anchor Foods, located in Pecos, Texas, processes onion rings as well as cheese and vegetable appetizers for the frozen-food market (O'Brien *et al.*, 2002). Wastewater was originally routed to two lagoons and then disposed of through flood irrigation on adjacent land. In 2001, the plant added two small sedimentation basins and a free water surface treatment wetland totaling 1.4 ha. The effluent from the wetland system is then land applied through flood irrigation. The permitted flow rate is 760 $m^3$/d.

A two-bed SSF wetland was built at Gradisce, Slovenia, in 1991 to treat food-processing effluents (Vrhovsek *et al.*, 1996). At hydraulic loadings of 2.6–5.5 cm/d, removals were 92%

**TABLE 25.9**
**Summary of Inlet and Outlet Water Quality for the Hopland Winery SSF Constructed Wetland**

| | Noncrush HRT = 5.5 d | | | Crush HRT = 1 h | | |
|---|---|---|---|---|---|---|
| **Parameter (mg/L)** | **Inlet** | **Outlet** | **Percent Removal** | **Inlet** | **Outlet** | **Percent Removal** |
| TSS | 1,042 | 110 | 85 | 1,428 | 808 | 30 |
| COD | 1,721 | 362 | 79 | 7,406 | 3,748 | 49 |
| Tannin | 55 | 12.1 | 78 | 55.2 | 30.0 | 46 |
| Nitrate | 1.8 | 0.5 | 73 | 13.1 | 10.9 | 17 |
| Ammonium | 118 | 45 | 62 | 37 | 26 | 29 |
| TKN | 159 | 54 | 66 | 43 | 32 | 25 |
| Sulfate | 35 | 2 | 95 | 83 | 62 | 25 |
| Sulfide | 0.56 | 0.12 | 78 | 0.88 | 0.70 | 20 |

*Note:* During the crush season, some overland flow occurred.

*Source:* Data from Grismer *et al.* (2003) *Water Environment Research* 75(5): 412–421.

**TABLE 25.10**
**Water Quality for Three Italian Winery Treatment Wetland Systems**

| Site | | Cecchi | | Ornellaia | | La Croce | |
|---|---|---|---|---|---|---|---|
| Type | | *SSF–FWS* | | *VSSF–SSF–FWS-Pond* | | *SSF* | |
| Area | $m^2$ | 1,330 | | 768 | | 215 | |
| Flow | $m^3/d$ | 35 | | 10 | | 8 | |
| **Parameter** | | **Inlet** | **Reduction** | **Inlet** | **Reduction** | **Inlet** | **Reduction** |
| TSS | mg/L | 222 | 89% | 102.7 | 75% | — | — |
| COD | mg/L | 4,045 | 98% | 1003 | 92% | 722 | 88% |
| $BOD_5$ | mg/L | 1793 | 98% | 425 | 93% | 354 | 92% |
| TN | mg/L | 14.7 | 82% | 26.6 | 90% | 65.2 | 58% |
| TP | mg/L | 4.9 | 73% | 1.92 | 94% | — | — |

*Source:* Data from Masi *et al*. (2002) *Winery High Organic Content Wastewater Treated by Constructed Wetlands in Mediterranean Climate*. Proceedings of the 8th International Conference on Wetland Systems for Water Pollution Control, 16–19 September 2002; IWA Publishing and University of Dar Es Salaam: Arusha, Tanzania.

for COD and 86% for ammonia. Del Bubba *et al.* (2004) operated a pilot-scale reed bed for olive mill wastewater treatment. Ten-to-one dilution was required to maintain healthy vegetation and gave sustainable removal efficiencies, especially for COD ($74.1 \pm 17.6\%$) and polyphenols ($83.4 \pm 17.8\%$).

## 25.3 LANDFILL LEACHATE

Old and new landfills produce leachates, typically formed from infiltrating waters and the products of solid-waste decomposition. Those contaminated leachate waters are a potential threat to surface and subsurface receiving waters. A very wide spectrum of pollutants is possible, reflecting the character of the materials contained in the stack. The principal categories of undesirable substances are (1) volatile organic compounds (VOCs), (2) nutrients, notably nitrogen, (3) heavy metals, and (4) priority toxic organic compounds. Leachates are produced when rainfall and percolated groundwater combine with inorganic and organic degraded waste. In unlined landfills, leachates frequently discharge to groundwater or appear as surficial drainage around the base of the landfill. In modern lined landfills, leachates are collected from the lined cells and routed to treatment units.

The highly variable nature of solid waste, differences in age and decomposition, and the diversity of chemical and biological reactions that take place in landfills result in a wide range of chemical quality of leachates. From a review of an "average" landfill, concentrations of COD, volatile acids, and nitrogenous compounds increase in landfill leachate during the first few years of operation and then decline over 10 or more years. Table 25.11 provides typical ranges encountered in landfill leachates. There is, consequently, a strong distinction between waters coming from open, operating landfills and waters emanating from closed landfills. The fresh leachate from an active or newly closed landfill system can have very high levels of organics and nitrogen. After a period of many years, the concentrations in the leachates are likely to be much lower, and continue to decrease with time. Flows are generally low but may vary depending on management and minimization of percolation from rainfall.

One common option for disposal of leachate is to send it to a municipal treatment plant. Depending upon the character of the refuse, the leachate can contain a number of constituents not normally present in municipal or domestic wastewater. These primarily include metals, a spectrum of volatile organics, and toxic compounds. These substances are of concern to municipal treatment plant operators for two reasons.

**TABLE 25.11**
**Typical Landfill Leachate Characteristics Over Time**

| | Concentration (mg/L) | | |
|---|---|---|---|
| **Parameter** | **1 Year** | **5 Years** | **15 Years** |
| BOD | 20,000 | 2,000 | 50 |
| TKN | 2,000 | 400 | 70 |
| $NH_4$-N | 1,500 | 350 | 60 |
| TDS | 20,000 | 5,000 | 2,000 |
| Chloride | 2,000 | 1,500 | 500 |
| Sulfate | 1,000 | 400 | 50 |
| Phosphate | 150 | 50 | — |
| Calcium | 2,500 | 900 | 300 |
| Sodium and potassium | 2,000 | 700 | 100 |
| Iron and magnesium | 700 | 600 | 100 |
| Aluminum and zinc | 150 | 50 | — |

*Source:* From McBean and Rovers (1999) In *Constructed Wetlands for the Treatment of Landfill Leachates*. Mulamoottil *et al*. (Eds.), Lewis Publishers, Boca Raton, Florida. Reprinted with permission.



**FIGURE 25.13** Marinha Grande, Portugal, HSSF leachate treatment wetlands. These are newly planted at the time of this photo.

First, the potential toxicity threatens the essential microbial populations that form the basis for processes such as activated sludge. Second, many of the potential contaminants are not treated by conventional processes, and pass through to the discharge point where they threaten compliance for the plant permit. Therefore, despite the low volumes of water involved compared to many treatment plant flows, the municipal treatment plant owners and operators do not want to accept landfill leachate. The leachate is usually of such small volume that pipelines to treatment plants are not economical, and the leachate must be trucked to the treatment plant. This is a costly procedure, requiring storage at the landfill site as well as transportation charges.

## Treatment Wetlands for Landfill Leachate

Natural wetlands are often the recipients of landfill leachates simply because many landfills are adjacent to wetlands or partially fill them. In such cases, the natural wetlands have been observed to improve water quality (Higgins and Lugowski, 1996; La Forge, 1997). Extensive studies of a very large wetland receiving leachate in Florida demonstrated that the ecosystem could easily absorb all leachate constituents (Schwartz *et al.*, 1999). Wetland treatment of the leachate is an attractive alternative method of water quality improvement (Loer *et al.*, 1999; Rew and Mulamoottil, 1999). The volumetric flow of leachate is often fairly small, compared to other wastewaters, such as domestic, animal, and industrial, or urban or agricultural runoff. Thus, long detention times are possible for small wetland–landfill area ratios. The use of constructed wetlands to treat these landfill leachates began to develop in the late 1980s through the mid-1990s, with both subsurface flow wetlands (Trautmann *et al.*, 1989; Staubitz *et al.*, 1989; Cooper and Hobson, 1989; Birkbeck *et al.*, 1990; Robinson, 1990) and surface flow wetlands (Mackie and Murphy, 1992; Martin *et al.*, 1993; Martin, 1994; Martin and Moshiri, 1994). Subsequently, wetland treatment of landfill leachates has been successfully tested at several locations (Figure 25.13). A brief review of existing work is given here.

Surface flow wetlands have been operating successfully in Escambia County, Florida, since 1990 (Martin *et al.*, 1993; Martin, 1994; Martin and Moshiri, 1994; DeBusk, 1999). Another warm-climate system exists near Mobile, Alabama (Martin *et al.*, 1999; Johnson *et al.*, 1999). Full-scale FWS constructed wetlands were implemented near Cambridge, Minnesota, which are operated only during the warm periods of the year (MPCA, 1995; Loer *et al.*, 1999; Kadlec, 2003e). A full-scale combination of a VF and a FWS wetland is in use at Saginaw, Michigan, for seasonal treatment of leachate (Kadlec, 2003e).

Cold-climate systems are functioning properly in Norway (Mæhlum, 1994; Jenssen *et al.*, 1994a; Mæhlum *et al.*, 1999; Mæhlum, 1999; Mæhlum *et al.*, 2002), as well as at several locations in Canada: Sarnia, Ontario (Higgins and Brown, 1999); Huneault, Ontario (Castonguay *et al.*, 1996; Sartaj *et al.*, 1999); Richmond, British Columbia (Birkbeck *et al.*, 1990; Hunter *et al.*, 1993); Sackville, Nova Scotia (Pries, 1994). More recently, a FWS system was placed in operation at Dartmouth, Nova Scotia (Galbrand, 2006).

A facility at Fenton, New York, has been operating since 1989 (Trautmann *et al.*, 1989; Grunder and Lauve, 1993). A facility in Tompkins County, near Ithaca, New York, has also been operating since 1989 (Staubitz *et al.*, 1989; Peverly *et al.*, 1993; Surface *et al.*, 1993; Sanford *et al.*, 1995a; Sanford *et al.*, 1995b; Sanford, 1999), which has utilized subsurface flow wetlands. There is also a nearby treatment wetland facility at Monroe County, New York (Eckhardt *et al.*, 1999).

Reedbeds are used to treat leachate in the United Kingdom (Cooper and Hobson, 1989; Robinson, 1990; Barr and Robinson, 1999; Robinson *et al.*, 1999), Slovenia (Urbanc-Bercic, 1994; Bulc *et al.*, 1997; Vrhovsek *et al.*, 2000; Sajn Slak *et al.*, 2004;

Bulc, 2006), Poland (Agopzowicz, 1991; Obarska-Pempkowiak *et al.*, 2005), and Sweden and Norway (Westholm, 2004; Keitliska *et al.*, 2005). Applications of treatment wetlands in warmer climates have also been reported (Aluko and Sridhar, 2005; Ferreira *et al.*, 2006).

Intensive research was conducted at a multicell pilot facility in New Hanover County, North Carolina (Rash and Liehr, 1999; Kozub and Liehr, 1999; Liehr *et al.*, 2000). Both surface and subsurface wetlands were tested with leachate lagoon effluent and mechanically pretreated leachate. The Water Environment Research Foundation has published a research report (Liehr *et al.*, 2000) on the pilot studies, which formed the basis of design for the full-scale New Hanover County facility. The full-scale facility consists of a free water surface wetland discharging to spray irrigation of a hay crop located on the landfill cap (CDM, 2001). A combination system involving several types of wetlands has been implemented near Des Moines, Iowa (Metro Waste Authority, 2005).

Recently, more intensive engineering has been added to wetland systems. Aeration and sedimentation (Mæhlum, 1999; Mæhlum *et al.*, 2002) and forced aeration within wetlands (Higgins and Brown, 1999; Wallace, 2001b; Nivala, 2005; Wallace *et al.*, 2006b; Nivala *et al.*, 2007) have produced favorable results. Wetlands have also been proposed for control of stormwater runoff from capped landfills (Mackie and Murphy, 1992). Cotreatment of municipal wastewater and leachate has been investigated, in which the wastewater actually provides dilution of the stronger leachate (Johansen and Brix, 1998). Vegetation response is an issue for very strong leachates and, consequently, there have been some botanical investigations as well as studies of other biota (Kylefors *et al.*, 1994; Bernard and Lauve, 1995; Bernard, 1999; Kelly-Hooper, 1999).

In addition to full-scale and pilot-scale system studies, there are numerous leachate treatability studies that have been undertaken. These involve the use of mesocosms or potted plants to gain an understanding of the possibilities for implementation of treatment wetland technology. However, they are usually of short duration, under more or less controlled conditions, and cannot be directly used in scale-up or design. These span a variety of wetland types: HSSF, VF, and FWS. The reader is referred to recent treatability studies (Kinsley *et al.*, 2002; Cao *et al.*, 2002; Silva *et al.*, 2003; Connolly *et al.*, 2004; De Feo *et al.*, 2005; Ujang *et al.*, 2005; Yang and Tsai, 2005; Sun and Austin, 2006; Kõiv *et al.*, 2006; Ling, 2006; Polprasert and Sawaittayothin, 2006; Rustige and Nolde, 2006; Sawaittayothin and Polprasert, 2006).

Ancillary benefits are difficult to ascribe to leachate systems. Benyamine *et al*. (2004) attempted to quantify such benefits for a multifunctional project near the municipality of Örebro, Sweden. The three goals of the project are to create an attractive recreational area, increase biodiversity, and manage pollution, with the major part of the project focused on increasing bird diversity. However, the leachate water treatment was designed primarily from an engineering perspective, and the biodiversity implications of the wetlands had a low priority. The success of the project could not be determined because of a lack of specific goals and data.

## Design Considerations

Design of leachate treatment wetlands can partially rely upon known wetland processes and performance, but unfortunately, the complexity of the leachate chemistry means that new concepts must be invoked. In other words, leachate treatment is not just about reducing TSS, BOD, and nutrients. Some of the important additional factors are as follows:

- Very small flows, which interact strongly with rainfall and evapotranspiration
- Survival of the wetland vegetation
- Very high amounts of ammonia to be removed
- Reduction of volatile hydrocarbons (BTEX, etc.)
- Reduction of trace toxicants (PAHs, PCBs, etc.)
- Removal of considerable amounts of iron
- Removal and storage of trace metals
- Sensitivity and protection of receiving waters

The reader will recognize that all of these subjects have been addressed in previous sections of this book, but leachate treatment moves into new territory compared to municipal wastewater treatment. Each of these peculiarities is explored further.

## Water Budget Considerations

Leachate is generated as a result of rainwater that falls on the landfill. Some of that rain runs off as surface flow, usually termed stormwater, and this water does not interact with the decomposing materials interior to the landfill. Another portion of the rainfall is evaporated and transpired by vegetation on the mound. The balance of the rainfall soaks into the mound and adds to the groundwater inside the mound. The relative proportion that soaks in is in part determined by the nature of the cap on the mound, if any. If the landfill is lined, then there is no other source of water adding to that groundwater mound. If the water table within the landfill is allowed to increase, at some point seeps will develop around the perimeter of the landfill. This seepage water carries pollutants acquired from the decomposing garbage. Given this scenario, it is possible to estimate the amount of leachate as a fraction of the precipitation received by the mound. Suppose a 10-ha mound receives 50 cm of rain annually, and one quarter of that soaks in, with the balance occurring as stormwater runoff and *ET*. The annual volume entering the mound would be 12,500 $m^3$, or 34 $m^3$/d. Next, consider a treatment wetland loaded at 1.0 cm/d, a very light loading that would produce about one month's nominal detention in a FWS wetland. The area would be 3,300 $m^2$, or 0.33 ha. Such a small wetland is very likely to fit within the project boundaries, without the acquisition of off-site property.

There are other ramifications of the relatively small flow. In cold climates, winter storage becomes possible in ponds of relatively small size. Therefore, it is possible to avoid the penalty of low microbial activity in cold waters. Another feature is that a treatment wetland interacts with rainfall and

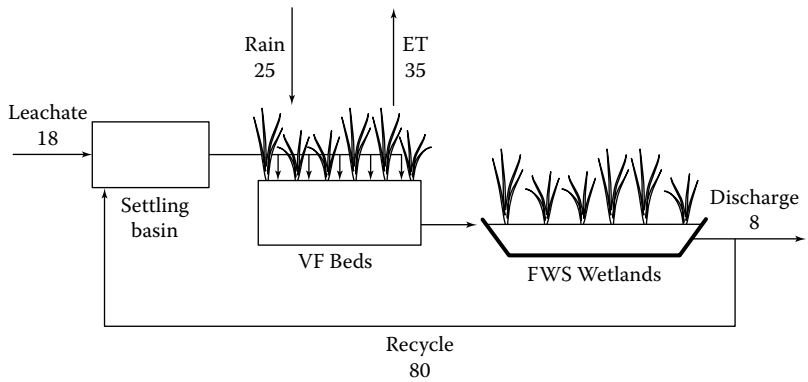



**FIGURE 25.14** Water flows during the operating season for the full-scale Saginaw, Michigan, leachate treatment wetland system ($m^3/d$). The values are averages for six years of operation, 2000–2005. Note that during periods of no rain, *ET* more than consumes the incoming water flow.

evapotranspiration (*ET*), which are seasonally not in balance. In arid climates, for instance, the warm period experiences high *ET* and low rainfall. If the wetland operates during that period, it will produce a net loss of water even to the point of having no discharge.

As an example, consider the performance of the Saginaw, Michigan, leachate treatment system (Kadlec, 2003c). This system operates only during the unfrozen season, approximately April through November. There is adequate storage in the landfill to accommodate any small winter water additions. The system consists of a $12 \times 12 \times 2$ m settling basin, a 600-$m^2$ VF wetland, and a 0.74-ha FWS wetland (Figure 25.14). On average, over the operational season of about six months per year, there is an excess of *ET* over rainfall of about 10 $m^3/d$. The incoming flows average 18 $m^3/d$, and so about half the water is lost on average. But, rainfall is episodic, and during inter-event periods, *ET* more than consumes the average inflow, leading to periods of zero discharge. During a six-year period of record, there was zero discharge 71% of the days during the warm operating season.

## Wetland Vegetation

Leachate water quality is potentially detrimental to a number of wetland plant species. In particular, ammonia levels can be exceptionally high (see Table 25.11). This narrows the choice of wetland plants to those that can tolerate elevated levels of ammonia, and even then steps may be needed to protect the plants. One such step is to recycle treated effluent to the front of the wetland, as shown in Figure 25.14 for the Saginaw, Michigan, system. The ammonia levels at various points in the system are shown in Figure 25.15. The averages for six years were raw influent: 335 mg/L, sedimentation basin outlet: 99 mg/L, vertical flow outlet: 22 mg/L, and FWS outlet: 2.5 mg/L. It is quite possible that wetland plants could not survive the raw influent—indeed, there were none to be

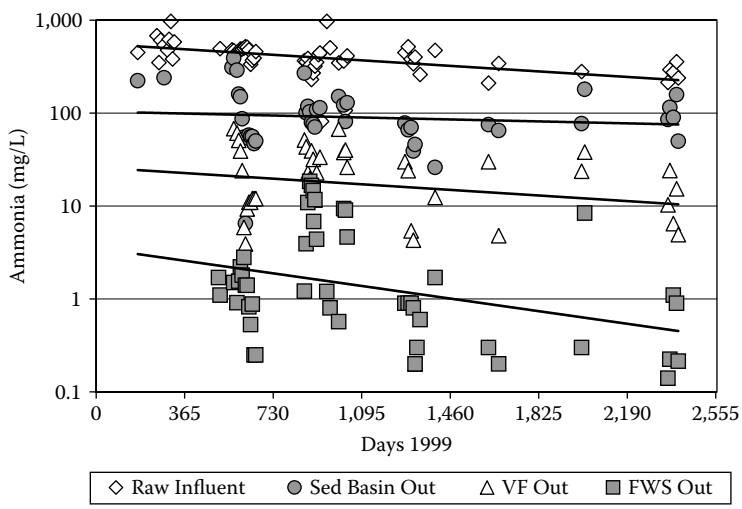



**FIGURE 25.15** Ammonia concentrations for the full-scale Saginaw, Michigan, leachate treatment wetland system, for six years of operation, 2000–2005. Note that during periods of no rain, *ET* more than consumes the incoming water flow.

found in the immediate vicinity of seeps prior to the project. The recycle plus the vertical flow unit reduced ammonia to 22 mg/L, which is an acceptable level for a wide variety of wetland plants.

### Ammonia Removal

High ammonia concentrations imply a need to remove very high fractions of the incoming load. Both "passive" and "active" wetland treatment systems have been used to treat landfill leachates (Kadlec, 1999b). Active wetland treatment systems typically employ some mechanism to boost oxygen transfer in order to oxidize the high concentrations of ammonia commonly found in landfill leachates (Vasel *et al.*, 2004). Single-unit constructed wetlands often cannot treat the high concentration of ammonia present in the leachate due in part to insufficient oxygen transfer but also due to the hydraulic inefficiencies of those single units. It has been shown in preceding chapters that oxygen transfer rates are of the order of 2–4 gO/m$^2$·d for FWS and HSSF systems, which constrains the capability of the wetland to microbially reduce ammonia. Preliminary evidence suggests that Anammox processes may or may not be present in leachate wetlands. Sun and Austin (2006) argued that completely autotrophic nitrogen removal over nitrite occurred in one of three leachate microcosms. However, Polprasert and Sawaittayothin (2007) found none of the microbes needed for such a process in their mesocosms. At this point in time, there is no known way to foster such advantageously efficient anomalous nitrogen processing, and this remains an area of wetland technology development.

Vertical flow wetlands are a proven technology for increasing oxygen transfer rates. They can be difficult to operate in cold climate conditions, and therefore the alternative of forced aeration can be considered to provide better oxygen availability (see preceding text).

However, no single unit can provide very high percent removals because of the constraint of hydraulic efficiency. If concentrations of hundreds of mg/L must be reduced to single-digit mg/L, reductions of 99+% are called for. Therefore, absolutely no short circuits can be tolerated, and must be prevented by multiple cells or multiple units.

The Saginaw system is an example of the use of VF and FWS to provide adequate oxygen transfer and multiple units in series (Figure 25.14). The VF units are loaded at about 15 cm/d, comprised of about 20% fresh feed and 80% recycle. The ammonia load removed is 11 gN/m$^2$·d, or 80% of the incoming ammonia, which has an implied oxygen requirement of 50 gO/m$^2$·d. The FWS units are comprised of two units in parallel, each compartmentalized by cross deep zones. These are loaded at 1,100 kg/ha·yr, which is in the zone of microbial control, and produces an outlet ammonia of 2.5 mg/L, or a 91% mass removal in the FWS wetland alone. This performance is in line with the expected performance of other FWS systems (see Figure 9.39, Chapter 9), and is represented by a first-order removal rate coefficient of 14 m/yr, which is the 60th percentile for FWS wetlands (see Table 9.31, Chapter 9).

The oxidation of ammonia in either an aerated HSSF bed or a VF bed leads to the production of oxidized nitrogen, as demonstrated in the leachate treatability study of Kinsley *et al.* (2002). Because of a deficiency of carbon after such aerobic treatment, it is necessary to supply the required carbon either by a chemical (BOD) addition (Yang and Tsai, 2005), or by reliance upon the carbon generated in a FWS wetland.

### Volatile Hydrocarbons

Perhaps the most challenging yet the most promising application of constructed wetlands is for the control of the large number of single organic compounds that are the subject of landfill leachate regulation. Very few of these have been studied with respect to wetland technology. Those that have been investigated show removals due to volatilization and biological degradation in the wetland environment (see Chapter 13).

The prospect of small pulses of individual organics at erratic frequency and variable concentration requires a broad-spectrum treatment device, which is always poised for the next event. A shallow wetland water body provides the opportunity for air stripping of volatiles. The efficiency is not as great as for mechanical devices, but the difference is more than compensated by the long detention times in the wetland. Half-lives (time to volatilize one-half the substance) range from two to four days for insoluble compounds such as benzene, toluene, and naphthalene (Mackay and Leinonen, 1975; Shugui *et al.*, 1994); and are less for more volatile compounds, such as vinyl chloride and chloromethane. Therefore, a long-detention, FWS wetland is effective in removal of volatile substances from leachate (Kadlec, 2003c). Because many of the organics are below detection on many occasions, another metric of their removal is needed. The hit frequency may be used, calculated as the fraction of samples that produced an above-detection result. For instance, out of 69 analytes tested at Isanti-Chisago, Minnesota, only 33 were ever detected in the raw water, and only 5 after the wetland (Table 25.12). The corresponding hit frequencies for those analytes that were detected were 0.16, 0.16, 0.12, 0.04, and 0.04. In other words, only 7% of the compounds were found in the wetland effluent (5/69), and those 5 were only detected in about 10% of the samples.

Clearly, the concepts of removal rate constants or loading charts are not applicable to forecast outlet concentrations for these occasional compounds, for which only left-censored data are available. However, it is possible to determine fractional removals for certain model compounds, such as benzene.

### Trace Toxicants

Heavier organics, such as polycyclic aromatic hydrocarbons (PAHs), are also trapped and degraded in treatment wetlands. PAHs partition very strongly to sediments. Because PAHs are not required for plant metabolism and growth, uptake is dependent on concentrations and supplies to the

**TABLE 25.12**
**Tracking of Inlet-Detectable Organic Contaminants through the Isanti-Chisago, Minnesota, Treatment System for Five Years**

| | Raw Water Concentration (μg/L) | Cascade Hit Frequency | Wetland Out Hit Frequency |
|---|---|---|---|
| *Number of Samples* | *34* | *34* | *25* |
| Acetone | 22.5 | 0.06 | 0.00 |
| *n*-Butyl benzene | 2.9 | 0.53 | 0.00 |
| *sec*-Butyl benzene | 0.8 | 0.27 | 0.00 |
| *tert*-Butyl benzene | 2.7 | 0.09 | 0.00 |
| Chlorobenzene | 0.7 | 0.77 | 0.00 |
| Chloroform | 0.2 | 0.03 | 0.00 |
| Chloromethane | 17 | 0.41 | 0.00 |
| 1,2 Dichlorobenzene | 0.3 | 0.35 | 0.00 |
| 1,4 Dichlorobenzene | 0.8 | 0.71 | 0.00 |
| Dichlorodifluoromethane | 6.7 | 0.32 | 0.00 |
| 1,1 Dichloroethane | 3.9 | 1.00 | 0.00 |
| 1,2 Dichloroethane | 0.3 | 0.53 | 0.00 |
| 1,2 Dichloroethylene *cis* | 2.2 | 1.00 | 0.00 |
| 1,2 Dichloroethylene *trans* | 0.4 | 0.88 | 0.00 |
| Dichlorofluoromethane | 3.2 | 0.94 | 0.00 |
| 1,2 Dichloropropane | 0.5 | 0.85 | 0.00 |
| Isopropyl benzene | 0.7 | 0.35 | 0.00 |
| *p*-Isopropyl toluene | 1.5 | 0.27 | 0.00 |
| Methylene chloride | 0.7 | 0.21 | 0.00 |
| Methyl *tert*-butyl ether | 2.6 | 0.32 | 0.00 |
| Naphthalene | 1.1 | 0.27 | 0.00 |
| Benzene | 3.1 | 1.00 | 0.00 |
| Toluene | 0.4 | 0.53 | 0.12 |
| Ethyl Benzene | 0.6 | 0.65 | 0.00 |
| Xylenes *m*, *p*, *o* | 1.2 | 0.79 | 0.04 |
| BTEX | 4.7 | 1.00 | 0.52 |
| 1,1,2,2 Tetrachloroethylene | 1.1 | 0.27 | 0.00 |
| 1,1,2 Trichloroethylene | 0.3 | 0.79 | 0.00 |
| 1,3,5-Trimethylbenzene | 0.6 | 0.09 | 0.00 |
| 1,2,4-Trimethylbenzene | 1.1 | 0.38 | 0.00 |
| Vinyl chloride | 6.7 | 1.00 | 0.00 |
| Chloroethane | 2.8 | 0.91 | 0.04 |
| Ethyl ether | 65 | 1.00 | 0.16 |
| Tetrahydrofuran | 38 | 0.62 | 0.16 |

*Note:* Concentrations are medians of the samples above detection. Hit frequency is fraction of samples above detection limit.

pore water in the root zone. Only a small percentage of those sorbed substances are biologically extractable by plants (Gomez-Hermosillo *et al.*, 2000). In contrast, the sorbed PAH is readily biodegraded by microbial processes (Jackson *et al.*, 1996; Machate *et al.*, 1997). Accretion of trace organics is mediated by gross wetland sediment accretion, but additionally, both anaerobic and aerobic zones exist in the wetland that can provide slow degradation. Heavy organic contaminants partition selectively to organic substrates, which are present in great abundance in a FWS wetland. For instance, 14 base-neutral PAHs were found in the leachate at the Saginaw, Michigan, site, but none have been found in the wetland system effluent over the seven-year period of record of sampling.

### Iron

A variety of iron items are discarded in trash, and therefore iron is a common constituent in landfill leachate. Concentrations of several milligram per liter are common. Under the anaerobic conditions within the mound, this iron exists in reduced (ferrous) form, but when the leachate surfaces in the treatment system, conditions become oxidizing, and the iron converts to the insoluble ferric form. The precipitates are

oxyhydroxides, which are gelatinous masses that can quickly clog any SSF system. At the Jones County Landfill HSSF facility near Anamosa, Iowa, iron oxidation and precipitation was observed to plug the bed and aeration lines within 18 months of start-up (Hoos, 2003; Nivala, 2005). Thus iron removal is often necessary prior to any SSF unit (see Figure 16.18, Chapter 16).

The two steps needed to effectively remove iron are aeration and settling. Passive devices, such as cascades, may be used for aeration, and ponds can be effective in sedimentation of the iron floc and coprecipitates. For example, at the Isanti -Chisago leachate treatment system, extracted water was cascaded down the side of the landfill in a polypropylene "step aerator," intended to increase dissolved oxygen concentrations and oxidize ferrous to ferric iron (Loer *et al.*, 1999; Kadlec, 2003c). A sedimentation basin was the second component of the treatment train, to allow settling of metal hydroxide flocs following cascade aeration (Figure 25.16). Basin size and residence time were selected based on settling rate studies and other factors.

### Trace Metals

A large number of metals find their way into leachate, resulting from the even bigger spectrum of metallic objects that are discarded into the trash. Metals are removed by cation exchange to wetland sediments, precipitation as sulfides and other insoluble salts, and plant uptake. The anaerobic sediments provide sulfate reduction to sulfide and facilitate chemical precipitation. A number of design options are available to take advantage of these wetland mechanisms, and these have been discussed in Chapter 11. Good choices are peat filters, usually VF (Sartaj *et al.*, 1999), because their large storage capacity, and FWS wetlands, because they renew the storage media by creating new peat each year. SSF sand/gravel wetlands are not as good a choice because of limited sorption capacity and the inability to form new storage sediments without clogging the bed.



**FIGURE 25.16** The inlet cascade and settling pond of the Isanti-Chisago, Minnesota, leachate treatment system. The FWS wetlands are in the background.

### Receiving Waters

There is a key question as to what should be the method of discharge or disposal of the treated leachate. Discharge to surface waters is an option if the treatment wetland is designed to capture or destroy all potential pollutants, and it is commonly assumed that surface water is to be the recipient. Regulators are justifiably nervous about this option because there are so many potential toxicants. Therefore, the usual suite of municipal treatment parameters becomes a necessary but not sufficient list of treatment goals. Understandably, treatment wetland studies have frequently borrowed the list of study variables from municipal applications. Besides TSS, BOD, and ammonia, each new situation is likely to require that other pollutants are monitored and controlled by the wetland system. The additional parameters are likely to include specific hydrocarbons and metals, especially those suspected or confirmed of being present in the leachate. In any case, it is likely, in the United States, that any such discharge to surface waters will be subject to a whole effluent toxicity test. Even if the treatment wetland system is adequately designed, the monitoring costs can escalate to onerous levels.

Another option for disposal of the treated water is to irrigate the top of the (closed) landfill mound for the purpose of sustaining a cover crop. This option is practiced at the New Hanover County, North Carolina, treatment wetland system (grasses) (CDM, 2001) and at Ormoz, Slovenia (trees) (Sajn Slak *et al.*, 2004). These systems have been successful, but there are caveats. First, leachates tend to have high levels of dissolved solids (TDS), which often are dominated by high chloride content (salt). Therefore, the cover crop to be irrigated must have high resistance to salinity. Secondly, irrigation needs are seasonal, and added water may be unnecessary in some seasons of the year. During such periods, the cover crop may be overwatered, and runoff may occur. Most closed landfills in the United States will have a stormwater system that receives mound runoff, and the treated leachate will sometimes augment that flow. However, the added water will have been treated in the wetland system and in what amounts to an overland flow system atop the mound. In other seasons, *ET* (crop consumption) may serve to eliminate any runoff, effectively creating a zero-discharge system. Any treated water that infiltrates an imperfect mound cap will add to subsequent leachate flows, effectively creating recycle through the mound.



**FIGURE 25.17** The infiltration wetland at the Isanti-Chisago, Minnesota, leachate treatment wetland facility. This is a constructed wetland whose purpose is disposal of the treated water.

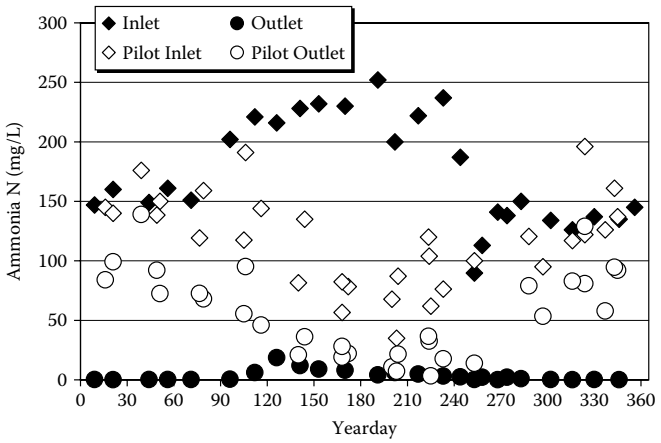



**FIGURE 25.18** Performance of the New Hanover County, North Carolina, FWS landfill leachate systems. The pilot project was 60 $m^2$ and operated at about 1.5 cm/d HLR for four years. The full-scale project is 21,000 $m^2$ and operated in its first year at 0.15 cm/d. Note that the influent concentration pattern of the full scale does not match that of the pilot.

Another, less frequent option is a groundwater discharge of the treated leachate. That option would usually require that there be no potable water usage from the receiving aquifer. A groundwater disposal option was implemented at the Isanti-Chisago, Minnesota, site (Figure 25.17), where the infiltrate system was a leaky constructed wetland. Local groundwater flows vented to downgradient wetlands of the region and were not tapped for human use.

### Scale-Up

There are well-known problems that properly limit the scale-up of mechanical plants of various sorts, and treatment wetlands are no exception. Conservative designs usually do not exceed a scale-up factor of the order of 10, or perhaps 100 if all variables seem to have been identified and are controlled in the larger system. This is an unwritten assumption in the many scale-up recommendations that have been promulgated for HSSF and VF wetlands treating municipal effluents. There is danger if the large-scale conditions do not closely match those of the small-scale prototype. Leachate treatment wetland technology is particularly susceptible to errors of scale-up because of the erratic composition time series and the known temporal drift in water quantity and water quality. Further, it has been noted that leachate wetlands will be strongly driven by rain and *ET*, which factors may be entirely different in a laboratory setting.

The New Hanover County, North Carolina, treatment wetland system provides an example of such scale-up pitfalls because there was a long-term pilot project preceding the full-scale project (Liehr *et al.*, 2000; CDM, 2001). The FWS pilot project was loaded at an HLR of 1.4 cm/d, which was not much affected by rain and *ET*, which were about equal. The pilot project achieved 50% reduction (127 to 63 mg/L) of ammonia over a four-year period of record. There were considerable drops in inlet ammonia during the warmest times of the year (Figure 25.18). Tracer testing showed the pilot wetland behaved as two TIS, and channels down the side of the wetland were visible. The full-scale project was 350 times larger and built with cross-deep zones. It operated in its first year at an HLR of 0.14 cm/d (10 times lower than design or the prototype), and reduced ammonia from 150 to 3 mg/L. The differences are attributable to: (1) lower hydraulic loading, (2) higher ammonia in summer high-removal period, (3) greater hydraulic efficiency, and (4) much greater effects of rain and *ET* at the very much lower HLR. In fact, the rate coefficients for ammonia are not much different if all factors are accounted. However, if "rules of thumb" are applied, there can be serious design mistakes.

## 25.4 INDUSTRIAL WASTEWATERS

Industrial wastewater is here a loosely defined category, including waters that are not from domestic, municipal, animal, or food product processing. A good many such treatment examples have already been discussed in connection with organics and metals. In this section, the focus is upon particular industries rather than particular contaminants. Only a few example categories have been chosen for illustration.

### Wood Leachates

Solid waste from the wood-processing industry includes wood chips, process trimmings, shredded bark, and sawdust from various saws, planes, and other process machinery. These materials, when stored unprotected outside and exposed to precipitation, will produce a leachate that is often characterized by a high carbon content, high oxygen demand, strong color, an array of dissolved metals, and low levels of nitrate, ammonia, and nitrogen (Frankowski, 2000). The carbonaceous component can contain high amounts of

tannins and lignins. Accordingly, researchers at the University of British Columbia have conducted mesocosm and pilot-scale studies to determine the effectiveness of FWS wetlands to treat these leachates. Based on successful lab treatability studies, Frankowski (2000) operated a set of six (6 m × 18 m) pilot-scale wetlands, treating wastewater with COD = 5,600 mg/L (BOD = 1,730 mg/L). The carbonaceous compounds included 500 mg/L of volatile fatty acids (VFA) and 870 mg/L of tannins and lignins, and the water had a pH of 3.8. At a hydraulic loading rate of about 3 cm/d (*n*HRT seven days), these cells were able to reduce COD (43%), VFA (14%), tannins and lignins (39%), and rainbow trout toxicity (49%). Continued operation (Masbrough, 2002) showed improved removals of COD (51%), VFA (69%), tannins and lignins (42%).

Based on the presumption that the concentrations of this wastewater were too high, a set of mesocosms were operated with the same waste diluted approximately 10:1 (Tao *et al.*, 2004). Further, this study explored the effect of changing detention time, with the expected result that increasing detention improved treatment. Process studies linked the performance to microbial activity (Tao and Hall, 2004; Tao *et al.*, 2006).

## Pulp and Paper Wastewaters

The pulp and paper industry converts wood products into cellulose pulp and paper. Raw wood and wood chips are converted to pulp (cellulose fibers) by mechanical grinding (groundwood) or through chemical degradation and leaching (sulfite and Kraft processes). Pulp mill effluents are complex mixtures of wood-derived organics. Raw wastewater from pulp and paper mills typically receives primary treatment through settling, either in ponds or in primary clarifiers. Secondary treatment is provided at most pulp and paper mills, but to meet reduced effluent limitations, some pulp and paper mills are being required to provide treatment beyond the secondary level. The goals of additional treatment depend on site conditions such as effluent quality and permit limits. Goals may include further reduction of $BOD_5$, TSS, nitrogen, phosphorus, color, and whole effluent toxicity. Constructed and natural wetland treatment systems have been tested at a number of pulp and paper mills to provide such advanced secondary or tertiary treatment (CH2M Hill, 1994; NCASI, 2004).

There is a difference in the character of the BOD between paper and municipal effluents. Lignin-based compounds that survive the pretreatment processes are not found in domestic wastewater, and are harder to degrade than starches and sugars. Nevertheless, wetlands have proven to be effective in the reduction of this specialized form of BOD. Data from six operating systems indicate that the *k*-values for degradation may be comparable to those for domestic wastewater.

Observations of pollutant assimilation in a partially vegetated facultative treatment pond at the Australian Paper Manufacturer's Maryville Mill in Victoria, Australia, led to the first published treatability experiments using wetland plants to treat pulp- and paper-mill wastewaters (Allender, 1984). In the experiment, five emergent wetland plant species were established in 4-L tubs. Researchers measured concentrations of lignosulphonate, color, total suspended solids, biochemical oxygen demand ($BOD_5$), and foaming tendency in FWS basins with and without plants over a very long holding period (three weeks). The presence of wetland vegetation was shown to enhance pollutant removal, but microbial populations were found to be responsible for most of the assimilation (CH2M Hill, 1994).

### Weyerhaeuser

In 1985, Weyerhaeuser Company began an initial pilot study using eight HSSF wetland mesocosms (2.6 $m^2$), 60 cm deep, to assess the effect of wetland vegetation. The wetland cells with plants were found to have higher removal efficiencies for ammonia-nitrogen, organic nitrogen, and phosphorus. Longer retention times resulted in higher removal efficiencies for most measured parameters (TSS, BOD, $NH_4$-N, Org-N, P). In the last stage of this study, the mesocosms were modified and replanted to study the effects of substrate depth. Table 25.13 presents the results of this experiment (Thut, 1993). The area of the shallow (15-cm) wetland was 2.7 times that of the deep (40-cm) wetland, and therefore utilized approximately 2.7 times the substrate and plant biomass. Thut (1993) ascribed the higher removal percentages for $BOD_5$, ammonia, and phosphorus in the 15-cm substrate depth to two reasons: (1) higher dissolved oxygen levels, and (2) the greater amount of plant biomass (both below- and aboveground).

Subsequently, a large HSSF pilot was placed in operation (Thut, 1993), with moderately good results. However, hydraulic issues were experienced, with a 15-cm head differential along the flow direction. The inlet zone became flooded (Figure 25.19), while the outlet zone became devoid of wetland plants because of too much dry gravel at the top of the profile (see, for example, Figure 21.14, Chapter 21).

**TABLE 25.13**
**Effect of Substrate Depth on Pollutant Removal in HSSF Wetlands Polishing Pretreated Pulp-Mill Effluent at a 15-Hour Retention Time**

| | Substrate Depth | |
|---|---|---|
| **Parameter** | **15 cm** | **40 cm** |
| TSS | 49% | 55% |
| $BOD_5$ | 46% | 28% |
| $NH_4$-N | 96% | 42% |
| Org-N | 31% | 33% |
| Phosphorus | 27% | 14% |

*Source:* Data from Thut (1993) In *Constructed Wetlands for Water Quality Improvement*. Moshiri (Ed.), Lewis Publishers, Boca Raton, Florida.



**FIGURE 25.19** The inlet zone of a HSSF wetland-treating pulp-mill effluent.

As a result of such difficulties, Weyerhauser went on to build a 27-ha FWS full-scale system, the only constructed full-scale facility. Eight flow paths with a total of 33 cells were used to treat 60,000 $m^3$/d of secondary water. Over a two-year period, about 80% of the TSS were removed (69 to 16 mg/L), and 48% of the BOD (27 to 14 mg/L). However, if a background BOD concentration is acknowledged, the high HLR (16 cm/d) means the removal rate coefficient of about 45 m/yr was near the central tendency for other types of BOD.

### Other Pilot Studies

In the early 1990s, several pilot projects were conducted. Boyd *et al*. (1993) attempted treatment of lightly pretreated raw wastewater with sprayed, inoculated SSF wetlands and found reductions of BOD from 1,200 to 300 mg/L. Hammer *et al*. (1993b) attempted color removal in HSSF mesocosms, with very little success. Tettleton *et al*. (1993) studied three FWS pilot wetlands (1,320 $m^2$) that received well-pretreated influents at HLR of 10 to 45 cm/d. BOD reduction was poor, at 6–7%; only nitrate reduction was moderately good, at 64–80%.

#### *Pope & Talbot*

The Pope & Talbot company began experimenting with constructed wetland treatment systems in 1990 at its Halsey, Oregon, mill (Moore *et al.*, 1994; Hatano *et al.*, 1994). Ten 0.14-ha wetland cells were constructed, one of which was a VSB cell; the remaining nine were FWS cells. Seven of the FWS wetland cells were supplied with pretreated pulp mill effluent, whereas two were fed clean water. TSS was reduced by 55–80% at an HLR = 24 cm/d, but $BOD_5$ was reduced by only 6–45%. Slight color removal was noted in the FWS wetland cells.

#### *Champion*

Champion International Corporation operated a pilot wetland system receiving secondary-treated effluent over two years at its Pensacola, Florida, bleached kraft mill facility. This pilot system consisted of six FWS wetland cells designed with varied length-to-width ratios, fractions of open water, and retention times (Knight *et al.*, 1994). A variety of contaminants were assessed, including biochemical oxygen demand ($BOD_5$), total suspended solids, ammonia, and color. The results of this study indicated that the cells with the longest L:W ratios (10:1) performed better than the other cells (L:W ratios of 5:1 and 2.5:1). The data from this study also showed that the inclusion of deep zones decreased the volumetric $k_V$ for aspect ratios of 5:1 and 10:1, whereas areal *k*-values were generally improved by the deep zones. No acute toxicity to cladocerans (*Ceriodaphnia dubia*) or fathead minnows (*Pimephales promelas*) was detected in any of the FWS wetland effluent samples. In contrast, all influent samples were toxic to cladocerans, and two out of three influent samples were chronically toxic to fathead minnows.

## Fertilizer Manufacture

### ICI Lambton

Imperial Chemical Industries Canada (ICI) operated a phosphate fertilizer plant at Courtright, Ontario, for many years, leaving behind a large gypsum waste pile, well-laced with phosphorus. Leachate from this and other "gyp stacks" is very high in sodium, calcium sulfate, and other chemicals. At this plant, leachate is subjected to pretreatment, including liming, and then passed through a series of ponds prior to discharge to the Detroit River. A decision was taken to convert the 12.5-ha terminal pond into a wetland, with both deep water and bench areas, to be fringed by appropriate upland plantings. Construction commenced in autumn 2002 and was completed in spring 2003. Ancillary benefits were a high priority, and the system contains nesting boxes, raptor perches, and basking logs for wildlife. Public access is facilitated with a parking area and three viewing and information kiosks. The area quickly developed to support a wide variety of indigenous trees, shrubs, and wetland vegetation.

### Simplot Brandon

Simplot Canada Limited has owned and operated a nitrogen fertilizer manufacturing facility in Brandon, Manitoba, since 1966. The original designs and operating procedures



**FIGURE 25.20** The Simplot wetland at Brandon, Manitoba.

of the facility in the 1960s and 1970s did not provide for treatment of process wastewaters or containment of leakages and spills. Plant wastewater, leakages, and spills were conveyed via unlined surface drainage ditches to an adjoining slough, which ultimately drained into the Assiniboine River. The result was nitrogen contamination of the unconfined groundwater aquifer below the facility. The waters that vent and travel to the river have about 200 mg/L of ammonia and a comparable amount of nitrate.

Simplot's engineered wetland was constructed in autumn 2002, consisting of a 4-cell, 22-ha FWS constructed wetland (Figure 25.20). A clay layer, located between 1 and 3 m below the surficial topsoil, provided for water retention. No liner was required, saving costs. Seeding of vegetative propagules (seeds, rhizomes) occurred in selected portions of the cell basins. Propagules were obtained from surface soils in ditches adjoining municipal road allowances in which nuisance aquatic macrophytic growth occurred. The species in the wetland cells include cattail, bulrush, arrowhead, and sedges. All species are native, and none are rare or endangered.

The system is designed to allow for gravity flow. The average annual flow in 2004 was 280 $m^3$/d. A steady reduction in both total nitrogen and ammonia was observed as water flowed through the wetland. A 39% reduction in TN and an 82% reduction in ammonia were measured in 2004. At that time wetland operations were still in the start-up phase, with very modest initial vegetative growth.

The day-to-day management of the wetland has been limited to Simplot's Environment Group at only one to two hours per week. Monitoring of the wetland is conducted via weekly tours, which involve recording water levels at each of the 12 control gates and taking samples where there is flow through the gates.

## SUMMARY

Although the majority of applications of treatment wetlands remain as municipal and domestic systems, there is a growing number of other application areas. A few of those other applications have been singled out in this chapter as examples of the growth of wetland technology. There is some removal of nearly every organic compound tested in wetland treatment systems, whether full scale, pilot scale, or microcosm. This includes organics from very important generators: pesticides, petrochemicals, paper and food wastes, and landfill leachates. In each of these categories, there are multiple studies that have initiated the design database. What is known thus far shows great promise for wetlands technology for organics control, especially in those situations where these passive systems can be fit into the landscape and accumulated organics will not pose a threat to wetlands biota. Animal wastewaters are similar to municipal effluents, but generally more concentrated. Therefore, with appropriate pretreatment, the application of treatment wetland technology to animal wastes is an easy extrapolation.

Perhaps the most challenging yet the most promising application of constructed wetlands is for the control of the large number of single organic compounds that are the subject of landfill leachate regulation. The prospect of small pulses of individual organics at erratic frequency and variable concentration requires a broad-spectrum treatment device, which is always poised for the next event.

Importantly, it is not possible or prudent to scale a wetland from one technology to another because of the flow and composition differences. Therefore, it is critical to examine each pollutant individually and apply removal forecasts based on a calibrated mechanistic model.

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# Appendix A: Lists of Basis Wetlands

This book is based on the analysis of data from many wetlands, and the associated experiences of cost and implementation. The appendix summarizes those wetland cells and systems that have contributed to this basis of analysis, by name and country (and by state when appropriate). Tables in the various chapters provide extensive referencing, dictated by availability of publications. In some instances, we have relied upon project reports and our own data compilations of published and unpublished results.

It is certain that differences in sampling and laboratory protocols have contributed significantly to the data scatter for wetland performance parameters. There is no simple way to evaluate data quality from the various wetlands, and any attempt to screen the data would be highly suspect in and of itself, as it would be a reflection of the bias of the reviewer. The periods of record vary, with some systems possessing many years of information. In general, very short periods of record, i.e., days or weeks, have been excluded. Laboratory microcosm studies are not included here because conditions are generally too far removed from field environments. We concluded early in the process of information analysis that there are no single "definitive" studies that are superior to others, despite the hopes of individual investigators. In fact, focusing on one study leads to the loss of understanding of intersystem variability and the full loading spectrum that has been explored by the greater number of wetlands.

The lists given here are intended to assist the reader in regionalizing a search for further information relevant to treatment wetlands in a particular climatic zone. However, the lists are not geographically balanced because treatment wetland technology is not geographically uniform, and data are not always accessible despite large numbers of systems in a given region.

The three main types of treatment wetlands are considered separately. Of the total of 950 basis wetlands, 488 are FWS, 362 are HSSF, and 100 are VF. It is understood that these proportions are not indicative of the entire universe of operating systems, but it is believed that the sample sizes are sufficient to characterize the three variants of the technology.

The systems listed in Tables A.1, A.2, and A.3 provided data that were utilized to determine *k*-rates, temperature coefficients, or background concentrations. This is by no means a definitive listing of treatment wetlands, but does provide the reader an indication of systems with significant monitoring data.

**TABLE A.1**
**Free Water Surface Wetlands**

| FWS System Name | Identifier | State/Province | Country |
|---|---|---|---|
| Bahco | | | Argentina |
| San Tomé | | | Argentina |
| Blacktown | | New South Wales | Australia |
| Bue Mountain | | New South Wales | Australia |
| Byron Bay | WB2-3 | New South Wales | Australia |
| Byron Bay | WB3-4 | New South Wales | Australia |
| Byron Bay | WB4-5 | New South Wales | Australia |
| Byron Bay | WB5-6 | New South Wales | Australia |
| Byron Bay | WB6-7 | New South Wales | Australia |
| Richmond | Open | New South Wales | Australia |
| Richmond | Myriophyllum | New South Wales | Australia |
| Tom's Gully | | Northern Territory | Australia |
| Woodcutters | | Northern Territory | Australia |
| Calamvale | | Queensland | Australia |
| Coorparoo | | Queensland | Australia |
| Piricicaba | | | Brazil |
| Vancouver | P15.1 | British Columbia | Canada |
| Vancouver | P15.2 | British Columbia | Canada |
| Vancouver | P30.1 | British Columbia | Canada |
| Vancouver | P30.2 | British Columbia | Canada |
| Vancouver | UP30.1 | British Columbia | Canada |
| Vancouver | UP30.2 | British Columbia | Canada |
| Oak Hammock | | Manitoba | Canada |

(*Continued*)

**TABLE A.1 (CONTINUED)**
**Free Water Surface Wetlands**

| FWS System Name | Identifier | State/Province | Country |
|---|---|---|---|
| Roblin | | Manitoba | Canada |
| Simplot | Brandon | Manitoba | Canada |
| Hay River | | Northwest Territories | Canada |
| Hebert | | Nova Scotia | Canada |
| Brighton | | Ontario | Canada |
| Cobalt | | Ontario | Canada |
| Elliot Lake | Elliot Lake | Ontario | Canada |
| Essex County | | Ontario | Canada |
| Falconbridge | | Ontario | Canada |
| Gloucester | | Ontario | Canada |
| Kanata | Kanata | Ontario | Canada |
| Kitchener | | Ontario | Canada |
| Lambton | | Ontario | Canada |
| Listowel | 1 | Ontario | Canada |
| Listowel | 2 | Ontario | Canada |
| Listowel | 3 | Ontario | Canada |
| Listowel | 4 | Ontario | Canada |
| Listowel | 5 | Ontario | Canada |
| Musselwhite | | Ontario | Canada |
| Perth County | | Ontario | Canada |
| Region of Ottawa–Carlton | | Ontario | Canada |
| Riverwalk | | Ontario | Canada |
| Arvida | | Quebec | Canada |
| Fontanges | | Quebec | Canada |
| Estevan | | Saskatchewan | Canada |
| Humboldt | | Saskatchewan | Canada |
| Guangdong | | | China |
| Alastaro | | | Finland |
| Flytträsk | | | Finland |
| Hovi | | | Finland |
| Kis-Balaton | | | Hungary |
| Warangal | | | India |
| County Galway | Bed 1 | | Ireland |
| County Galway | Bed 2 | | Ireland |
| Johnstown Castle | | | Ireland |
| Kilmeadan | | | Ireland |
| Rathermon | | | Ireland |
| Waterford | Anne Valley | | Ireland |
| Cecchi | | | Italy |
| Chiampo Valley | | | Italy |
| La Croce | | | Italy |
| Ornellaia | | | Italy |
| Harutori | 1 | | Japan |
| Harutori | 2 | | Japan |
| Harutori | 3 | | Japan |
| Kahokugata | | | Japan |
| Kojima | 1 | | Japan |
| Kojima | 2 | | Japan |
| Mizomoto | | | Japan |
| Nagano | 1 | | Japan |
| Nagano | 2 | | Japan |
| Nagano | 3 | | Japan |
| Nagano | 4 | | Japan |
| Sanno | 1 | | Japan |

**TABLE A.1 (CONTINUED)**
**Free Water Surface Wetlands**

| FWS System Name | Identifier | State/Province | Country |
|---|---|---|---|
| Sanno | 2 | | Japan |
| Sanno | 3 | | Japan |
| Sanno | 4 | | Japan |
| Sanno PWRI | 1 | | Japan |
| Sanno PWRI | 2 | | Japan |
| Sanno PWRI | 3 | | Japan |
| Sanno PWRI | 4 | | Japan |
| Sanno PWRI | 5 | | Japan |
| Sanno River | River | | Japan |
| Seimei | | | Japan |
| Watarase | | | Japan |
| Yasato-machi | | | Japan |
| Marrakech | | | Morrocco |
| Texel | 1 | | Netherlands |
| Texel | 2 | | Netherlands |
| Texel | 3 | | Netherlands |
| Texel | 4 | | Netherlands |
| Texel | 5 | | Netherlands |
| Texel | 6 | | Netherlands |
| Texel | 7 | | Netherlands |
| Texel | 8 | | Netherlands |
| Beachlands/Maraetai | | | New Zealand |
| Blenheim | | | New Zealand |
| Christchurch | | | New Zealand |
| Hautoriu | | | New Zealand |
| Hikumutu WWTP | | | New Zealand |
| Horotiu | | | New Zealand |
| Kohukohu | | | New Zealand |
| Opunake | | | New Zealand |
| Portland | | | New Zealand |
| Pukekohe | | | New Zealand |
| Rangiuru | | | New Zealand |
| Taumaranui | | | New Zealand |
| Titoki | | | New Zealand |
| Toenepi | | | New Zealand |
| Wellsford | | | New Zealand |
| Whangarei | | | New Zealand |
| Bolstad | | | Norway |
| Braskerud | A | | Norway |
| Braskerud | C | | Norway |
| Braskerud | F | | Norway |
| Braskerud | G1 | | Norway |
| Braskerud | G2 | | Norway |
| Esval | | | Norway |
| Orcopampa | | | Peru |
| Tucush | | | Peru |
| Frambork | | | Poland |
| Motorway | | | Slovenia |
| Mpophomeni | | | South Africa |
| Esklistuna | Ekeby | | Sweden |
| Hassleholm | Magle | | Sweden |
| Isgrannatorp | | | Sweden |
| Linkoping | | | Sweden |

(*Continued*)

**TABLE A.1 (CONTINUED)**
**Free Water Surface Wetlands**

| FWS System Name | Identifier | State/Province | Country |
|---|---|---|---|
| Oxelösund | | | Sweden |
| Snogeröd | | | Sweden |
| Vomb East | | | Sweden |
| Vomb West | | | Sweden |
| Kaohsiung | | | Taiwan |
| Airton | | | United Kingdom |
| Bishop Wilton | | | United Kingdom |
| Brentwood | | | United Kingdom |
| Carperby | | | United Kingdom |
| Cherry Burton | | | United Kingdom |
| Crow Edge | | | United Kingdom |
| Dagenham | | | United Kingdom |
| Meltham | | | United Kingdom |
| Northumberland | Shilbottle | | United Kingdom |
| Slingsby | | | United Kingdom |
| Stockley | Bed 1 | | United Kingdom |
| Stockley | Bed 2 | | United Kingdom |
| Stockley | Bed 3 | | United Kingdom |
| Tupton | | | United Kingdom |
| Wentworth Castle | Bed 1 | | United Kingdom |
| Wentworth Castle | Bed 2 | | United Kingdom |
| Wentworth Castle | Bed 3 | | United Kingdom |
| Wentworth Castle | Bed 4 | | United Kingdom |
| Wetwang | Bed 1 | | United Kingdom |
| Wetwang | Bed 2 | | United Kingdom |
| Wetwang | Bed 3 | | United Kingdom |
| Wheal Jane | | | United Kingdom |
| Auburn Poultry | 1 | Alabama | United States |
| Auburn Poultry | 2 | Alabama | United States |
| Auburn Poultry | 3 | Alabama | United States |
| Fort Deposit | T:1–2 | Alabama | United States |
| Hurtsboro | 1 | Alabama | United States |
| Hurtsboro | 2 | Alabama | United States |
| Muscle Shoals | | Alabama | United States |
| Sand Mountain | 1 | Alabama | United States |
| Sand Mountain | 2 | Alabama | United States |
| Sand Mountain | 3 | Alabama | United States |
| Sand Mountain | 4 | Alabama | United States |
| Sand Mountain | 5 | Alabama | United States |
| Widows Creek | | Alabama | United States |
| Anchorage | | Alaska | United States |
| Fairbanks | | Alaska | United States |
| Kingman | | Arizona | United States |
| Show Low | | Arizona | United States |
| Tres Rios | C1 | Arizona | United States |
| Tres Rios | C2 | Arizona | United States |
| Tres Rios | H1 | Arizona | United States |
| Tres Rios | H2 | Arizona | United States |
| Tres Rios | R1 | Arizona | United States |
| Tres Rios | R2 | Arizona | United States |
| Tres Rios | R3 | Arizona | United States |
| Tres Rios | R4 | Arizona | United States |
| Tres Rios | R5 | Arizona | United States |
| Tres Rios | R6 | Arizona | United States |

**TABLE A.1 (CONTINUED)**
**Free Water Surface Wetlands**

| FWS System Name | Identifier | State/Province | Country |
|---|---|---|---|
| Tres Rios | R7 | Arizona | United States |
| Tres Rios | R8 | Arizona | United States |
| Tres Rios | R9 | Arizona | United States |
| Tres Rios | R10 | Arizona | United States |
| Tres Rios | R11 | Arizona | United States |
| Tres Rios | R12 | Arizona | United States |
| Tucson | Sweetwater | Arizona | United States |
| Arcata Enhancement | Gearheart | California | United States |
| Arcata Enhancement | Allen | California | United States |
| Arcata Enhancement | Hauser | California | United States |
| Arcata Treatment | | California | United States |
| Brawley | | California | United States |
| Corcoran | | California | United States |
| DUST | | California | United States |
| Gustine | 1A | California | United States |
| Gustine | 1B | California | United States |
| Gustine | 1C | California | United States |
| Gustine | 1D | California | United States |
| Gustine | 2A | California | United States |
| Gustine | 2B | California | United States |
| Gustine | 6D | California | United States |
| Hayward | | California | United States |
| Hemet | | California | United States |
| Imperial | | California | United States |
| Irvine Ranch | | California | United States |
| Lake Tahoe | | California | United States |
| Mountain View Sanitary District | | California | United States |
| Oxnard | | California | United States |
| Prado | | California | United States |
| Richmond | | California | United States |
| Sacramento | 1 | California | United States |
| Sacramento | 2 | California | United States |
| Sacramento | 3 | California | United States |
| Sacramento | 4 | California | United States |
| Sacramento | 6 A & B | California | United States |
| Sacramento | 7 | California | United States |
| Sacramento | 8 | California | United States |
| Sacramento | 9 | California | United States |
| Sacramento | 10 | California | United States |
| San Diego | | California | United States |
| Santa Rosa | Kelly Farm | California | United States |
| Avondale | | Colorado | United States |
| Bennett | | Colorado | United States |
| Crowley | | Colorado | United States |
| Crowley Correctional Facility | | Colorado | United States |
| Delta | | Colorado | United States |
| Dove Creek | | Colorado | United States |
| La Veta | | Colorado | United States |
| Manzanola | | Colorado | United States |
| Ouray | | Colorado | United States |
| Platteville | | Colorado | United States |
| Shop Creek | | Colorado | United States |
| Silt | | Colorado | United States |

(*Continued*)

**TABLE A.1 (CONTINUED)**
**Free Water Surface Wetlands**

| FWS System Name | Identifier | State/Province | Country |
|---|---|---|---|
| University of Connecticut | T: 1–3 | Connecticut | United States |
| Apalachicola | | Florida | United States |
| Apopka | | Florida | United States |
| Boney Marsh | | Florida | United States |
| Boot | | Florida | United States |
| Champion | A | Florida | United States |
| Champion | B | Florida | United States |
| Champion | C | Florida | United States |
| Champion | D | Florida | United States |
| Champion | E | Florida | United States |
| Champion | F | Florida | United States |
| Clermont | Plot H | Florida | United States |
| Deer Park | 1A | Florida | United States |
| Escambia County | | Florida | United States |
| Everglades Full Scale | STA2 | Florida | United States |
| Everglades Full Scale | STA1E | Florida | United States |
| Everglades Full Scale | STA1W | Florida | United States |
| Everglades Full Scale | STA3/4 | Florida | United States |
| Everglades Full Scale | STA 5 | Florida | United States |
| Everglades Full Scale | STA6 | Florida | United States |
| Everglades Nutrient Removal Project | ENRP | Florida | United States |
| Everglades Nutrient Removal Project | STC1 | Florida | United States |
| Everglades Nutrient Removal Project | STC2 | Florida | United States |
| Everglades Nutrient Removal Project | STC3 | Florida | United States |
| Everglades Nutrient Removal Project | STC4 | Florida | United States |
| Everglades Nutrient Removal Project | STC5 | Florida | United States |
| Everglades Nutrient Removal Project | STC6 | Florida | United States |
| Everglades Nutrient Removal Project | STC7 | Florida | United States |
| Everglades Nutrient Removal Project | STC8 | Florida | United States |
| Everglades Nutrient Removal Project | STC9 | Florida | United States |
| Everglades Nutrient Removal Project | STC10 | Florida | United States |
| Everglades Nutrient Removal Project | STC11 | Florida | United States |
| Everglades Nutrient Removal Project | STC12 | Florida | United States |
| Everglades Nutrient Removal Project | STC13 | Florida | United States |
| Everglades Nutrient Removal Project | STC14 | Florida | United States |
| Everglades Nutrient Removal Project | STC15 | Florida | United States |
| Everglades Nutrient Removal Project | NTC1 | Florida | United States |
| Everglades Nutrient Removal Project | NTC2 | Florida | United States |
| Everglades Nutrient Removal Project | NTC3 | Florida | United States |
| Everglades Nutrient Removal Project | NTC4 | Florida | United States |
| Everglades Nutrient Removal Project | NTC5 | Florida | United States |
| Everglades Nutrient Removal Project | NTC6 | Florida | United States |
| Everglades Nutrient Removal Project | NTC7 | Florida | United States |
| Everglades Nutrient Removal Project | NTC8 | Florida | United States |
| Everglades Nutrient Removal Project | NTC9 | Florida | United States |
| Everglades Nutrient Removal Project | NTC10 | Florida | United States |
| Everglades Nutrient Removal Project | NTC11 | Florida | United States |
| Everglades Nutrient Removal Project | NTC12 | Florida | United States |
| Everglades Nutrient Removal Project | NTC13 | Florida | United States |
| Everglades Nutrient Removal Project | NTC14 | Florida | United States |
| Everglades Nutrient Removal Project | NTC15 | Florida | United States |
| Everglades Periphyton | F1 | Florida | United States |
| Everglades Periphyton | F2 | Florida | United States |
| Everglades Periphyton | F3 | Florida | United States |

**TABLE A.1 (CONTINUED)**
**Free Water Surface Wetlands**

| FWS System Name | Identifier | State/Province | Country |
|---|---|---|---|
| Everglades Periphyton | F4 | Florida | United States |
| Gainesville | | Florida | United States |
| Greenwood | | Florida | United States |
| Hidden River | | Florida | United States |
| Lakeland | 1 | Florida | United States |
| Lakeland | 2 | Florida | United States |
| Lakeland | 3 | Florida | United States |
| Lakeland | 4 | Florida | United States |
| Lakeland | 5 | Florida | United States |
| Lakeland | 6 | Florida | United States |
| Lakeland | 7 | Florida | United States |
| Orange County Eastern Service Area | 1 | Florida | United States |
| Orange County Eastern Service Area | 2 | Florida | United States |
| Orlando Easterly | Stratum 1 | Florida | United States |
| Orlando Easterly | Stratum 2 | Florida | United States |
| Orlando Easterly | Stratum 3 | Florida | United States |
| Orlando Easterly | Stratum 4 | Florida | United States |
| Orlando Easterly Wetland | Stratum 1 | Florida | United States |
| Orlando Easterly Wetland | Stratum 2 | Florida | United States |
| Orlando Easterly Wetland | Stratum 3 | Florida | United States |
| Orlando Easterly Wetland | Stratum 4 | Florida | United States |
| Palm Beach County | | Florida | United States |
| Poinciana | T:1 | Florida | United States |
| Reedy Creek OFWTS | 2 | Florida | United States |
| Reedy Creek WTS1 | 1 | Florida | United States |
| Sanford | | Florida | United States |
| Silver Springs Shores | Lake Coral | Florida | United States |
| Tampa Office | | Florida | United States |
| Titusville | | Florida | United States |
| Waldo | | Florida | United States |
| Augusta | | Georgia | United States |
| Clayton County | | Georgia | United States |
| McMichael Dairy | | Georgia | United States |
| Cataldo | | Idaho | United States |
| Idaho Springs | Cell A | Idaho | United States |
| Des Plaines | 3 | Illinois | United States |
| Des Plaines | 4 | Illinois | United States |
| Des Plaines | 5 | Illinois | United States |
| Des Plaines | 6 | Illinois | United States |
| Embarrass River | A | Illinois | United States |
| Embarrass River | B | Illinois | United States |
| Embarrass River | D | Illinois | United States |
| Norwood Farms | | Indiana | United States |
| Purdue University | A1 | Indiana | United States |
| Purdue University | A2 | Indiana | United States |
| Purdue University | A3 | Indiana | United States |
| Purdue University | A4 | Indiana | United States |
| Purdue University | B1 | Indiana | United States |
| Purdue University | B2 | Indiana | United States |
| Purdue University | B3 | Indiana | United States |
| Purdue University | B4 | Indiana | United States |
| Purdue University | C1 | Indiana | United States |
| Purdue University | C2 | Indiana | United States |

(*Continued*)

**TABLE A.1 (CONTINUED)**
**Free Water Surface Wetlands**

| FWS System Name | Identifier | State/Province | Country |
|---|---|---|---|
| Purdue University | C3 | Indiana | United States |
| Purdue University | C4 | Indiana | United States |
| Purdue University | D1 | Indiana | United States |
| Purdue University | D3 | Indiana | United States |
| Purdue University | D4 | Indiana | United States |
| Tom Brothers Farm | | Indiana | United States |
| Army Ammunition Plant | | Iowa | United States |
| Brenton Cattle | | Iowa | United States |
| Norwalk | | Iowa | United States |
| Norwalk | | Iowa | United States |
| Benton | 1 Cattail | Kentucky | United States |
| Benton | 2 Woolgrass | Kentucky | United States |
| Crittenden County | | Kentucky | United States |
| Mercer County | | Kentucky | United States |
| Pembroke | | Kentucky | United States |
| Russelville | | Kentucky | United States |
| Union County | | Kentucky | United States |
| Norco | | Louisiana | United States |
| Saint Louis University | 1 | Louisiana | United States |
| Saint Louis University | 2 | Louisiana | United States |
| Saint Louis University | 3 | Louisiana | United States |
| University of SW Louisiana | 1 | Louisiana | United States |
| University of SW Louisiana | 2 | Louisiana | United States |
| University of SW Louisiana | 3 | Louisiana | United States |
| Braun | Braun | Maryland | United States |
| Eastern Shore | B1 | Maryland | United States |
| Eastern Shore | B2 | Maryland | United States |
| Foster | Foster | Maryland | United States |
| Mays Chapel | | Maryland | United States |
| Queen Anne | | Maryland | United States |
| Great Meadows | | Massachusetts | United States |
| Ann Arbor | | Michigan | United States |
| Bellaire | | Michigan | United States |
| Commerce Township | | Michigan | United States |
| Houghton Lake | 1 | Michigan | United States |
| Houghton Lake | 2 | Michigan | United States |
| Kinross (Kincheloe) | 1 | Michigan | United States |
| Lansing | | Michigan | United States |
| Lapeer | | Michigan | United States |
| Onaway | | Michigan | United States |
| Vermontville | | Michigan | United States |
| Biwabik | | Minnesota | United States |
| Clear Lake | | Minnestoa | United States |
| Dunka Mine | | Minnestoa | United States |
| Isanti-Chisago | | Minnestoa | United States |
| Lake McCarrons | | Minnestoa | United States |
| Spring Lake | | Minnestoa | United States |
| Hernando | 1 | Mississippi | United States |
| Hernando | 2 | Mississippi | United States |
| Hernando | 3 | Mississippi | United States |
| Hernando | 4 | Mississippi | United States |
| Leaf River | Pond 1 | Mississippi | United States |
| Leaf River | Pond 2 | Mississippi | United States |
| Leaf River | Pond 3 | Mississippi | United States |

**TABLE A.1 (CONTINUED)**
**Free Water Surface Wetlands**

| FWS System Name | Identifier | State/Province | Country |
|---|---|---|---|
| Newton | 1 | Mississippi | United States |
| Newton | 2 | Mississippi | United States |
| Newton | 3 | Mississippi | United States |
| Newton | 4 | Mississippi | United States |
| Newton | 5 | Mississippi | United States |
| Newton | 6 | Mississippi | United States |
| Pontotoc | 1 | Mississippi | United States |
| Pontotoc | 2 | Mississippi | United States |
| West Jackson County | 1–2 | Mississippi | United States |
| West Jackson County | T:1–7 | Mississippi | United States |
| Weyerhauser | | Mississippi | United States |
| Columbia | | Missouri | United States |
| Butte | | Montana | United States |
| Incline Village | 1 | Nevada | United States |
| Incline Village | 2 | Nevada | United States |
| Incline Village | 3 | Nevada | United States |
| Incline Village | 4 | Nevada | United States |
| Incline Village | 5 | Nevada | United States |
| Incline Village | 6 | Nevada | United States |
| Incline Village | 7 | Nevada | United States |
| Brookhaven | Meadow Marsh Pond System | New York | United States |
| Irondequoit Creek | | New York | United States |
| Monroe County | | New York | United States |
| Moodna Basin | | New York | United States |
| Seneca Army Depot | | New York | United States |
| Duplin County | 1 | North Carolina | United States |
| Duplin County | 2 | North Carolina | United States |
| Duplin County | 3 | North Carolina | United States |
| Duplin County | 4 | North Carolina | United States |
| Greensboro | | North Carolina | United States |
| Herrings | | North Carolina | United States |
| Hilton Head Plantation | Whooping Crane | North Carolina | United States |
| New Hanover Township | Raw | North Carolina | United States |
| New Hanover Township | Treated | North Carolina | United States |
| Sea Pines | Boggy Gut | North Carolina | United States |
| Hillsboro | | North Dakota | United States |
| Mandan | | North Dakota | United States |
| Minot | | North Dakota | United States |
| Franklin County | | Ohio | United States |
| Franklin County | | Ohio | United States |
| Lick Run | | Ohio | United States |
| Olentangy | 1 | Ohio | United States |
| Olentangy | 2 | Ohio | United States |
| Cannon Beach | | Oregon | United States |
| Hillsboro | Jackson Bottoms 1–17 | Oregon | United States |
| Mt. Angel | | Oregon | United States |
| Oregon State University | 1 | Oregon | United States |
| Oregon State University | 2 | Oregon | United States |
| Oregon State University | 3 | Oregon | United States |
| Oregon State University | 4 | Oregon | United States |
| Oregon State University | 5 | Oregon | United States |
| Oregon State University | 6 | Oregon | United States |
| Silverton | | Oregon | United States |

(*Continued*)

**TABLE A.1 (CONTINUED)**
**Free Water Surface Wetlands**

| FWS System Name | Identifier | State/Province | Country |
|---|---|---|---|
| Conestoga | | Pennsylvania | United States |
| Cumberland County | | Pennsylvania | United States |
| Friendship Hill | | Pennsylvania | United States |
| Iselin | | Pennsylvania | United States |
| Neshaminy | | Pennsylvania | United States |
| Palmerton | | Pennsylvania | United States |
| Springdale | | Pennsylvania | United States |
| Central Slough | T:1 | South Carolina | United States |
| Savannah River | | South Carolina | United States |
| Vereen | Bear Bay 1 | South Carolina | United States |
| Arlington | | South Dakota | United States |
| Alcoa | Blue Pond | Tennessee | United States |
| Alcoa | Duck Spring | Tennessee | United States |
| Milan | | Tennessee | United States |
| Beaumont | | Texas | United States |
| Tarrant | 1 | Texas | United States |
| Tarrant | 2 | Texas | United States |
| Tarrant | 3 | Texas | United States |
| Tarrant | 4 | Texas | United States |
| Tarrant | 5 | Texas | United States |
| Tarrant | 6 | Texas | United States |
| Tarrant | 7 | Texas | United States |
| Tarrant | 8 | Texas | United States |
| Tarrant | 9 | Texas | United States |
| Victoria | | Texas | United States |
| Crestwood | | Virginia | United States |
| Franklin Farms | | Virginia | United States |
| Bellevue | | Washington | United States |
| Connell | Pilot | Washington | United States |
| Connell | Full | Washington | United States |
| Renton | | Washington | United States |
| Albright | | West Virginia | United States |
| Brillion | | Wisconsin | United States |
| David Gerrits Farm | 1 & 2 | Wisconsin | United States |
| David Gerrits Farm | 3 & 4 | Wisconsin | United States |
| Drummond | | Wisconsin | United States |
| Lake Nebagamon | | Wisconsin | United States |

**TABLE A.2**
**Horizontal Subsurface Flow Wetlands**

| HSSF System Name | Identifier | State/Province | Country |
|---|---|---|---|
| Lismore | GB1 | New South Wales | Australia |
| Lismore | GB2 | New South Wales | Australia |
| Richmond | Gravel | New South Wales | Australia |
| Richmond | Mix A | New South Wales | Australia |
| Richmond | Mix B | New South Wales | Australia |
| Richmond | Schoeno | New South Wales | Australia |
| Richmond | *Typha* | New South Wales | Australia |
| Brisbane | A | Queensland | Australia |
| Brisbane | B | Queensland | Australia |
| Brisbane | C | Queensland | Australia |

**TABLE A.2 (CONTINUED)**
**Horizontal Subsurface Flow Wetlands**

| HSSF System Name | Identifier | State/Province | Country |
|---|---|---|---|
| Brisbane | D | Queensland | Australia |
| Alstonville | | | Australia |
| Edmonton | | Alberta | Canada |
| Princeport | | Nova Scotia | Canada |
| Monteverde | 1 | | Costa Rica |
| Monteverde | 2 | | Costa Rica |
| Brehov | | | Czech Republic |
| Chmelna | | | Czech Republic |
| Doksy | | | Czech Republic |
| Kacice | | | Czech Republic |
| Koberovy | | | Czech Republic |
| Kolodeje | | | Czech Republic |
| Krucemburk | | | Czech Republic |
| Liszny | | | Czech Republic |
| Morina | | | Czech Republic |
| Nucice | | | Czech Republic |
| Ondrejov | | | Czech Republic |
| Onsov | | | Czech Republic |
| Osova Bityyska | | | Czech Republic |
| Velescice | | | Czech Republic |
| Zintenice | | | Czech Republic |
| Abu Attwa | | | Egypt |
| Ismalia | | | Egypt |
| Brandenburg | | | Germany |
| Jülich | *Phragmites* | | Germany |
| Jülich | *Typha* | | Germany |
| Langenreichenbach | | | Germany |
| Wiedersburg | | | Germany |
| Thrace | MG-Z | | Greece |
| Thrace | MG-C | | Greece |
| Thrace | MG-R | | Greece |
| Sakhnin | | | Israel |
| Cecchi | | | Italy |
| Chiampo Valley | *Phragmites* | | Italy |
| Chiampo Valley | Carex | | Italy |
| Florence | | | Italy |
| La Croce | | | Italy |
| Ornellaia | | | Italy |
| Putignano | | | Italy |
| San Michele de Ganzaria | | | Italy |
| Akumal | | | Mexico |
| Pachuaca | | | Mexico |
| UNAM | | | Mexico |
| Rabat | | | Morocco |
| Drury | | | New Zealand |
| Hamilton | 1 Bulrush | | New Zealand |
| Hamilton | 1 Unplanted | | New Zealand |
| Hamilton | 2 Bulrush | | New Zealand |
| Hamilton | 2 Unplanted | | New Zealand |
| Hamilton | 3 Bulrush | | New Zealand |
| Hamilton | 3 Unplanted | | New Zealand |
| Hamilton | 4 Bulrush | | New Zealand |
| Hamilton | 4 Unplanted | | New Zealand |

(*Continued*)

**TABLE A.2 (CONTINUED)**
**Horizontal Subsurface Flow Wetlands**

| HSSF System Name | Identifier | State/Province | Country |
|---|---|---|---|
| Hamilton | 5 Bulrush | | New Zealand |
| Hamilton | Dairy 1 | | New Zealand |
| Hamilton | Dairy 2 | | New Zealand |
| Hamilton | Dairy 3 | | New Zealand |
| Hamilton | Dairy 4 | | New Zealand |
| Hamilton | Dairy 5 | | New Zealand |
| Hamilton | Meat 1 | | New Zealand |
| Hamilton | Meat 2 | | New Zealand |
| Hamilton | Meat 3 | | New Zealand |
| Hamilton | M1 | | New Zealand |
| Hamilton | D1 | | New Zealand |
| Hamilton | D2 | | New Zealand |
| Hamilton | D2A | | New Zealand |
| Omaha | | | New Zealand |
| Opononi | | | New Zealand |
| Waipoua | | | New Zealand |
| Fagerness | | | Norway |
| Haugstein | | | Norway |
| Jølle | | | Norway |
| Jordforsk | | | Norway |
| Ostegarden | | | Norway |
| Stange | | | Norway |
| Tveter | 1 | | Norway |
| Tveter | 2 | | Norway |
| Golczew | | | Poland |
| Gralewo | | | Poland |
| Gronowo | | | Poland |
| Lubiejewo | | | Poland |
| Nieskorz | | | Poland |
| Nieskorz | | | Poland |
| Rokitno | | | Poland |
| Ryjewo | | | Poland |
| Mpophomeni | | | South Africa |
| Pretoria | 1 | | South Africa |
| Pretoria | 2 | | South Africa |
| Pretoria | 3 | | South Africa |
| Barcelona | A1 | | Spain |
| Barcelona | A2 | | Spain |
| Barcelona | B1 | | Spain |
| Barcelona | B2 | | Spain |
| Barcelona | C1 | | Spain |
| Barcelona | C2 | | Spain |
| Barcelona | D1 | | Spain |
| Barcelona | D2 | | Spain |
| Dar es Salaam | 1 | | Tanzania |
| Dar es Salaam | 2 | | Tanzania |
| Chiang Mai | | | Thailand |
| Pathumthani | | | Thailand |
| Acle | | | United Kingdom |
| Ashby Folville | | | United Kingdom |
| Audlem | 1 | | United Kingdom |
| Audlem | 2 | | United Kingdom |
| Audlem | 3 | | United Kingdom |
| Audlem | 4 | | United Kingdom |

**TABLE A.2 (CONTINUED)**
**Horizontal Subsurface Flow Wetlands**

| HSSF System Name | Identifier | State/Province | Country |
|---|---|---|---|
| Audlem | 5 | | United Kingdom |
| Audlem | 6 | | United Kingdom |
| Audlem | 7 | | United Kingdom |
| Audlem | 8 | | United Kingdom |
| Audlem | 9 | | United Kingdom |
| Audlem | 10 | | United Kingdom |
| Broxted | Lower | | United Kingdom |
| Broxted | Upper | | United Kingdom |
| Budds Farm | | | United Kingdom |
| Castleroe | 1 | | United Kingdom |
| Castleroe | 2 | | United Kingdom |
| Castleroe | 3 | | United Kingdom |
| Castleroe | 4 | | United Kingdom |
| Castleroe | Peat bed | | United Kingdom |
| Cherry Burton | | | United Kingdom |
| Claverley | | | United Kingdom |
| Culcrow | 1 | | United Kingdom |
| Ditton Priors | | | United Kingdom |
| Drointon | | | United Kingdom |
| Earlswood | | | United Kingdom |
| Earlswood | | | United Kingdom |
| East Haddon | 2 | | United Kingdom |
| East Haddon | | | United Kingdom |
| Forton | | | United Kingdom |
| Freethorpe | | | United Kingdom |
| Gailey | | | United Kingdom |
| Gravesend | 1 | | United Kingdom |
| Gravesend | 2 | | United Kingdom |
| Gravesend | 3 | | United Kingdom |
| Himley | 1 | | United Kingdom |
| Himley | 2 | | United Kingdom |
| Hognaston | | | United Kingdom |
| Holtby | | | United Kingdom |
| Kingston And Madley | | | United Kingdom |
| Kirmington | | | United Kingdom |
| Knowbury | | | United Kingdom |
| Leek Wootton | | | United Kingdom |
| Lighthorne Heath | | | United Kingdom |
| Little Stretton | | | United Kingdom |
| Little Wenlock | | | United Kingdom |
| Lydbury North (Old) | | | United Kingdom |
| Malham | | | United Kingdom |
| Meltham | | | United Kingdom |
| Middleton (Shropshire) | | | United Kingdom |
| Middleton (Warwickshire) | | | United Kingdom |
| Naseby | | | United Kingdom |
| Needingworth | | | United Kingdom |
| Norton Lindsey | | | United Kingdom |
| Nun Monkton | | | United Kingdom |
| Oaklands Park | 1 | | United Kingdom |
| Oaklands Park | 2 | | United Kingdom |
| Rugeley | | | United Kingdom |
| Saxby | | | United Kingdom |

(*Continued*)

**TABLE A.2 (CONTINUED)**
**Horizontal Subsurface Flow Wetlands**

| HSSF System Name | Identifier | State/Province | Country |
|---|---|---|---|
| Southwick | 2 | | United Kingdom |
| Southwick | 3 | | United Kingdom |
| Stockley | 1 | | United Kingdom |
| Stockley | 2 | | United Kingdom |
| Stockley | 3 | | United Kingdom |
| Thorpe Satchville | | | United Kingdom |
| Valleyfield | 1 | | United Kingdom |
| Valleyfield | 2 | | United Kingdom |
| Valleyfield | 3 | | United Kingdom |
| Valleyfield | 4 | | United Kingdom |
| Wentworth Castle | 1 | | United Kingdom |
| Wentworth Castle | 2 | | United Kingdom |
| Wentworth Castle | 3 | | United Kingdom |
| Wentworth Castle | 4 | | United Kingdom |
| West Harwood | | | United Kingdom |
| Westow | 1 | | United Kingdom |
| Westow | 2 | | United Kingdom |
| Wetwood | | | United Kingdom |
| Alabama | 1 | Alabama | United States |
| Alabama | 2 | Alabama | United States |
| Alabama | 3 | Alabama | United States |
| Alabama | 4 | Alabama | United States |
| Degussa Corporation | | Alabama | United States |
| Phillips High School | | Alabama | United States |
| Pima County | | Arizona | United States |
| Sahuarita | | Arizona | United States |
| Tubac | | Arizona | United States |
| Wickenburg | | Arizona | United States |
| Clarendon | | Arkansas | United States |
| Dierks | | Arkansas | United States |
| Foreman | | Arkansas | United States |
| Gould | | Arkansas | United States |
| Gurndon | | Arkansas | United States |
| Judsonia | | Arkansas | United States |
| Lewisville | | Arkansas | United States |
| North Crossett | | Arkansas | United States |
| Ola | | Arkansas | United States |
| Pocahontas | | Arkansas | United States |
| Rector | 1 | Arkansas | United States |
| Rector | 2 | Arkansas | United States |
| Smackover | | Arkansas | United States |
| Tuckema | | Arkansas | United States |
| Waldo | | Arkansas | United States |
| Hopland | | California | United States |
| Oxnard | Train 1 | California | United States |
| Oxnard | Train 2 | California | United States |
| Sacramento | 11 | California | United States |
| Santee | Gravel | California | United States |
| Santee | Cattail | California | United States |
| Santee | Bulrush | California | United States |
| Santee | Phragmites | California | United States |
| Calahan | | Colorado | United States |

**TABLE A.2 (CONTINUED)**
**Horizontal Subsurface Flow Wetlands**

| HSSF System Name | Identifier | State/Province | Country |
|---|---|---|---|
| Highland Acres W&S District | | Colorado | United States |
| Las Animas | | Colorado | United States |
| Fish Royer | | Indiana | United States |
| Anamosa | | Iowa | United States |
| Beery | | Kansas | United States |
| Brown | | Kansas | United States |
| Carlson | | Kansas | United States |
| Manhattan | PFL | Kansas | United States |
| Manhattan | PCL | Kansas | United States |
| Manhattan | PCR | Kansas | United States |
| Manhattan | UFL | Kansas | United States |
| Manhattan | UCL | Kansas | United States |
| Manhattan | UCR | Kansas | United States |
| Anderson | | Kentucky | United States |
| Anderson County | | Kentucky | United States |
| Benton | 3 | Kentucky | United States |
| Greenville | 1 | Kentucky | United States |
| Greenville | 2 | Kentucky | United States |
| Greenville | 3 | Kentucky | United States |
| Greenville | 4 | Kentucky | United States |
| Greenville | 5 | Kentucky | United States |
| Hardin | 1 | Kentucky | United States |
| Hardin | 2 | Kentucky | United States |
| Paris | | Kentucky | United States |
| Scott | | Kentucky | United States |
| Denham Springs | 1 | Louisiana | United States |
| Denham Springs | 2 | Louisiana | United States |
| Denham Springs | 3 | Louisiana | United States |
| Greenleaves Subdivision | | Louisiana | United States |
| Haughton | | Louisiana | United States |
| Mandeville | | Louisiana | United States |
| Keyser's Ridge | | Maryland | United States |
| Mayo Peninsula | | Maryland | United States |
| Cambridge-Isanti School District | | Minnesota | United States |
| Carlson Residence | | Minnesota | United States |
| Cedar Mills | | Minnesota | United States |
| Chisholm | | Minnesota | United States |
| Clearwater Forest | | Minnesota | United States |
| Darfur | | Minnesota | United States |
| Delft | | Minnesota | United States |
| Eichten Cheese | | Minnesota | United States |
| Fields of St. Croix - North | | Minnesota | United States |
| Grand Lake | | Minnesota | United States |
| Happiness Resort | | Minnesota | United States |
| Jackson Meadow | South | Minnesota | United States |
| Jackson Meadow | North | Minnesota | United States |
| Lake Elmo | | Minnesota | United States |
| Lakes of Fairhaven | | Minnesota | United States |
| Lake Washington | | Minnesota | United States |
| Lutsen East Flank | | Minnesota | United States |
| Lutsen Sea Villas | | Minnesota | United States |
| Morton Farm Preserve | | Minnesota | United States |
| Mulberry Meadows | | Minnesota | United States |

(*Continued*)

**TABLE A.2 (CONTINUED)**
**Horizontal Subsurface Flow Wetlands**

| HSSF System Name | Identifier | State/Province | Country |
|---|---|---|---|
| NAWE Office | | Minnesota | United States |
| NERCC | 1 | Minnesota | United States |
| NERCC | 2 | Minnesota | United States |
| The Ponds | | Minnesota | United States |
| The Preserve | | Minnesota | United States |
| Northern Tier High Adventure Base | | Minnesota | United States |
| Opole | | Minnesota | United States |
| Prairie Hamlet | | Minnesota | United States |
| Prinsburg | | Minnesota | United States |
| Spring Hill | | Minnesota | United States |
| St. George | | Minnesota | United States |
| Super America - Ham Lake | | Minnesota | United States |
| Super America - Wyoming | | Minnesota | United States |
| Tamarack | | Minnesota | United States |
| Tamarack Farm Estates | | Minnesota | United States |
| Wildflower Shores | | Minnesota | United States |
| Columbus | | Mississippi | United States |
| Utica, North | 1 | Mississippi | United States |
| Utica, North | 2 | Mississippi | United States |
| Utica, South | 1 | Mississippi | United States |
| Utica, South | 2 | Mississippi | United States |
| Lake Capri | Vegetated | Missouri | United States |
| Lake Capri | Unvegetated | Missouri | United States |
| Firethorn | | Nebraska | United States |
| Lincoln | | Nebraska | United States |
| Albuquerque | | New Mexico | United States |
| Minoa | 1 | New York | United States |
| Minoa | 2 | New York | United States |
| Minoa | 3 | New York | United States |
| Craven County | | North Carolina | United States |
| North Carolina Ranger Station | | North Carolina | United States |
| North Carolina Residence | 1 | North Carolina | United States |
| North Carolina Residence | 2 | North Carolina | United States |
| Ohio Residences | 13 | Ohio | United States |
| Ohio Residences | 14 | Ohio | United States |
| Ohio Residences | 15 | Ohio | United States |
| Ohio Residences | 16 | Ohio | United States |
| Ohio Residences | 17 | Ohio | United States |
| Ohio Residences | 19 | Ohio | United States |
| Ohio Residences | 20 | Ohio | United States |
| Ohio Residences | 21 | Ohio | United States |
| Halsey | | Oregon | United States |
| Iselin | | Pennsylvania | United States |
| Baxter | A | Tennessee | United States |
| Baxter | B | Tennessee | United States |
| Baxter | C | Tennessee | United States |
| Baxter | D | Tennessee | United States |
| Baxter | E | Tennessee | United States |
| Baxter | F | Tennessee | United States |
| Baxter | G | Tennessee | United States |
| Baxter | H | Tennessee | United States |
| Baxter | I | Tennessee | United States |
| Baxter | J | Tennessee | United States |

**TABLE A.2 (CONTINUED)**
**Horizontal Subsurface Flow Wetlands**

| HSSF System Name | Identifier | State/Province | Country |
|---|---|---|---|
| Baxter | K | Tennessee | United States |
| Baxter | L | Tennessee | United States |
| Baxter | M | Tennessee | United States |
| Baxter | N | Tennessee | United States |
| Kingston Power Plant | | Tennessee | United States |
| Bryan | | Texas | United States |
| College Station | | Texas | United States |
| Dessau Mobile Home Park | | Texas | United States |
| D'Harris | | Texas | United States |
| Dublin | | Texas | United States |
| Houston | | Texas | United States |
| Johnson City | 1 | Texas | United States |
| Johnson City | 2 | Texas | United States |
| Stephenville | | Texas | United States |
| Tomball | | Texas | United States |
| Weslaco | | Texas | United States |
| Kentland | 1 | Virginia | United States |
| Kentland | 2 | Virginia | United States |
| Kentland | 3 | Virginia | United States |
| Kentland | CT | Virginia | United States |
| Kentland | WG | Virginia | United States |
| Monterey | | Virginia | United States |
| Monterrey | | Virginia | United States |
| Powell | 1 | Virginia | United States |
| Powell | 2 | Virginia | United States |
| Powell | 3 | Virginia | United States |
| Powell | CT | Virginia | United States |
| Powell | WG | Virginia | United States |
| Dungeness | | Washington | United States |
| Morgantown | | West Virginia | United States |
| St. Croix Chippewa | | Wisconsin | United States |

**TABLE A.3**
**Vertical Flow Wetlands**

| VF System Name | Identifier | State/Province | Country |
|---|---|---|---|
| Ernsthofen | | | Austria |
| Gleisdorf (Binder) | | | Austria |
| Hollerberg-West | | | Austria |
| Horbach | | | Austria |
| Lappach (Kurzmann) | | | Austria |
| Wolfern | | | Austria |
| YX | | | Austria |
| Strengberg | | | Austria |
| Birkfeld (Hoeller) | | | Austria |
| Lasnitzhöhe (Laposa) | | | Austria |
| Lang Fr. | | | Austria |
| Jennersdorf (Doppelhofer) | | | Austria |
| Pischelsdorf (Hart) | | | Austria |
| Hart b. Graz (Koberg) | | | Austria |
| Zeutschach (Hofer) | | | Austria |

(*Continued*)

**TABLE A.3 (CONTINUED)**
**Vertical Flow Wetlands**

| VF System Name | Identifier | State/Province | Country |
|---|---|---|---|
| Bokrijk | | | Belgium |
| PDLT | | | Belgium |
| PIME | | | Belgium |
| Sint-Niklass Heimolen | | | Belgium |
| System A | | | Belgium |
| System B | | | Belgium |
| System C | | | Belgium |
| System D | | | Belgium |
| System E | | | Belgium |
| Zemst-Larbeek | | | Belgium |
| Blenterin | | Ontario | Canada |
| Vinland | | Ontario | Canada |
| Ansaster | | | Canada |
| Saint Ann's | | | Canada |
| Saint Andrés | | | Colombia |
| Bijar Cres | | | Croatia |
| Aarhus | | | Denmark |
| Paistu | | | Estonia |
| Roussillon | | | France |
| Saint Thomé | | | France |
| Langenreichenbach | | | Germany |
| Leipzig | | | Germany |
| Casolara Acqua Chiara | | | Italy |
| Cerbaia | | | Italy |
| Cerbaia | | | Italy |
| Contrada Petrosa | | | Italy |
| Firenze, Hotel RC | | | Italy |
| Florence | | | Italy |
| Gorizia | | | Italy |
| La Collina | | | Italy |
| Dhulikhel | | | Nepal |
| Adema | | | Netherlands |
| Appelscha | | | Netherlands |
| Boxtel | | | Netherlands |
| Deurna | | | Netherlands |
| Groot Ehrental | | | Netherlands |
| Hamming | | | Netherlands |
| Heislum | | | Netherlands |
| Hobbitstee | | | Netherlands |
| Klein Profijt | | | Netherlands |
| Koudum | | | Netherlands |
| Lauersoog | | | Netherlands |
| Lemmer | | | Netherlands |
| Nijkerk | | | Netherlands |
| Nooyen | | | Netherlands |
| Onnes | | | Netherlands |
| Oosthem | | | Netherlands |
| Oud-Beijerland | | | Netherlands |
| Oudega | | | Netherlands |
| Spijkerman | | | Netherlands |
| ter Veer | | | Netherlands |
| Terpstra | | | Netherlands |
| van Oirschot | | | Netherlands |

**TABLE A.3**
**Vertical Flow Wetlands**

| VF System Name | Identifier | State/Province | Country |
|---|---|---|---|
| van Ravenhorst | | | Netherlands |
| Veening | | | Netherlands |
| Vrieling | | | Netherlands |
| Wapserveen | | | Netherlands |
| Fagernes | | | Norway |
| Haugstein | | | Norway |
| Jolle | | | Norway |
| Østergårde | | | Norway |
| Tveter I | | | Norway |
| Carion de los Céspedes | | | Spain |
| Mojacar | | | Spain |
| Castleroe | Gravel bed | | United Kingdom |
| Castleroe | Peat bed | | United Kingdom |
| Drointon | 1 | | United Kingdom |
| Drointon | 2 | | United Kingdom |
| Gailey | | | United Kingdom |
| Medmenham | 1 | | United Kingdom |
| Medmenham | 2 | | United Kingdom |
| Oaklands Park | 1 | | United Kingdom |
| Oaklands Park | 2 | | United Kingdom |
| Rugeley | 1 | | United Kingdom |
| Rugeley | 2 | | United Kingdom |
| Wills Barn | | | United Kingdom |
| LaGrange County | | Indiana | United States |
| North Carolina School | | North Carolina | United States |
| Laguna Niguel | | California | United States |
| Salem | Bed 1 | Oregon | United States |
| Salem | Bed 2 | Oregon | United States |
| Saginaw | | Michigan | United States |
| Connell | Pilot | Washington | United States |
| Connell | Full scale | Washington | United States |

# Appendix B: Tracer Testing and Flow-Pattern Modeling

The large majority of information on the contaminant removal capabilities of treatment wetlands has been in the form of a relatively continuous time series of inlet and outlet concentrations, under conditions of known flow. There is a second realm of data acquisition and analysis that revolves around the spike addition of substances to the wetland. A wide variety of substances have been used to trace the progress of water through treatment wetlands. These have included the salt ions lithium, bromide, chloride, iodide, and fluoride; the fluorescent dyes rhodamine RWT and B fluorescein; and tritiated water. Most often, these are pulse injected into the wetland inlet, and the concentration response is determined at the wetland outlet. The purpose of hydraulic tracer testing is to determine the distribution of detention times for the wetland.

Detention time distributions (DTDs) for treatment wetlands have been extensively investigated at many wetland sites, and thus there exist numerous examples of the functional forms that are characteristic of wetlands. Single-shot tracer injection with effluent concentration monitoring is usually employed (Kadlec and Knight, 1996). Because the tracer does not (theoretically) interact with wetland soils or biota, it serves as a marker of the water with which it enters. Typical distributions are bell shaped, with some tracer exiting at short times, and some exiting at longer times.

This evidence is conclusive: imperfect flow patterns pervade the universe of treatment wetlands. It is necessary to account for this in design, and it is accomplished via nonideal flow models. In turn, the understanding and development of nonideal flow models derive principally from tracer testing. The purpose of this appendix is to expand on the practicalities, pitfalls, and results of wetland tracer testing.

One of the principal results derived from tracer testing is presented in Chapter 2—the volumetric efficiency of the wetland, i.e., how much of the nominal wetland water volume is involved in its flow. It is the ratio of tracer detention time to nominal detention time: $e_V = \tau/\tau_n$.

Another principal result is the dimensionless variance of the outlet response, $\sigma_\theta^2$, which is briefly discussed in Chapter 6. This measure of the spread of the distribution of detention times may be used in a number of ways as an aid to pollutant removal performance. The dimensionless variance may be used to determine the parameters in DTD models, such as the tanks-in-series (TIS) model. Or, it may be used to compute the DTD efficiency of the wetland (Persson *et al.*, 1999):

$$e_{DTD} = \left(1 - \sigma_\theta^2\right) \tag{B.1}$$

This efficiency is unity for plug flow (PF), and zero for perfect mixing. As shown by Kadlec and Knight (1996), the DTD efficiency is an approximate interpolator between the performance of one TIS, as measured by $F_{1TIS}$, and PF performance, as measured by $F_{PF}$.

$$F = \frac{C - C^*}{C_i - C^*} = e_{DTD}\ F_{PF} + (1 - e_{DTD})\ F_{1TIS} \tag{B.2}$$

For plug flow,

$$F_{PF} = \frac{C_{PF} - C^*}{C_i - C^*} = \exp\left(\frac{-k_A \tau}{h}\right) \tag{B.3}$$

For *N*TIS,

$$F_{TIS} = \frac{C_{TIS} - C^*}{C_i - C^*} = \left(1 + \frac{k_A \tau}{hN}\right)^{-N} \tag{B.4}$$

where

$C$ = outlet concentration, mg/m$^3$
$C_i$ = average inlet concentration, mg/m$^3$
$C^*$ = background concentration, mg/m$^3$
$F$ = fraction remaining to background
$k_A$ = intrinsic reaction rate constant, m/d
$N$ = number of tanks
$\tau$ = detention time, d

The root-mean-square degree of fit of the approximation for Equation B.2 ranges 1–6% for $1 \le (k_A\tau/h) \le 3$.

Persson *et al.* (1999) go on to suggest a combined measure, the hydraulic efficiency ($\lambda$), that reflects the excluded volume and the mixing pattern of constructed wetlands. This measure is defined as the product of the volumetric efficiency and the DTD efficiency:

$$\lambda = e_V \cdot e_{DTD} \tag{B.5}$$

This is useful for ranking wetlands for their combined efficiency, but is not directly useful for parameter estimation. The two pieces are needed separately for quantitative estimates of performance. For a TIS model, the combined hydraulic efficiency is given by a very simple result:

$$\lambda = \frac{\tau_p}{\tau_n} \tag{B.6}$$

where

$\tau_p$ = tracer peak time
$\tau_n$ = nominal hydraulic detention time

Persson *et al*. (1999) have explored both real wetland systems and, via calibrated two-dimensional dynamic modeling, simulated situations. They found $0.11 < \lambda < 0.90$. The same parameter has been evaluated for other hypothetical wetland situations (Jenkins and Greenway, 2005).

## IDEAL FLOW REACTORS

Wetlands can be thought of as a cross section between two theoretically ideal reactors: the plug flow reactor and continuously-stirred tank reactor (CSTR). The plug flow (PF) reactor represents a situation where there is no internal mixing within the reactor, and water parcels move in unison from the inlet to the outlet. The CSTR represents the ideal of perfect mixing: water entering the system is instantaneously mixed throughout the reactor.

### Nominal Hydraulic Detention Time

PF and CSTR reactors behave very differently in response to the input of a conservative tracer. To discuss tracer behavior, it is useful to review the concepts of hydraulic detention time from Chapter 2. For a free water surface (FWS) wetland, the nominal wetland water volume is defined as the volume enclosed by the upper water surface, and the bottom and sides of the impoundment.

$$\tau_n = \frac{V_n}{Q} = \frac{(LWh)_n}{Q} \tag{B.7}$$

where

$\tau_n$ = nominal hydraulic detention time, days
$L$ = wetland length, m
$W$ = wetland width, m
$h$ = water depth, m
$LWh_n$ = nominal wetland volume, $m^3$
$Q$ = flow rate, $m^3/d$

For a subsurface flow (SSF) wetland, it is that enclosed volume multiplied by the porosity of the clean (unclogged) bed media.

$$\tau_n = \frac{V_n}{Q} = \frac{\varepsilon(LWh)_n}{Q} \tag{B.8}$$

where

$\varepsilon$ = bed media porosity, dimensionless

Dimensionless time, $\theta$, can be used instead of nominal hydraulic detention time, $\tau_n$, when comparing tracer response curves:

$$\theta = \frac{t}{\tau_n} \tag{B.9}$$

where

$t$ = elapsed time in days

### Tracer Response in PF and CSTR Reactors

A spike input of tracer entering a PF reactor will move through the system with zero mixing. As a result, the tracer spike will exit the reactor unchanged at $\tau_n$ ($\theta = 1$). In a CSTR reactor, the tracer impulse is instantaneously and uniformly distributed among the tank contents (Levenspiel, 1972). As flow continues to enter the tank, tracer-contaminated water is displaced, resulting in a declining tracer output curve with a long tail. Figure B.1 displays both types of ideal reactors and their associated tracer response curves.

### Real-World Tracer Movement

It is well documented that the flow patterns through treatment wetland systems are nonideal and do not conform to either the PF or CSTR ideals (see Tables 6.1 and 6.2, Chapter 6). In SSF wetland systems, dispersion and mixing occurs within the bed as water flows between the gravel particles or sand grains. In SSF wetlands, roots may create preferential flow paths near the bottom of the wetland cell (Liehr *et al*., 2000). In FWS wetlands, water near the surface is less subject to bottom drag and moves faster than the flow that is deeper in the water column. Water must detour around plant bases, which act as stagnant pockets that exchange water with adjacent flow channels by diffusion. Open water zones are subject to wind-driven mixing. The bottom topography may form deeper pathways, further contributing to short circuiting.

These combined phenomena produce a distribution of transit times for water parcels. The combined effect of these processes can be demonstrated by passing an inert tracer through the wetland. An impulse of the tracer, added across the flow width, moves with water through the wetland as a spreading cloud. Many treatment wetlands have been tracer tested, and all exhibit a significant departure from plug flow (Kadlec, 1994a; Stairs and Moore, 1994; King *et al*., 1997). Figure B.2 displays the movement of a bromide tracer impulse through an aerated horizontal subsurface flow (HSSF) wetland, as inferred from a $4 \times 4$ lateral and longitudinal array of sampling ports (Nivala, 2005). Similar two-dimensional profiles are shown in Figure 6.14 in Chapter 6.

Real-world flow patterns, such as the ones illustrated in Figure B.2, can be approximated using a variety of different flow models. The simplest, and most widely used, is to assume that the wetland can be represented as a series of CSTRs. This model, the TIS model, is addressed in the next section of this appendix.

## THE TANKS-IN-SERIES FLOW MODEL

The TIS flow model bridges the gap between the idealized extremes of the PF and CSTR reactor types. In the TIS model, the wetland is represented by a number of CSTRs in series, as shown in Figure 6.19 in Chapter 6. The flow enters

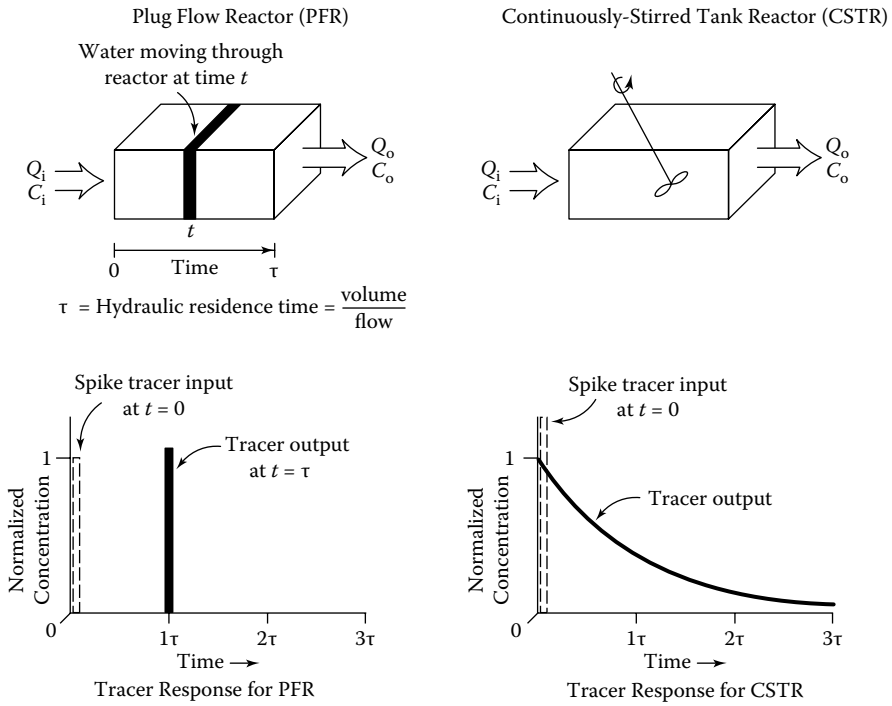



**FIGURE B.1** Comparison of ideal reactors.

the first CSTR, is mixed, and then flows into the next CSTR. The number of tanks in the series, *N*, is an important parameter in the description of the movement of both reactive and nonreactive substances.

When $N = 1$, this TIS model simplifies to the CSTR ideal reactor. As the number of CSTRs increases, the flow comes closer to approximating plug flow (Crites and Tchobanoglous, 1998), as shown in Figure B.3. If there are an infinite number of tanks in series, the internal mixing goes to zero, and the TIS model simplifies to the ideal PF reactor. Thus, the tracer response curve generated by the TIS model is a function of *N*.

It is important to note that *N* is a mathematical fitting parameter. It does not represent the physical configuration of the wetland. A treatment wetland with three cells will not have $N = 3$.

### Tracer versus Nominal Hydraulic Detention Times

The number of tanks ($N$) that best represents the hydraulic characteristics of the wetland is not known *a priori*. If an impulse tracer test is conducted, the wetland will generate a tracer response curve at the outlet (or other monitoring location), similar to the ones shown in Figures 6.15–6.17, Chapter 6.

The tracer detention time, $\tau$, can be calculated from the tracer output data (Equation 6.36, Chapter 6):

$$\tau = \frac{1}{M_o}\int_0^\infty tQCdt$$

where

$C(t)$ = tracer exit concentration, g/m$^3$ = mg/L
$M_o$ = mass of tracer in outflow, g
$\tau$ = tracer detention time, d
$t$ = time, d
$Q$ = average flow rate, m$^3$/d

The tracer detention time, $\tau$, is often less than the nominal detention time $\tau_n$. This is because not all the wetland volume is involved in the flow path, as was assumed in the calculation of $\tau_n$. As discussed in Chapter 2, the volumetric efficiency, $e_V$, is an important parameter relating $\tau$ and $\tau_n$. For FWS wetlands, $e_V$ can be defined as follows (Equation 2.5):

$$e_V = \frac{V_{active}}{(LWh)_n} = \frac{\varepsilon\eta h}{h_n}$$

where

$e_V$ = wetland volumetric efficiency, dimensionless
$V_{active}$ = active wetland volume, m$^3$
$\varepsilon$ = fraction of volume occupied by water, dimensionless
$\eta$ = gross areal efficiency, dimensionless
$h$ = water depth, m
$h_n$ = nominal water depth, m
$LWh_n$ = nominal wetland volume, m$^3$

It is then clear that

$$\tau = e_V\tau_n \tag{B.10}$$

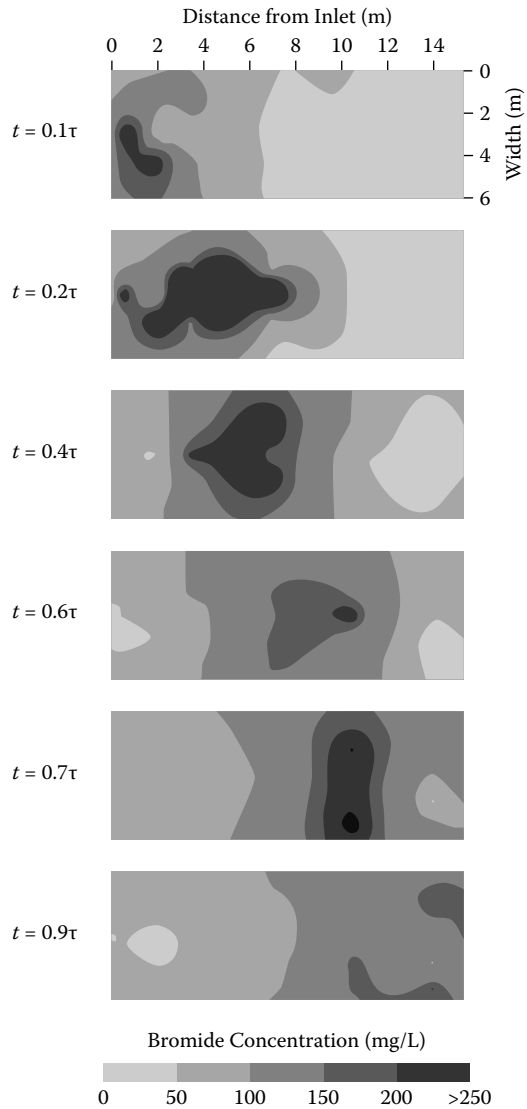



**FIGURE B.2** Tracer movement in a HSSF wetland. Time-series tracer contours were plotted from influent/effluent data along with 18 internal sampling points using Groundwater Modeling System (GMS) software. During this study, the effluent tracer concentration was modeled as 6.2 TIS. (From Nivala *et al*. (2004) *Hydraulic investigation of an aerated, subsurface flow constructed wetland in Anamosa, Iowa*. Poster presentation at the 9th International Conference on Wetland Systems for Water Pollution Control, 26–30 September 2004, Association Scientifique et Technique pour l'Eau et l'Environnement (ASTEE), Cemagref, and IWA, Avignon, France.)

Table B.1 lists volumetric efficiency results from tracer testing efforts at the Orlando Easterly treatment wetland (Martinez and Wise, 2003b), where 15 of 17 FWS wetland cells were tracer tested.

## The Detention Time Distribution

Calculation of the tracer detention time, τ, tells us a useful characteristic of the treatment wetland, namely, the average time water spends in the wetland. However, observation of tracer response curves, such as the ones shown in Figures 6.15–6.17, Chapter 6, clearly indicates that water is moving at different speeds within the wetland. Thus, the tracer response curve illustrates the entire range of detention times observed in the wetland. This range of detention times is termed as the *detention time distribution*, or DTD.

The DTD can be defined as

$$f(t)\Delta t = \text{fraction of incoming water that stays in the wetland for a length of time between } t \text{ and } \Delta t \tag{B.11}$$

where

$f$ = DTD function, $d^{-1}$
$t$ = time, d
$\Delta t$ = time increment, d

The exit tracer concentration is related to the DTD function. For an impulse tracer entering a system, the concentration curve, $C(t)$, can be related to the DTD function, $f(t)$, by

$$f(t) = \frac{QC(t)}{\int_0^\infty QC(t)dt} \tag{B.12}$$

where

$C(t)$ = exit tracer concentration, $g/m^3$ = mg/L
$Q$ = water flow rate, $m^3/d$

The numerator is the mass flow of the tracer in the wetland effluent at any time $t$ after the time of the impulse addition. The denominator is the sum of all the tracer collected and thus should equal the total mass of tracer injected. Equation B.12 represents the observed DTD function.

If flow rate is constant, $Q$ may be deleted from the numerator and denominator, and Equation B.12 simplifies to

$$f(t) = \frac{C(t)}{\int_0^\infty C(t)dt} \tag{B.13}$$

The tracer concentration can be measured at interior wetland points as well as at the outlet. Equation B.12 or B.13 may then be used to determine the distribution of transit times to that internal point. In this broader sense, the DTD becomes a function of internal position, $f(x,t)$.

The TIS model described in Figure B.3 is defined as a number ($N$) of equally sized, perfectly mixed tanks arranged in series. The number of tanks can be any integral number between 1 and ∞. The response of this series of tanks is

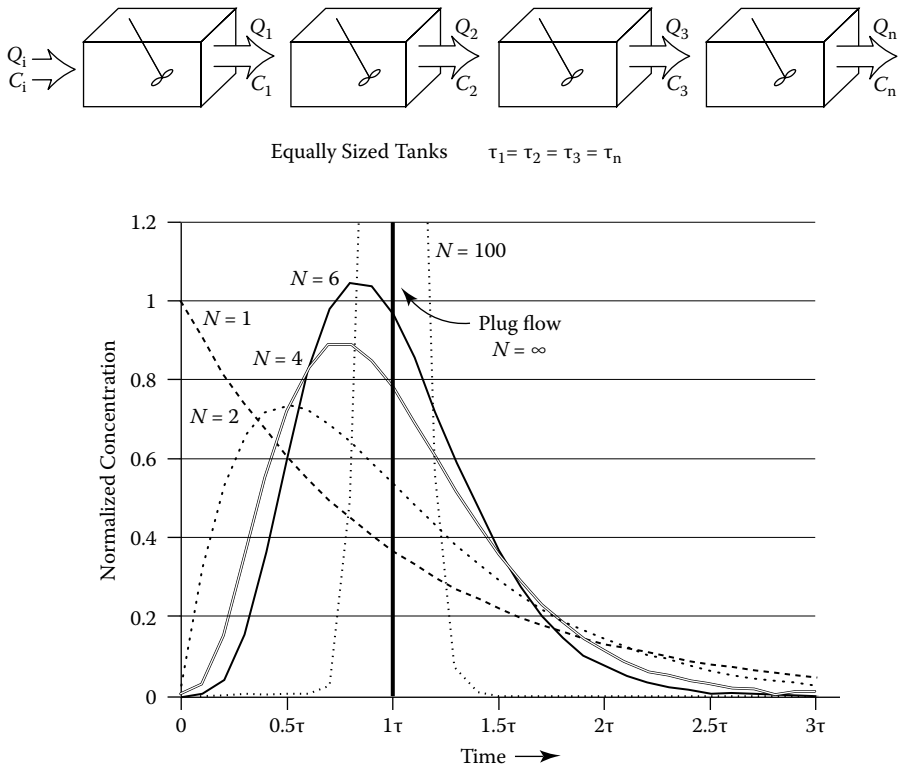



**FIGURE B.3** Response of a closed vessel to a unit impulse of an ideal inert tracer as a function of the number of tanks in series. (Adapted from Levenspiel (1972) *Chemical Reaction Engineering.* First Edition, John Wiley and Sons, New York.)

**TABLE B.1**
**Results of Cell-Wise Tracer Testing at the Orlando Easterly Wetlands**

| Cell | Cell Area (ha) | Mass Recovery (%) | Actual Residence Time (days) | Nominal Residence Time (days) | Cell Volumetric Efficiency |
|---|---|---|---|---|---|
| 1 | 23 | 109 | 1.38 | 4.53 | 0.30 |
| 2 | 25 | 123 | 5.67 | 6.47 | 0.88 |
| 3 | 6 | 91 | 1.34 | 2.57 | 0.52 |
| 4 | 6 | 95 | 2.28 | 8.97 | 0.25 |
| 5 | 6 | 87 | 0.44 | 4.02 | 0.11 |
| 6 | 6 | 84 | 1.62 | 2.29 | 0.71 |
| 7 | 12 | 97 | 2.63 | 13.00 | 0.20 |
| 8 | 12 | 112 | 1.24 | 2.95 | 0.42 |
| 9 | 11 | 101 | 2.02 | 6.74 | 0.30 |
| 10 | 11 | 111 | 1.96 | 3.44 | 0.57 |
| 11 | 23 | 105 | 29.5 | 60.5 | 0.49 |
| 12 | 24 | 105 | 8.12 | 11.7 | 0.69 |
| 13 | 52 | 75 | 8.39 | 20.8 | 0.40 |
| 14 | 47 | 92 | 5.62 | 12.3 | 0.46 |
| 15 | 54 | 94 | 9.19 | 12.5 | 0.74 |

*Source:* Based on Martinez and Wise (2003b) *Journal of Environmental Engineering (ASCE)* 129(6): 553–560.

calculated from the dynamic tracer mass balance equations for the tanks:

$$V_j \frac{dC_j}{dt} = Q(C_{j-1} - C_j) \qquad j = 1, 2, \ldots, N \quad \text{(B.14)}$$

where

$C_j$ = tracer concentration in unit $j$, mg/L = g/m$^3$
$t$ = time, d
$V_j$ = volume of unit $j$, m$^3$

Equation B.14 is easily rearranged:

$$\tau_j \frac{dC_j}{dt} + C_j = C_{j-1} \qquad j = 1, 2, \ldots, N \quad \text{(B.15)}$$

where

$\tau_j$ = tracer detention time in unit $j$, d

Because all the units are of equal volume, the tracer detention time of the entire system is $\tau = N\tau_j$. If a unit impulse of concentration is fed to the series of tanks as a feed concentration condition, the resulting effluent concentration from the *Nth* tank is the tracer concentration response according to the model. Thus, Levenspiel (1972) demonstrates that the DTD curve for the TIS model can be represented by

$$f(t) = \frac{N^N t^{N-1}}{\tau^N (N-1)!} \exp\left(\frac{-Nt}{\tau}\right) \quad \text{(B.16)}$$

### Moment Analysis

The moments of the DTD define the key parameters that characterize the wetland, the two most important being the actual detention time and spreading of the concentration pulse due to mixing (variance of the pulse). The *nth* moment about the origin is defined by

$$M_n = \int_0^\infty t^n f(t) dt \quad \text{(B.17)}$$

The zeroth moment represents the definition of the fractional character of the DTD function. Because the term $f(t)\Delta t$ represents the fraction of tracer that spends between time $t$ and $t + \Delta t$ in the system, the sum of these fractions is unity:

$$\int_0^\infty f(t) dt = 1 \quad \text{(B.18)}$$

The first absolute moment is the tracer detention time $\tau$. This value defines the centroid of the exit tracer concentration distribution:

$$\int_0^\infty t\, f(t) dt = \tau \quad \text{(B.19)}$$

It is sometimes useful to work with a dimensionless time variable, defined as the actual time divided by the tracer detention time:

$$\theta = \frac{t}{\tau} \quad \text{(B.20)}$$

A second parameter that can be determined directly from the residence time distribution is the variance ($\sigma^2$), which characterizes the spread of the tracer response curve about the mean of the distribution, which is $\sigma^2$. This is the second central moment about the mean:

$$\int_0^\infty (t-\tau)^2\, f(t)\, dt = \sigma^2 \quad \text{(B.21)}$$

where

$\sigma^2$ = DTD variance, d$^2$

The variance of the DTD is created by the mixing of water during passage, or, equivalently, by a distribution of the velocities of passage. This can be lateral, longitudinal, or vertical mixing. This measure of dispersive processes may be rendered dimensionless by dividing by the square of the tracer detention time:

$$\sigma_\theta^2 = \frac{\sigma^2}{\tau^2} \quad \text{(B.22)}$$

The new parameter is $\sigma_\theta^2$, the dimensionless variance of the tracer pulse.

For the TIS model, it is possible to define simple relations between the parameters of the distribution and these moments. For instance, the TIS conceptual model (which produces a gamma distribution) has a dimensionless variance, given by

$$\sigma_\theta^2 = \frac{\tau - \tau_p}{\tau} \quad \text{(B.23)}$$

$$\sigma_\theta^2 = \frac{1}{N} \quad \text{(B.24)}$$

### Gamma Distribution Fitting

Virtually the entire early literature on tracer testing of wetlands and ponds utilized (archaic) parameter estimation methods that reflected the computational tools available when they were developed around 35 years ago. The most common method involves computation of the first and second moments of the experimental outlet concentration distribution (Equations B.19 and B.21) via numerical integration.

## TABLE B.2
## Sample Calculations for Workup of a Tracer Test

Background = 9 µg/L
Tracer Added = 782 kg
Recovery = 98%

$\tau$ = 12.95 d
$\sigma^2$ = 61.15 d$^2$
$\sigma_\theta^2$ = 0.365

Moment
N = 2.74
$t_i$ = 4.722 d
$\tau$ = 12.95 d

Least Squares
N = 5.47
$t_i$ = 2.04 d
$\tau$ = 11.17 d
SSQE = 0.004

| | | | | | | | Moment | | | Least Squares | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Elapsed Time (days)** | **Flow (m³/d)** | **Raw Concentration (µg/L)** | **Adjusted Concentration (µg/L)** | ***QCΔt* (kg)** | **Data DTD (d⁻¹)** | ***tQCΔt*** | **$(t-\tau)^2$ *tQΔt*** | **Gamma DTD (d⁻¹)** | **Pred. Conc. (µg/L)** | **Gamma DTD (1/d)** | **Error Squared** | **Predicted Concentration (µg/L)** |
| 0.0 | 275,558 | 9 | 0 | | 0.000 | | | 0.000 | 9 | 0.000 | 0.000 | 9 |
| 0.9 | 275,558 | 9 | 0 | 0 | 0.000 | 0 | 0 | 0.000 | 9 | 0.000 | 0.000 | 9 |
| 1.2 | 284,672 | 8 | −1 | 0 | 0.000 | 0 | −6 | 0.006 | 9 | 0.000 | 0.000 | 10 |
| 1.5 | 284,672 | 9 | 0 | 0 | 0.000 | 0 | −3 | 0.009 | 9 | 0.001 | 0.000 | 12 |
| … | … | … | … | … | … | … | … | … | … | … | … | … |
| 3.9 | 278,499 | 44 | 35 | 3 | 0.013 | 12 | 288 | 0.038 | 91 | 0.025 | 0.000 | 79 |
| 4.2 | 263,747 | 109 | 100 | 6 | 0.034 | 25 | 471 | 0.041 | 103 | 0.031 | 0.000 | 100 |
| 4.5 | 263,747 | 100 | 91 | 8 | 0.031 | 36 | 620 | 0.044 | 113 | 0.037 | 0.000 | 118 |
| 4.9 | 263,747 | 191 | 182 | 12 | 0.062 | 57 | 808 | 0.047 | 129 | 0.044 | 0.000 | 136 |
| … | … | … | … | … | … | … | … | … | … | … | … | … |
| 32.6 | 223,775 | 20 | 11 | 5 | 0.003 | 161 | 1,776 | 0.005 | 35 | 0.000 | 0.000 | 10 |
| 35.6 | 143,877 | 14 | 5 | 5 | 0.001 | 161 | 1,998 | 0.004 | 29 | 0.000 | 0.000 | 9 |
| 37.6 | 260,816 | 13 | 4 | 2 | 0.001 | 65 | 1,008 | 0.002 | 27 | 0.000 | 0.000 | 9 |
| 39.6 | 298,678 | 16 | 7 | 3 | 0.003 | 124 | 2,144 | 0.002 | 30 | 0.000 | 0.000 | 9 |
| 42.6 | 596,050 | 16 | 7 | 10 | 0.005 | 396 | 7,838 | 0.001 | 16 | 0.000 | 0.000 | 9 |
| 44.6 | 507,213 | 14 | 5 | 6 | 0.003 | 280 | 5,991 | 0.001 | 13 | 0.000 | 0.000 | 9 |
| 46.6 | 234,219 | 14 | 5 | 4 | 0.001 | 160 | 3,696 | 0.001 | 11 | 0.000 | 0.000 | 9 |
| **Sum or average** | **276,069** | | | **769** | | **9,953** | **46,990** | | | | **0.004** | |

A serious failing of the moment method of parameter estimation is that it emphasizes the "tail" of the response much more than the central portion—i.e., the peak area. Minor concentration anomalies on the tail of the concentration response curve may yield spurious parameter values. Often, a better procedure is to utilize a robust parameter determination routine, such as a search to minimize the sum of the squared errors between the selected DTD function and the data.

From Figure B.3, it is easy to see that the shape of the DTD is quite sensitive to changes in $N$ when $N$ is small. Note the change in magnitude and shape as the number of tanks increases from the ideal CSTR ($N = 1$) to 2 TIS and from 2 to 6 TIS. Because most wetland systems operate as a few (3–8) TIS, it is advantageous to be able to change $N$ from a discrete integer variable to a continuous (noninteger) variable. This enables the modeler to utilize fractional values of $N$, increasing the flexibility with which a dataset can be fit with a model. The gamma distribution $f(t)$ is defined as:

$$f(t) = \frac{N^N t^{N-1}}{\tau^N \, \Gamma(N)} \exp\left(\frac{-Nt}{\tau}\right) \qquad \text{(B.25)}$$

The gamma function $\Gamma(N)$ is defined by

$$\Gamma(N) = \int_0^\infty \exp(-x) x^{N-1} dx \qquad \text{(B.26)}$$

Equation B.25 represents a DTD function that may be fit to data. The GAMMADIST function is available in Microsoft Excel™ and returns values of $f$ for the time $t$ and the parameters $N$ and $\tau$.

The SOLVER application in Microsoft Excel™ allows the modeler to simultaneously solve for the variables $N$ and $\tau$ that minimize the difference between the observed DTD (Equation B.13) and the predicted DTD (Equation B.25). Examples of results of this approach are shown in Figures 6.13 and 6.15, Chapter 6.

As an illustration of the potential problems of the old moment analysis procedure and the ability of the sum of the squared errors (SSQE) minimization, consider the data set for the lithium tracer test of Cell 2 of the Everglades Nutrient Removal Project, Florida, FWS wetland (Figure B.4). Computations are illustrated in Table B.2. SSQE fits the peak area of the response, whereas moment calculations fit the tail. Moment analysis produces a higher tracer detention time (12.95 days versus 11.17 days) and a lower number of TIS (2.74 versus 5.47). The moment parameters produce a poorly appearing "fitted" gamma curve. The bulk of the tracer, and hence the important part of the response, is contained in the peak zone. Accordingly, the SSQE minimization analysis of tracer data is recommended, rather than moment calculations.

Here, the goodness of fit is measured via the root mean square (RMS) error between model and data DTD values. The range $0 < \theta < 4$ is chosen to eliminate repeated zero errors associated with the tail of the distribution. The RMS error is divided by the peak height of the distribution to provide shape scaling. The RMS error for the moment fit in Figure B.4 is 24.5%, whereas the SSQE fit yields an RMS error of 9.2%.

## TRACER TEST OBSERVATIONS

Treatment wetlands have been built to many different specifications and in many geometric layouts. Tracer tests for many have produced similar results, without evidence of pathological hydraulic behavior. Here, examples are given to illustrate commonly encountered features.

### FWS Wetland Systems

The results of an illustrative set of tracer test results for FWS wetlands is given in Table 6.1, Chapter 6. The average recovery for those 35 systems was 84%, the mean $N$TIS = 4.1, and the median $N$TIS = 3.6. More detailed information for a FWS system is informative.

The Tres Rios, Arizona, demonstration project conducted over 20 tracer tests of FWS wetlands used to polish municipal effluent. The Tres Rios Demonstration Constructed Wetland Project consisted of three discrete sites totaling 5.8 ha, of which 4.3 ha comprised two demonstration-scale free water surface (FWS) wetland facilities: a hayfield riparian site (H1 and H2) and a cobble site (C1 and C2). Wetland H2 was a 1.28-ha with a typical detention time of four days. Basin H2 had approximately 25% of its surface area as open-water deep zones, obtained using two large internal deep-zones with waterfowl islands. The primary vegetation consisted of two species of bulrush, *Scirpus validus* (soft-stem bulrush) and *Scirpus olneyi* (three-square bulrush). Tracer results have been discussed in Whitmer *et al.* (2000) and Keefe *et al.* (2004b).

The data from one of the bromide impulse tests are shown in Figure B.5, and the conditions for the test are given in Table B.3. The nominal detention time was 75 hours. A typical bell-shaped response is seen, with the first tracer appearing at the outlet at 16 hours. The tracer recovery was 88%, which indicates relatively conservative behavior. Three methods of analysis are illustrated: TIS from moments, TIS from least squares, and delayed (shifted) TIS from least squares. The RMS goodness of fit improves in that order (Table B.3). Depending on the fitting technique, the volumetric efficiency ranges 71–76%, indicating that most of the wetland water is involved in flow. The dimensionless variance is 0.237, corresponding to overall TIS = 4.2. However, if the shifted TIS DTD was used, the wetland behaved similar to a plug flow unit of 16 hours detention, combined with 38 hours detention in 2.8 TIS. The TIS moment fit of the DTD appears good only for the tail of the DTD.

The conversions for first-order, zero-background removal are given in Table B.3. In general, the plug flow approximation is not good, except for very low removals ($Da = 1$).

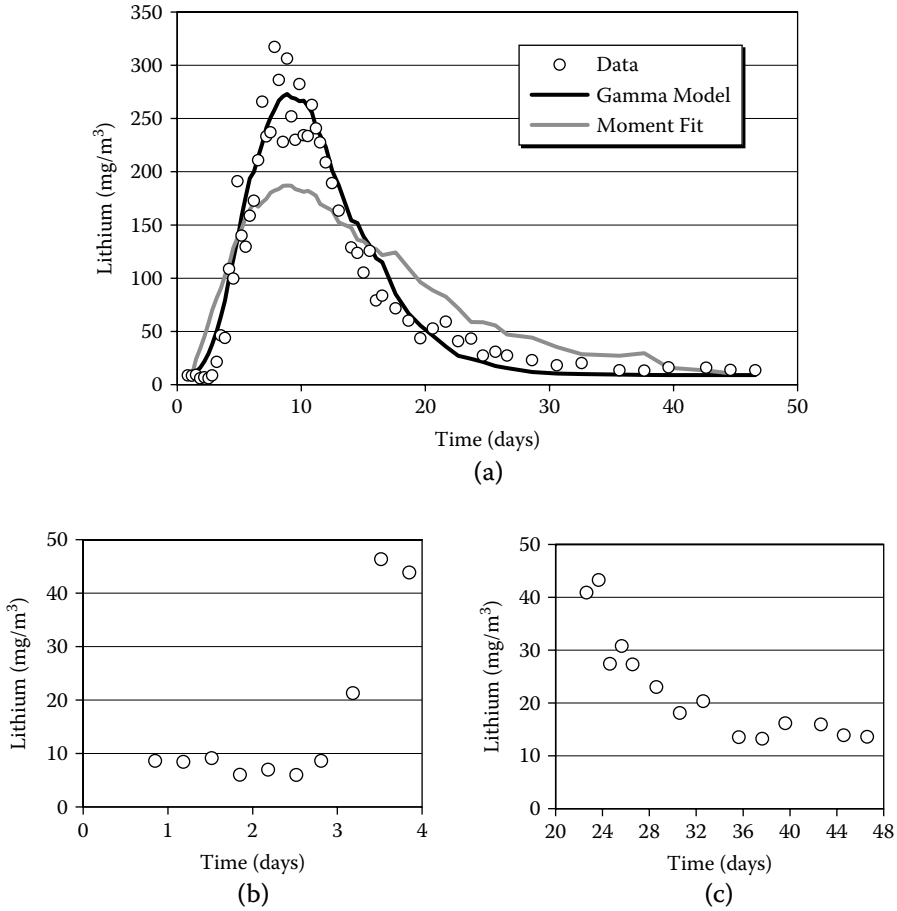



**FIGURE B.4** Tracer response for Cell 2 of the ENRP project, together with the least squares fit to a gamma function. The lower panels display expansions of the starting and ending periods of the test.

## HSSF Wetland Systems

The results of an illustrative set of tracer test results for HSSF wetlands are given in Table 6.2, Chapter 6. The average recovery for the 37 systems was 92%, the mean *N*TIS = 11.0, and the median *N*TIS = 8.3. More detailed information for a HSSF system is informative.

The HSSF wetland system at Minoa, New York, consisted of three flow paths with two cells in series in each, totaling 0.67 ha. The average design flow was 600 $m^3$/d, corresponding to a nominal detention time of three days. The substrate was 10–15 cm of 6-mm pea gravel on top of 75 cm of 20 mm gravel, and the cells were vegetated with *Phragmites australis* and *Scirpus validus*. Tracer results have been discussed in Marsteiner *et al*. (1996) and Marsteiner (1997). For the test example, the full flow was directed to cell 1.

The data from one of the bromide impulse tests are shown in Figure B.6, and the conditions for the test are given in Table B.3. The nominal detention time was 20.6 hours. A typical bell-shaped response is seen, with the first tracer appearing at the outlet at 9.6 hours. The tracer recovery was 98%, which indicates conservative behavior. Three methods of analysis are illustrated: TIS from moments, TIS from least squares, and delayed (shifted) TIS from least squares. The RMS goodness of fit improves in that order (Table B.3). Depending on the fitting technique, the volumetric efficiency ranges from 75–79%, indicating that most of the wetland water is involved in the flow. The dimensionless variance is 0.089, corresponding to overall TIS = 11.2. However, if the shifted TIS DTD was used, the wetland behaved similar to a plug flow unit of 9.6 hours' detention, combined with 11 hours' detention in 6.9 TIS. The TIS moment fit of the DTD appears good.

The conversions for first-order, zero-background removal are given in Table B.3. In general, the plug flow approximation is not good, except for very low removals (*Da* = 1).

## Event-Driven Wetland Systems

Tracer analysis of event-driven systems is complicated by two factors: the flow is not steady, and all the tracer may not be flushed out of the wetland by a single event. Despite these difficulties, it has been shown that the hydraulic flow patterns are similar under event-driven and continuous flow (Werner and

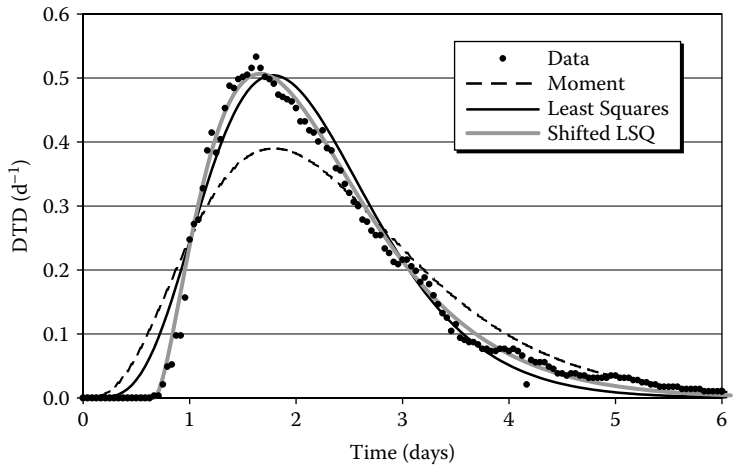



**FIGURE B.5** FWS tracer test results for Tres Rios Hayfield wetland 2, together with three different TIS fits. Conditions are given in Table B.3.

Kadlec, 1996). Methods for dealing with unsteady flows and depths have been outlined by Holland *et al*. (2004; 2005).

It is instructive to examine the hypothetical response of a 4-TIS wetland to a tracer addition to an inflow that terminates before flushing the wetland. A nominal detention time of three days is selected, corresponding to a fixed inflow of 500 $m^3$/d and a full-flow volume of 1,500 $m^3$. During the tracer test, the wetland receives 500 $m^3$/d for a period of three

**TABLE B.3**
**Details of Tracer Tests for the Tres Rios Hayfield, Arizona (FWS), and Minoa, New York (HSSF), Examples**

**Hayfield 2 FWS**

| | |
|---|---|
| Nominal detention time (h) | 74.6 |
| % Recovery | 88% |
| Dimensionless variance | 0.237 |

| | Moment | Least Squares | Shifted Least Squares | Plug Flow |
|---|---|---|---|---|
| Tracer detention time (h) | 56.4 | 53.0 | 53.5 | — |
| Volumetric efficiency, % | 76% | 71% | 72% | — |
| Plug flow fraction, % | 0 | 0 | 30% | — |
| Number of TIS | 4.2 | 6.2 | 2.8 | — |
| RMS goodness of fit, % | 15.9 | 6.6 | 3.4 | — |
| Remaining @ Da = 1, % | 40.8% | 39.6% | 39.7% | 36.8% |
| Remaining @ Da = 3, % | 10.4% | 8.7% | 8.5% | 5.0% |
| Remaining @ Da = 5, % | 3.7% | 2.6% | 2.3% | 0.7% |

**Minoa HSSF Cell 1**

| | |
|---|---|
| Nominal detention time (h) | 20.4 |
| % Recovery | 98 |
| Dimensionless variance | 0.089 |

| | Moment | Least Squares | Shifted Least Squares | Plug Flow |
|---|---|---|---|---|
| Tracer detention time (h) | 16.1 | 15.4 | 16.0 | — |
| Volumetric efficiency, % | 79% | 75% | 78% | — |
| Plug flow fraction, % | 0 | 0 | 47% | — |
| Number of TIS | 11.2 | 14.3 | 3.7 | — |
| RMS Goodness of Fit, % | 13.3 | 9.6 | 6.9 | — |
| Remaining @ Da = 1, % | 38.4% | 38.0% | 38.1% | 36.8% |
| Remaining @ Da = 3, % | 7.0% | 6.6% | 6.5% | 5.0% |
| Remaining @ Da = 5, % | 1.6% | 1.4% | 1.3% | 0.7% |

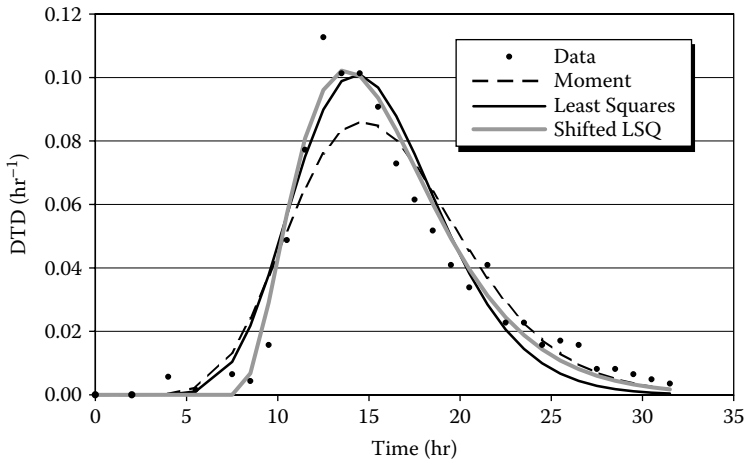



**FIGURE B.6** Tracer test results for a HSSF wetland at Minoa, New York, together with three different TIS fits. Conditions are given in Table B.3.

days, but then the inflow stops. It is presumed that the wetland outflow is governed by a weir structure. The tracer concentration response at the system outlet is shown in Figure B.7. Two artifacts of the test results are apparent. First, there is a delay as the water level builds up in the wetland, with low outflows as the height over the weir increases. If there were no outflow, it would take three days to fill the wetland to the new depth. After the inflow ceases, the wetland drains back down to the elevation of the weir. Outflows decrease back down to zero over the next few days. There is residual tracer in the wetland, corresponding to a recovery of 68% (32% remaining). As a second result, the concentration near the outflow point, which has become stagnant, remains at an elevated value until another event again causes outflow.

Such distortion of the DTD makes it difficult to ascertain the hydraulic parameters of the wetland. A flow-weighted time, proportional to the volume of water that has exited the wetland, removes these two artifacts. On that basis, the response changes to the shape characteristic of continuous flow systems. There are alternative choices for scaling, as discussed by Werner and Kadlec (1996), who provide the mathematical background for rescaling to the volumetric approach.

## Variability in Tracer Results

As for any other treatment wetland performance parameter, there is variability in the volumetric and DTD efficiency results. That variability may be caused by seasonal variables, such as litter or algal density; or it may be caused by meteorological factors, such as wind, evapotranspiration (*ET*), or rainfall. A set of six warm-season tests on FWS wetland EW3 at Des Plaines, Illinois, gave $e_V = 0.70 \pm 0.13$ and $e_{DTD} = 0.62 \pm 0.10$ (Kadlec, 1994). Results from the Tres Rios, Arizona, demonstration wetlands ($N = 3$) showed a narrower range for $e_{DTD}$, with a typical coefficient of variation of 0.03–0.06.

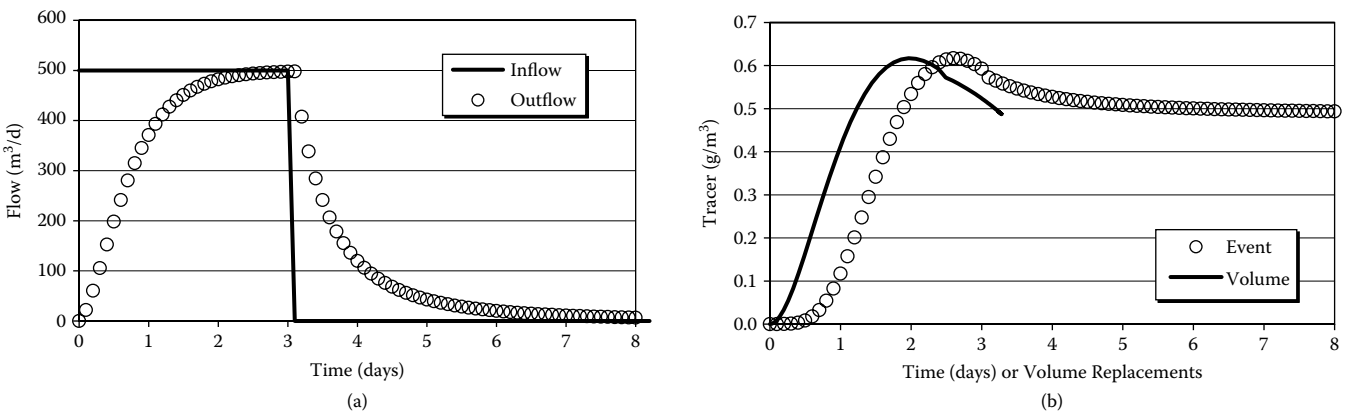



**FIGURE B.7** Tracer response for a hypothetical stormwater wetland. The inflow is presumed to last for three days (a), during which the system fills to a new operating depth. The tracer is not completely flushed (b), resulting in a residual concentration. The solid line shows a rescaling of "time" to a cumulative volume outflow basis.



**FIGURE B.8 (A color version of this figure follows page 550)** Progress of Rhodamine WT dye tracer through a FWS wetland, cell 4 of the ENRP in Florida. The dye was introduced along the upper boundary, and flow proceeds from the top to the bottom. Note the short-circuit along the left-hand side, which is partially redistributed by the central cross canal. (Photo courtesy T. DeBusk.)

However, there was wider variability for $e_V$, with a typical coefficient of variation of 0.13–0.33. Thus, it appears that the coefficient of variation for efficiency results may be in the range 5–30% for FWS systems.

## COMMON ABERRATIONS

Although it is tempting to think that the entire wetland water volume is swept by the flow in a moderately uniform manner giving rise to a gamma DTD, deviations from that pattern are commonly found. Two pathological situations are the existence of short-circuits from inlet to outlet and the existence of deadwater zones that are not swept by flow.

### Short-Circuiting

When part of the water follows a fast, preferential path through the wetland, the result is termed channeling or short-circuiting. On the ground, these fast flow paths may result from bathymetry, such as deeper channels from inlet to outlet (see Figure B.8). The tracer detention time will be less than the nominal. A pathological case is illustrated in Figure B.9, Lakeland, Florida, cell 1 (Keller and Bays, 2001). The recovery was 106%. The nominal detention time was 17.3 days; the tracer detention time was 5.8 days, for a volumetric efficiency of 33%. The dimensionless variance was 0.66, corresponding to *N*TIS = 1.5. The DTD curve for this wetland was clearly bimodal, indicating two flow paths. When the DTD is fit with a two-path

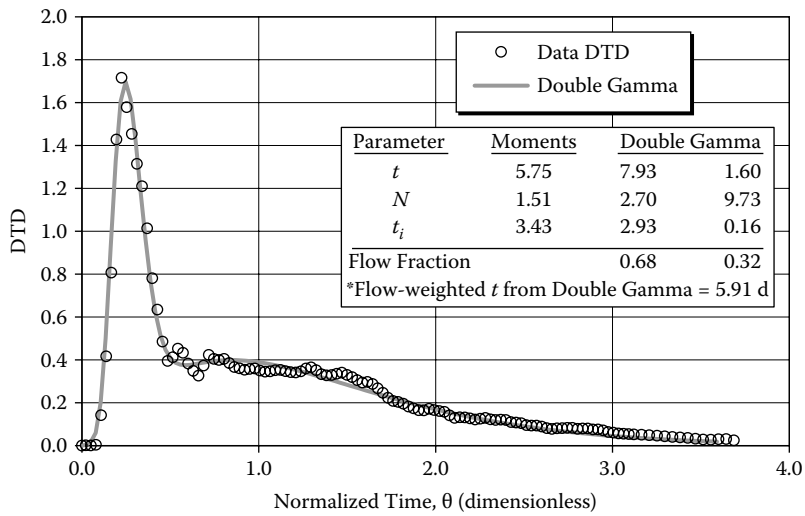



**FIGURE B.9** Tracer response for a short-circuited FWS wetland, Cell 1 at Lakeland, Florida.

model, the fit is excellent (see section titled Other Flow Models). About a third of the water had a mean detention time of 1.6 days, whereas the remaining two thirds had a mean detention time of 7.9 days. Cell 1 contained large internal areas of linear spoil mounds parallel to the direction of flow and, therefore, presumably, was highly channelized between these areas.

### Dead Zones

Wetlands often contain parcels of water that are not in the main flow path and are not flushed by flowing water. Such zones may be due to water trapped in a mass of algae or a dense clump of vegetation or litter in the FWS wetland. In a HSSF wetland, there may be dead-end pores. Tracer enters and departs these zones by diffusion, which is usually a slow process compared to flow. As a consequence, the tracer response will exhibit a long tail corresponding to the tracer that parked temporarily in the dead zones. An example from Sacramento, California, is shown in Figure B.10, along with a TIS fit (Nolte and Associates, 1998b). The nominal detention time for this test was 5.5 days, the tracer detention time was 5.3 days, and the recovery was 97%. If the moment method is used to fit the data, the tail region controls, and the peak zone is poorly represented. Alternative methods of analysis are discussed in the section on wetland–tracer interactions that can account for the dead zones, but at the expense of additional modeling parameters. The implications for pollutant removal are discussed in the section on pollutant removal effects.

## EXTENSIONS TO THE TIS FLOW MODEL

The simplest TIS model, and the gamma distribution that represents it, does not allow for some of the very real phenomena that may be encountered. DTD may result from velocity profile effects rather than mixing. When that is the case, it has a zero portion for short times, up to the shortest travel time experienced by rapidly moving water. For instance, that rapid path is typically associated with surface water layers in unvegetated areas of a FWS wetland. In HSSF wetlands, the rapid paths are the most direct routes between the media particles, as opposed to paths that wander off to the side to move around particles. In either case, there is a nonzero minimum of the travel time. This concept is discussed in detail in the engineering literature (see for example, Levenspiel, 1972). Another set of unaccounted processes includes water gains and losses, in the form of seepage, rainfall, and evaporation. These factors may be included by modifying the TIS model structure.

### TIS Plus a Delay

The TIS model has been described earlier (in the section on the TIS flow model) as an example of the use of least squares fitting of DTDs. As noted, it is a two-parameter fit, using detention time $\tau$, and number of tanks $N$. It does not suffer from a constraint of small degrees of nonideality and can cover the entire range from one well-mixed unit to a plug flow. Drawbacks are the inability to describe either the breakthrough delay or long tail resulting from retardation.

The tanks plus a delay have been utilized as a model for dealing with the breakthrough delay evidenced in most tracer response curves. This model may also be easily coded in a spreadsheet, but there are three parameters: the delay time $t_D$, the detention time $\tau$, and the number of tanks $N$. Many authors recommend the inclusion of this component of the response (Kadlec *et al*., 1993; Kadlec and Knight, 1996; Chazarenc *et al*., 2004; Marsili-Libelli and Checchi, 2005).

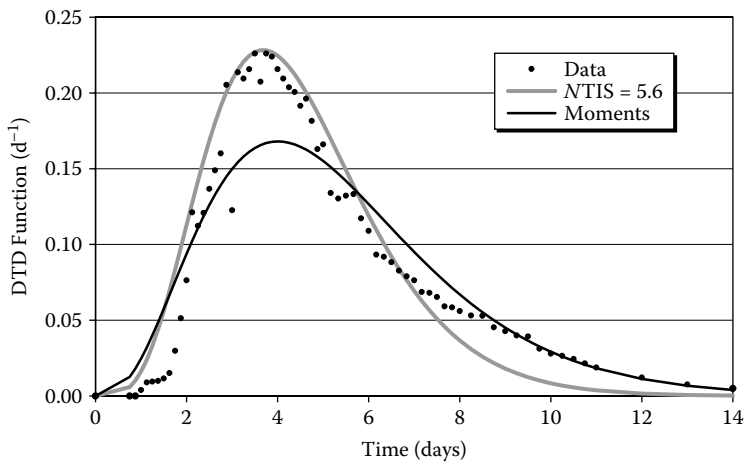



**FIGURE B.10** Tracer test result from the Sacramento, California, wetland Cell 7. The least squares TIS model does not account for the long tail of the experimental curve. The moment method fits the tail, but misses the peak area. (Data from Nolte and Associates (1998a) *Sacramento Regional Wastewater Treatment Plant Demonstration Wetlands Project*. 1997 Annual Report to Sacramento Regional County Sanitation District, Nolte and Associates.)

$$f(t)\begin{cases} = \dfrac{1}{t_i\,(N-1)!}\left(\dfrac{t-t_D}{t_i}\right)^{N-1} e^{\left(-\frac{t-t_D}{t_i}\right)} & t > t_D \\ = 0 & t < t_D \end{cases} \quad \text{(B.27)}$$

where
$t$ = detention time, d
$t_D$ = delay time, d
$t_i$ = mean detention time in one tank, d

### Infiltration

Some wetlands may safely infiltrate water into the ground. For example, the Tres Rios, Arizona, cobble site wetland, C1, infiltrated 60–80% of the incoming water (Kadlec, 2001c). Similarly, the Imperial, California, wetlands also leaked a considerable fraction of the incoming water, 40–60% (TTI and WMS, 2006). The tracer test theory does not usually include this leak effect on the water mass balance.

For illustration, let the model framework be the TIS concept discussed previously. The volume, depth, and planar area are considered to be the same for each unit. The leak rate is also assumed to be the same in each unit. For simplicity, the effects of rain and *ET* will be omitted from this analysis. The steady-flow water mass balance equation for the *jth* well-mixed unit is

$$Q_{j-1} = Q_j + L \qquad j = 1, 2, \ldots, N \quad \text{(B.28)}$$

where
$L$ = leak from unit $j$, $m^3/d$
$N$ = total number of units in the TIS model, integer
$Q_j$ = overland flow from unit $j$, $m^3/d$

Leakage acts to reduce the flow as water moves from inlet to outlet. Equation B.28 may be solved sequentially to determine the flow exiting each unit:

$$Q_j = Q_i - jL \qquad j = 1, 2, \ldots, N \quad \text{(B.29)}$$

where
$Q_i$ = inlet flow rate, $m^3/d$

The total leakage is

$$NL = \alpha Q_i \quad \text{(B.30)}$$

where
$\alpha$ = total fraction of inlet flow that is lost through leakage, dimensionless

At constant water volume in each unit, the tracer mass balances are

$$V_j \frac{dC_j}{dt} = Q_{j-1}C_{j-1} - (Q_j + L)C_j \qquad j = 1, 2, \ldots, N \quad \text{(B.31)}$$

where
$C_j$ = tracer concentration in unit $j$, mg/L = $g/m^3$
$t$ = time, d
$V_j$ = volume of unit $j$, $m^3$

Equations B.28 and B.31 combine to give

$$\tau_j \frac{dC_j}{dt} + C_j = C_{j-1} \qquad j = 1, 2, \ldots, N \quad \text{(B.32)}$$

where

$$\tau_j = \frac{\tau_{in}}{\left[N - \alpha(j-1)\right]} \qquad j = 1, 2, \ldots, N \quad \text{(B.33)}$$

The individual unit detention times $\tau_j$ are based on the combination of surface flow and leakage leaving the *jth* unit and its water volume. The nominal system detention time based on inlet flow $Q_i$ and the total system water volume is

$$\tau_{in} = \frac{NV}{Q_i} \quad \text{(B.34)}$$

If there is no leakage, then $\alpha = 0$, and $\tau_j = \tau_{in}/N$.

$$\tau_{an} = \tau_{in}\left(\sum_{j=1}^{N} \frac{1}{N - \alpha(j-1)}\right) \quad \text{(B.35)}$$

where
$\alpha$ = water loss fraction, dimensionless ($\alpha = 1 - R = 1 - Q_o/Q_i$)
$j$ = tank number counter, dimensionless
$N$ = total number of tanks, integer
$\tau_{an}$ = actual nominal detention time, d
$\tau_{in}$ = flow-based inlet nominal detention time, d

The effect of leakage on the measured detention time can be quite large for high values of $\alpha$, especially for high values of *N*TIS (Figure B.11). For example, if half the incoming water is lost, the true detention time for a 3-TIS wetland will be 23% greater than the inlet nominal detention time. The use of an average flow rate will always give an overestimate of the actual detention time.

The tracer is lost to leakage, and so the recovery will be less than 100%. The lowered surface outflow leads to a low and late peak (Figure B.12). However, the shape of the response is not affected, and hence the analysis of the response will give the same dimensionless variance (same number of TIS).

### Rain and Evapotranspiration

The loss or gain of water to or from the atmosphere does not carry tracer in or out of the wetland. However, *ET* does cause the water to slow as it passes through the system, and rain causes it to accelerate. Because there is normally level control at the outlet, the depth remains unchanged, but the linear

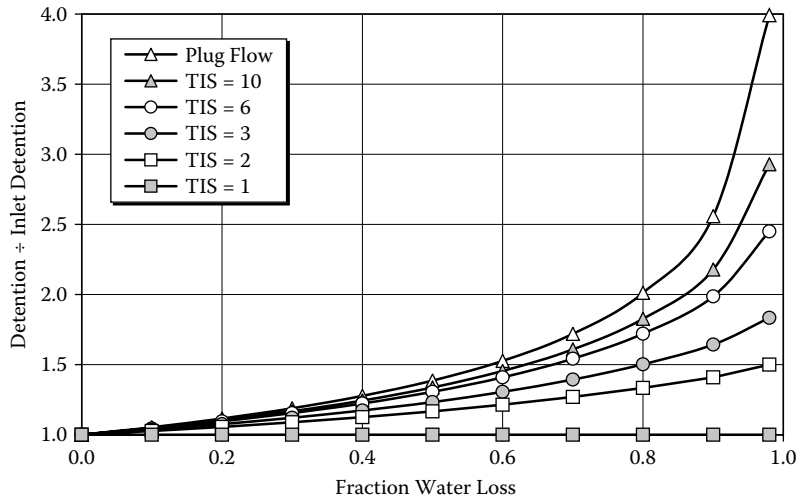



**FIGURE B.11** Detention times in leaking wetland systems, expressed as a ratio to the nominal detention time, computed from the inlet flow rate. The fractional water loss is the total for the entire wetland.

velocity of the remaining water is altered. Accordingly, for *ET*, the detention time is increased (see Chapter 2), and dissolved constituents become more concentrated. Conversely, for rain, the detention time is decreased, and dissolved constituents become diluted. The effects on tracer response have been discussed by Chazarenc *et al*. (2003; 2004), who present the theoretical result for detention time in plug flow systems with *ET* (Chapter 2, Equation 2.8):

$$\tau_{an} = \tau_{in}\left(\frac{\ln(R)}{R-1}\right)$$

where

$R$ = water recovery fraction, dimensionless ($R = Q_o/Q_i$)
$\tau_{an}$ = actual nominal detention time, d
$\tau_{in}$ = flow-based inlet nominal detention time, d

This plug flow formula works equally well for rainfall gains. However, this is not the case for a TIS system, in which the influence of precipitation and *ET* is greater. The actual detention time (tracer detention time) in a TIS wetland is

$$\tau_{an} = \tau_{in}\left(\frac{1}{N}\sum_{j=1}^{N}\left(\frac{1}{1-\frac{\alpha j}{N}}\right)\right) \qquad \text{(B.36)}$$

where

$\alpha$ = water loss fraction, dimensionless
($\alpha = 1 - R = 1 - Q_o/Q_i$)
$j$ = tank number counter, dimensionless
$N$ = total number of tanks, integer

In the limit, as $N$ becomes very large, Equation B.36 reduces to Equation B.35.

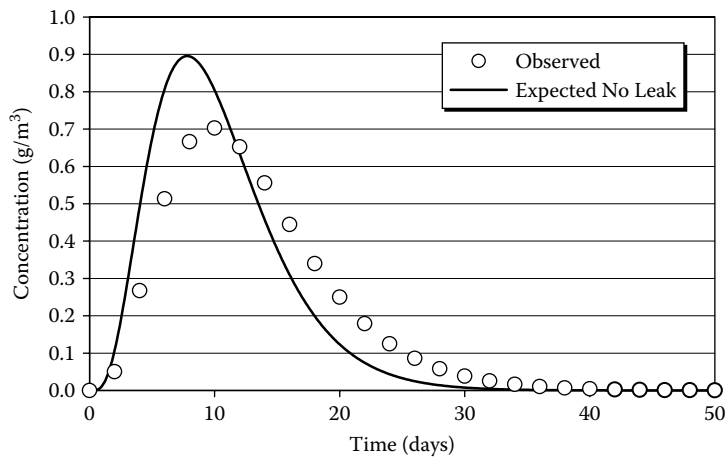



**FIGURE B.12** The effect of leakage on a tracer response for 4 TIS. The line is the forecast result for no leakage loss. The circles represent the result for 50% water loss due to leakage.

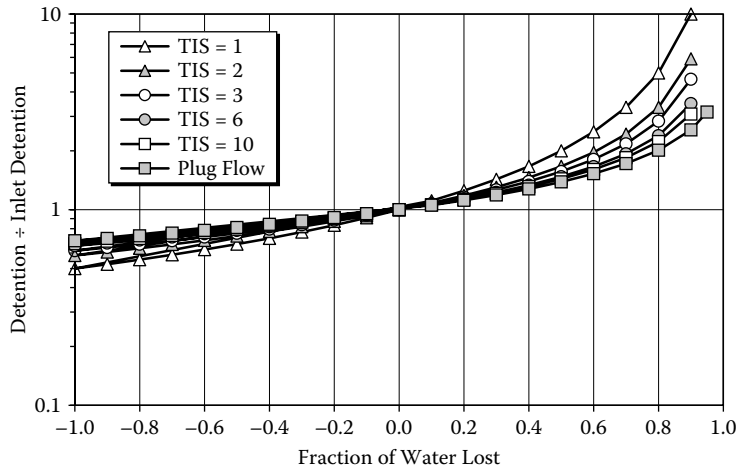



**FIGURE B.13** Detention times in wetland systems with rain or evapotranspiration, expressed as a ratio to the nominal detention time computed from the inlet flow rate. The fractional water loss or gain is the total for the entire wetland.

The effect of *ET* on the measured detention time can be quite large for high values of $\alpha$, especially for low *N*TIS (Figure B.13). For example, if half the incoming water is lost, the true detention time for a 3-TIS wetland will be 57% greater than the inlet nominal detention time. The use of an average flow rate will always give an underestimate of the actual detention time for *ET* and an underestimate for rain cases. The error is large for relatively large amounts of *ET*.

None of the tracer is lost to *ET*, so the theoretical recovery should be 100%. The combination of the evaporative concentration and lowered outflow leads to a high and late peak (Figure B.14). The combination of rain dilution and increased outflow leads to a low and early peak. However, the shape of the response is not affected, and hence the analysis of the response will give the same dimensionless variance (the same number of TIS).

## WETLAND–TRACER INTERACTIONS

In the foregoing, it has been assumed that the wetland water body can be considered as vertically uniform and that the tracer does not interact with any of the solids or biota in the system. In some cases, these assumptions are not warranted.

### Wind

If the wetland contains significant areas of open water, wind can be a factor in tracer movement. In ponds, field studies with drogues and computational fluid dynamics (CFD) modeling (HYDRO-3D) have shown that surface velocities may be 30-fold greater than deep-layer velocities and that the velocity profile is most affected in the top 10–15 cm (Guganessharajah, 2001). Lloyd *et al.* (2003) implemented windbreaks around a shallow pond (1.1 m deep), thus isolating the system from

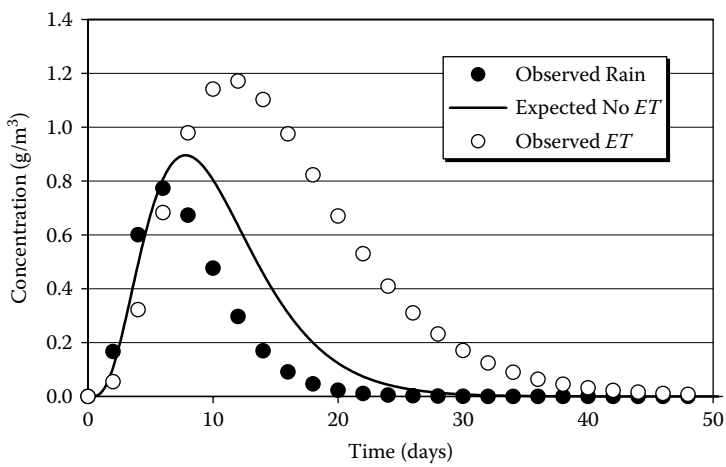



**FIGURE B.14** The effects of *ET* and rain on a tracer response for 4 TIS. The line is the forecast result for no *ET* loss. The open circles represent the result for 50% water loss due to *ET*. The closed circles represent the result for 50% water gain from rain.

winds that averaged only 0.8 m/s. The volumetric efficiency was increased from 50% to 74%, and the dispersion number was decreased from 0.66 to 0.40.

In laboratory flume experiments, with wooden dowels representing plants, Stephan *et al*. (2004) showed that tracer response curves differed in shape depending on whether the wind was with or against the flow. Wind acting on surface layers drives a surface current in the wind direction and promotes vertical mixing via recirculation in the water column. Therefore, Stephan *et al*. (2004) found that the tail of the response curve was shortened for a wind of 2.4 m/s blowing against the flow direction. The volumetric efficiency was higher with the tail wind (83%) than with the head wind (74%).

Stairs (1993) found that a tail wind of 5.7 m/s caused a very early peak concentration in the response compared to more quiescent conditions for an unvegetated FWS wetland.

There are no sufficient experimental results on vegetated operating FWS systems to formulate quantitative measures of wind effects at this point in time.

### Ice

Tracer tests for frozen conditions were conducted by Smith *et al*. (2005), who concluded that flow conditions "appeared to be good in both ice-covered and unfrozen conditions." The shapes of the tracer responses were very similar, and the volumetric efficiencies were nearly the same, for ice-free and under ice conditions.

### Degradation and Loss

Recovery of tracer is an extremely important quality check for the measured DTD. Failure to recover 100% of the tracer may mean that the tracer adsorbed or was degraded during passage through the wetland. Organic compounds, including the fluorimetric dyes often used for water tracing in mineral systems, are notorious in this respect: they disappear in wetlands. Noninteractive, inorganic substances are to be preferred, lithium ions and bromide ions being two of the more popular ones. Batch microcosm tests can help to establish the degree of interaction between a specific tracer and the wetland in question. Interactive, disappearing tracers produce serious errors in the inferred mixing parameters.

The use of a sorbing tracer can distort the tracer response curve and lead to errors in calculating hydraulic characteristics. An irreversibly sorbing tracer such as rhodamine WT may cause the peak time to be shorter than it really is, whereas a reversibly sorbing tracer will cause a flattening of the DTD and an unrepresentative extension of the tail. Dierberg and DeBusk (2005) reported that the recovery of rhodamine WT declined as the initial concentration of the impulse was reduced.

It is possible that molecular tracers may degrade during travel through the wetland. This is especially true for the organic dyes, such as rhodamine. These may be subject to photolytic decomposition, or to the action of microbes that use the tracer as a carbon source. Irreversible sorption has the same effect, because it removes the tracer from the water, with no chance of return. Additionally, it is possible that the tracer is permanently removed by plant uptake, at least on the time scale of a tracer test. Although such losses cannot be entirely prevented, it is possible to assess its potential effects on the results of the test.

We begin by supposing that the removal is first-order and apportioned equally in all portions of the water column. Therefore, each parcel of water traversing the wetland will lose tracer exponentially with respect to its travel time. We also suppose that the DTD is given by a gamma function, according to Equation B.25. If the rate coefficient is $k$, then the fractional concentration of the tracer exiting with a given parcel between time = $t$ and time = $t + dt$ is

$$\frac{C(t)}{C_i} = g(t)e^{-kt} = \frac{1}{t_i\,\Gamma(N)}\left(\frac{t}{t_i}\right)^{N-1} e^{\left(-t/t_i\right)}e^{-kt} \qquad \text{(B.37)}$$

where

$C$ = tracer outlet concentration, mg/L
$C_i$ = tracer inlet concentration, mg/L
$k$ = tracer loss rate coefficient, 1/d
$t$ = detention time, d
$t_i$ = mean detention time in one tank, d
$N$ = number of tanks

This equation is still a gamma distribution with the same $N$ as that for the nondestructive case. However, the detention time ($t = Nt_i$) has been shortened:

$$\tau_{app} = N\frac{t_i}{1+kt_i} = \frac{\tau_{act}}{1+k\dfrac{\tau_{act}}{N}} \qquad \text{(B.38)}$$

where

$\tau_{act}$ = actual detention time, d
$\tau_{app}$ = apparent detention time, d

Because the apparent detention time has been shortened, the volumetric efficiency will appear to be reduced accordingly. However, the number of TIS is not affected. If the distribution of Equation B.25 is averaged over all parcels, the mass recovery of tracer is given by

$$\text{Fraction recovery} = \left(1-k\frac{\tau_{app}}{N}\right)^N \qquad \text{(B.39)}$$

An example of the effect of tracer loss is given in Figure B.15. This effect is illustrated in the work of Bowmer (1987).

### Retardation

The tracer may also be retained in the wetland, but not lost. This can occur because of the presence of dead zones, either large or small scale, that gain the tracer by diffusion, retain it until the pulse has passed, and then release it back to the flowing water. Reversible sorption on any of several wetland solids can lead to the same phenomenon. Tracer is sorbed on

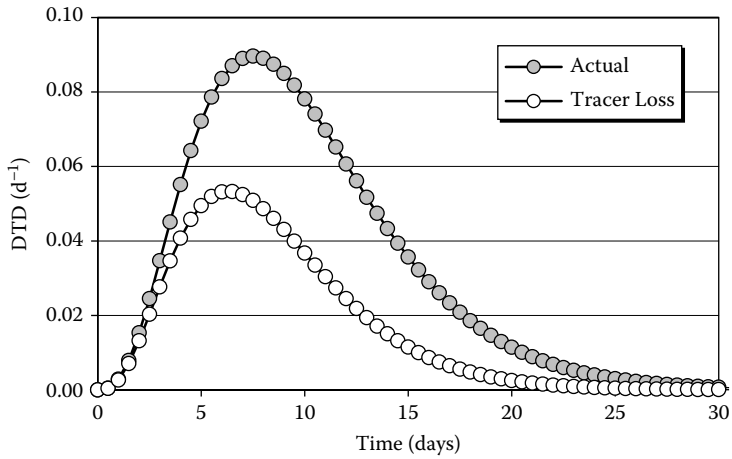



**FIGURE B.15** The effect of tracer loss on the DTD for a wetland with $N = 4$. The actual detention time was ten days, the measured tracer detention was 8.4 days, and the mass loss of tracer was 50%.

the rising limb of the tracer pulse and released on the falling limb.

There are several wetland "compartments" that can serve to store inert solutes. The wetland sediments or media may reversibly sorb the tracer, and this possibility may be qualitatively checked with laboratory tests. However, there may be several other types of "parking places" in the wetland environment. These include stagnant pools of water, pore water associated with litter or clumps of filamentous algae, and the upper layer of soil pore water. Side pools are accessible by wind-driven cross currents. Litter and biomass pore waters are accessible by diffusion. Topsoil pore waters are accessible by diffusion and transpiration flows. All of these mechanisms have two common features: the presence of a storage location, and exchange with the main streams of water. At a simple level, a common model can be used for all.

There are two components to the dynamic model needed to track a tracer with the possibility of storage. The one-dimensional mass balance for tracer for the flowing water is

$$\frac{\partial C}{\partial t} + u\frac{\partial C}{\partial x} = \Phi(C_s - C) \tag{B.40}$$

and the mass balance for storage is

$$\Omega\frac{\partial C_s}{\partial t} = -\Phi(C_s - C) \tag{B.41}$$

where

$C$ = tracer water concentration, mg/L
$C_s$ = tracer storage concentration, mg/L
$t$ = time, d
$u$ = actual water velocity along flow path, m/d
$x$ = distance along flow path, m
$\Phi$ = exchange flow with storage, $(m^3/d)/m^3$
$\Omega$ = storage ratio, $m^3/m^3$

These equations are those used to track pulses of constituents in chromatography, in which delays are due to adsorption. If there is equilibrium between storage and moving water, then $C_s = KC$. We may then add Equations B.40 and B.41, and write

$$(1+\Omega K)\frac{\partial C}{\partial t} + u\frac{\partial C}{\partial x} = 0 \tag{B.42}$$

The factor $(1 + \Omega K)$ is represented as $\rho$, the retardation factor for the system. In ideal circumstances, there would be no storage, and $\rho$ will be 1. However, that will rarely be the case in real wetland environments. The best we can hope for is that $\rho$ is close to unity, and, luckily, this is often the case (see, for instance, Richardson *et al*., 2004).

Equations B.40 and B.41, either in continuous form or discretized, form the basis for the stage models of wetland tracer response. In continuous form, they have been termed the zones of the diminished mixing (ZDM) model, which has been calibrated to many FWS and HSSF wetlands (Werner and Kadlec, 2000a; 2000c). In discrete form, it is termed the finite stage model (Mangelson, 1972; U.S. EPA, 2000a). These will be discussed in more detail in the next section; here, the purpose is to determine the possible effects of storage and exchange on the parameterization of a tracer test.

The cases of water-dead zone storage and solid sorption are somewhat different in that dead water would normally be included in the calculation of nominal detention time, whereas solid sorption storage would not.

For the case of solid sorption storage, the effect of storage is to delay and broaden the tracer response curve. This means that the tracer detention time is lengthened compared to the nominal, and the number of TIS is reduced.

For the case of water storage in dead zones, the effect of storage is to accelerate and broaden the tracer-response curve. That means that the tracer detention time is shortened compared to the nominal, and the number of TIS is reduced.

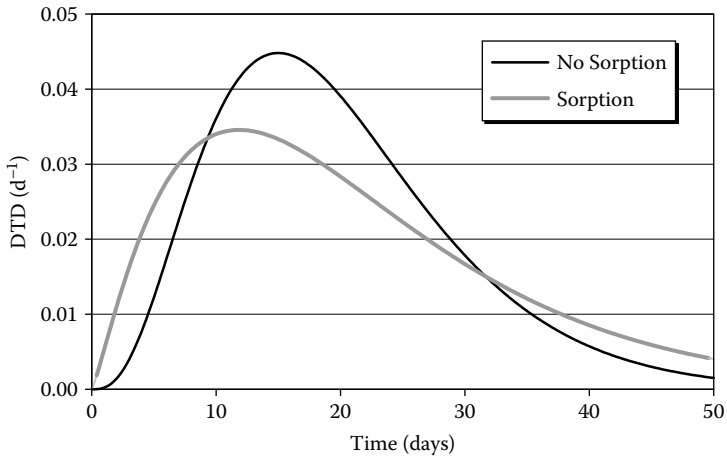



**FIGURE B.16** The effects of reversible sorption on a tracer response for 4 TIS. The "no sorption" line is the forecast result for no sorption. The "sorption" line represents the result for 50% augmentation of water column storage on the solids.

### An Example

These ideas are illustrated by considering a FWS wetland receiving 50 $m^3$/d of water. It is assumed that the system contains an estimated 1,000 $m^3$ of water, computed from stage and bathymetric data. The nominal detention time is easily seen to be 20 days. We further suppose that the system behaves similar to 4 TIS. Equations B.40 and B.41 may be used to assess the effects of added sorption storage on the form of the tracer response. For illustration, suppose that sorption storage provides 50% augmentation of the storage in the water column and the exchange rate of tracer is 50% of the water flow rate. It is found that the measured tracer detention time would be 21.7 days and the system displays behavior characteristic of 2.2 TIS. The nonsorbing and sorbing tracer response curves are shown in Figure B.16.

The illustration is continued for the dead water case. Suppose that dead water storage provides 50% of the storage in the water column and the exchange rate of tracer is 50% of the water flow rate. It is found that the measured tracer detention time would be 13.8 days, and the system displays behavior characteristic of 1.7 TIS. The non-dead-water and dead-water tracer response curves are shown in Figure B.17.

## OTHER FLOW MODELS

Whereas the TIS model described previously is the most popular flow model, other flow models are also used in the wetland literature.

### Plug Flow with Dispersion

Another model uses a dispersion process superimposed on a plug flow model (PFD). Mixing is presumed to follow a convective diffusion equation. Although the PFD model has been advocated for wetlands (e.g., Pardue *et al*., 2000; Wang, 2006), it is doubtful whether it is the most appropriate

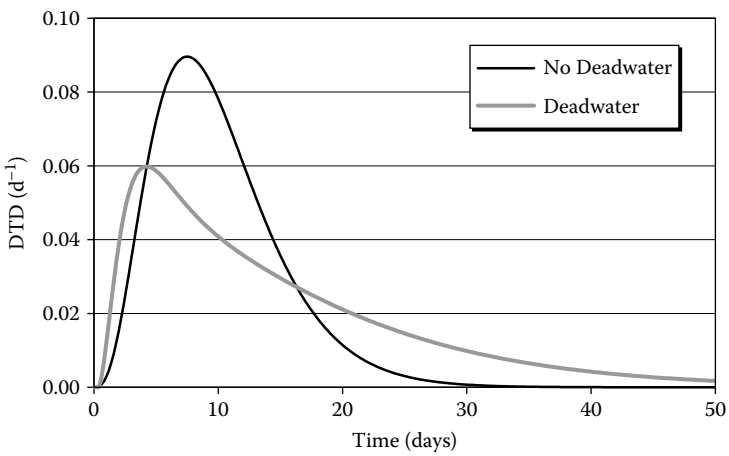



**FIGURE B.17** The effects of dead water storage on a tracer response for 4 TIS. The "no dead water" line is the forecast result for no dead water. The "dead water" line represents the result for 50% of water column in dead zone storage.

model of comparable complexity. The dispersion coefficient describes eddy transport of water elements both upstream and downstream. In FWS wetlands, such mixing may not occur because flow is often predominantly laminar.

A one-dimensional spatial model is chosen because analytical expressions are available for computation of pollutant removal for one-dimensional cases (Fogler, 1992). A two-dimensional version requires a two-dimensional velocity field, which are virtually nonexistent for treatment wetlands. The tracer mass balance equation includes both spatial and temporal variability:

$$D\frac{\partial^2 C}{\partial x^2} - \frac{\partial (uC)}{\partial x} = \frac{\partial C}{\partial t} \qquad \text{(B.43)}$$

where
$u$ = actual water velocity, m/d
$D$ = dispersion coefficient, $m^2/d$
$x$ = distance from inlet toward outlet, m

The appropriate wetland boundary conditions for this mass balance are known as the *closed-closed* boundary conditions (Fogler, 1992). These imply that no tracer can diffuse back from the wetland into the inlet pipe or up the exit structure at the wetland outlet. These are different from the *open-open* boundary conditions that are appropriate for river studies. There are analytical, closed form solutions to the latter case, which have led to their repeated misapplication to wetlands (Bavor *et al*., 1988; Stairs, 1993). There are no closed-form solutions for tracer responses for the wetland case, but numerical solutions to the closed-closed tracer mass balance have been available for more than three decades (Levenspiel, 1972). It is possible to calculate the dispersion constant that fits a particular dataset, although there are issues of accuracy. This model is not advocated here, because the PFD model is only infrequently applicable to treatment wetlands.

The dimensionless parameter that characterizes Equation B.43 is the Peclet number (Pe), or, its inverse, the wetland dispersion number ($\mathfrak{D}$).

$$\mathfrak{D} = \frac{D}{uL} = \frac{1}{\text{Pe}} \qquad \text{(B.44)}$$

where
$\mathfrak{D}$ = wetland dispersion number, dimensionless
$D$ = dispersion coefficient, $m^2/d$
$L$ = distance from inlet to outlet, m
Pe = Peclet number, dimensionless
$u$ = superficial water velocity, m/d

A primary interesting result from the model is the dimensionless variance, which can be written in explicit form for the PFD model:

$$\sigma_\theta^2 = 2\mathfrak{D} - 2\mathfrak{D}^2(1 - e^{-1/\mathfrak{D}}) \qquad \text{(B.45)}$$

The principal problems with the PFD model to wetlands have to do with meeting the assumptions implicit in the model. Levenspiel (1972) notes as follows:

> In trying to account for large extents of backmixing with the dispersion model we meet with numerous difficulties. With increased axial dispersion it becomes increasingly unlikely that the assumptions of the dispersion model will be satisfied by the real system.

The condition of an "intermediate" amount of axial dispersion (or less) should be met in order to apply the PFD model, which is nominally taken to be $D < 0.025$ (Levenspiel, 1972) and corresponds to about 20 TIS. The DTDs for FWS wetland systems are characterized by a large amount of apparent dispersion, with $0.07 \le D \le 0.35$ (Kadlec, 1994). Therefore, generally, neither FWS nor HSSF wetlands are within the acceptable mixing range, although HSSF systems may sometimes be marginally within the range (see Tables 6.1 and 6.2, Chapter 6). However, a bigger obstacle to accepting the PFD model consists of the concentration profiles that are predicted for reactive constituents, because they predict features not seen in treatment wetlands.

The first-order concentration reduction produced by the closed-system PFD model is available in explicit form and is well known (see, for instance, Fogler, 1992):

$$\frac{(C - C^*)}{(C_o - C^*)} = \frac{4b\, e^{(\text{Pe}/2)}}{(1+b)^2 e^{(b\text{Pe}/2)} - (1-b)^2 e^{(-b\text{Pe}/2)}} \qquad \text{(B.46)}$$

$$\begin{aligned} &\text{(a)} \quad b = \sqrt{1 + 4\frac{Da}{\text{Pe}}} \\ &\text{(b)} \quad Da = k\tau / h \\ &\text{(c)} \quad \text{Pe} = uL / D \end{aligned} \qquad \text{(B.47)}$$

where
$D$ = dispersion coefficient, $m^2/d$
$L$ = distance from inlet to outlet, m
$u$ = average superficial water velocity, m/d

This result is typically credited to Wehner and Wilhelm (1956).

Note that there are also two parameters in this reaction model: the rate coefficient $k$ (or, equivalently, the Damköhler number $Da$), and the dispersion coefficient $D$. This formulation has been advocated for wetlands and ponds (Reed *et al*., 1995; Crites and Tchobanoglous, 1998; Shilton and Mara, 2005; Crites *et al*., 2006). However, the longitudinal profiles that this model predicts for large degrees of mixing are not realistic.

## PFD Longitudinal Profiles

Longitudinal profiles may be used to test the validity of alternative modeling assumptions. For instance, the PFD model

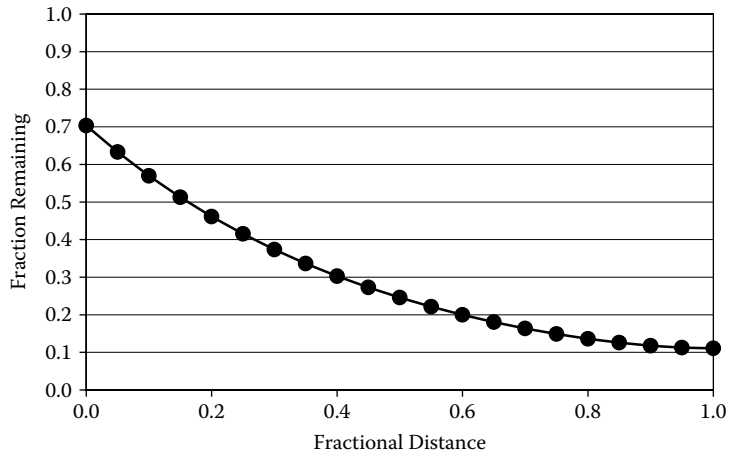



**FIGURE B.18** Fractional amount of reactant remaining for the first-order PFD model for closed-closed boundary conditions. The dispersion is $D/uL = 0.20$, and the reaction rate is $Da = k\tau h = 3.0$.

forecasts the concentration profile through the wetland by

$$\frac{C(x)}{C_i} = \frac{2e^{(x\mathrm{Pe}/2)}\left[(1+b)e^{(b\mathrm{Pe}(1-x)/2)} - (1-b)e^{(-b\mathrm{Pe}(1-x)/2)}\right]}{(1+b)^2 e^{(b\mathrm{Pe}/2)} - (1-b)^2 e^{(-b\mathrm{Pe}/2)}} \tag{B.48}$$

where

$x$ = distance in flow direction, wetland, m
$C(x)$ = concentration at length x, g/m$^3$

The longitudinal concentration profile is predicted to display an instantaneous drop at the wetland inlet. For $\mathfrak{D} = 0.2$ and $Da = 3$, the decrease at the inlet is 30% (Figure B.18). For larger $\mathfrak{D}$ (more dispersion), the instantaneous drop is even larger, increasing to 55% at $\mathfrak{D} = 1.0$. This unrealistically large concentration drop at the very start of the gradient has not been observed in wetland practice and, hence, the PFD model is not an acceptable alternative for most treatment wetland situations.

Interestingly, this predicted profile has only infrequently been examined in any of the literature pertaining to applications to ponds or wetlands, although the PFD model has been extensively utilized. In a pond environment, the situation is often one of considerable mixing because of wind and recirculation currents, and the concept of a sudden concentration change on entry is not unlikely. But in a treatment wetland, this model conceives of swirls that move back into the inlet region. That is a most unlikely scenario in a vegetated wetland environment.

### Parallel Paths

The parallel paths model has been described previously as an example of a method for dealing with the long tails evidenced in some tracers. This model may also be easily coded in a spreadsheet, but there are five parameters: the flow split, plus the detention time, and the number of tanks along both paths. Some authors recommend the two-path response model (Keller and Bays, 2001; Chazarenc *et al*., 2003; Wang, 2006; Wang *et al*., 2006). However, if a two-path model is warranted, then there is something seriously wrong with the wetland. It is the wetland that needs improvement, not the model.

### Finite and Infinite Stages

The breakthrough delay and the long tail may be described by a model that incorporates a plug flow component and side-tank storage in a series of units (Figure B.19). A single stage of

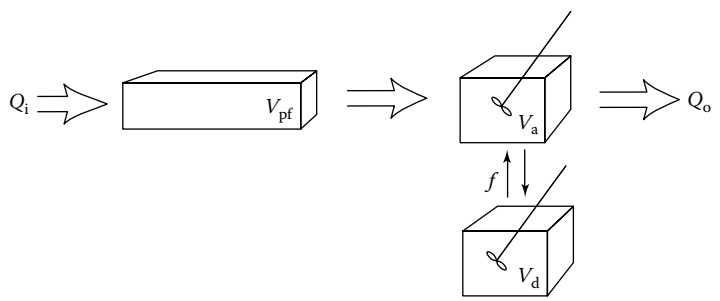



**FIGURE B.19** A single stage of the finite or infinite stage model. A plug flow unit is followed by a well-mixed unit that exchanges material with a side-storage unit.

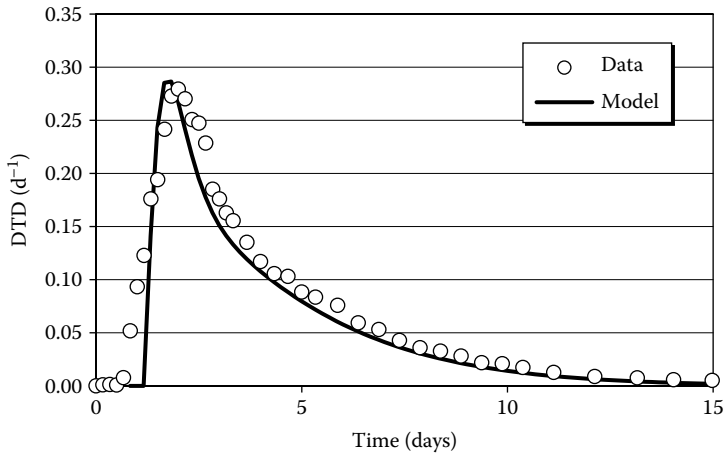



**FIGURE B.20** Finite ($N = 3$) stage model of the tracer response of ENRP cell 4. The tracer detention time was 3.4 days. The plug flow fraction was 20%, active volume 40% and dead volume 40%. The exchange flow ratio was 0.4.

this model consists of a plug flow unit (of volume $V_{PF}$), followed by a well-mixed unit (of volume $V_a$) that exchanges material with a side-storage unit (of volume $V_d$). There are five parameters in this model: the number of stages, the detention time, the fraction plug flow, the fraction active well-mixed, and the exchange flow ratio. The exchange flow between $V_a$ and $V_d$ is $f$, and the ratio to the through-flow is $f/Q$. The finite stage model comprises a small number (2–5) of these stages in series. The infinite stage model utilizes a large number (~100) of the stages.

The concept of the finite stage model was advanced by a number of early authors (Hovorka, 1961; Mangelson, 1972; Levenspiel, 1972). Buffham *et al.* (1970) set forth a family of models that rest on the concept of water elements being delayed a number of times during passage through a packed bed. These are based on the abstraction that water would flow uniformly through the bed (plug flow) if it were not that elements are detained at points during their travel, and elements so detained eventually rejoin the main flow after a time delay.

The finite stage model can provide an excellent fit of data from both ponds and wetlands (Mangelson, 1972; Nolte and Associates, 1998b). An example of the fit is given for cell 4 of the ENRP (Figure B.20). In this case, the response model is supported by the visual tracer study (see Figure B.8), in which the main flow path and side storages may easily be seen.

Werner and Kadlec (2000c) showed that the model could assume a limiting form for a very large number of stages. They termed this the zones of diminished mixing (ZDM) model, which contains four parameters: the detention time, the fraction dead zones, the fraction bypassing, and a dispersion number for the main flow path. Martinez and Wise (2003a) used the infinite stage model via the code of Runkel (1998), which is called OTIS (one-dimensional transport with inflow and storage).

Stage models, by virtue of their one-dimensional character, are not capable of providing a good fit in situations where there is clearly more than one flow path (Martinez and Wise, 2003a). Stage models do not lend themselves to an easy representation for computing pollutant removal. Thus, although they often can do an excellent job of representing the tracer response, they are of limited use in wetland evaluation and design.

## Computational Fluid Dynamics (CFD)

Many treatment wetlands are not linear in their design and assume irregular shapes with irregular bathymetry. Flow distribution may be at a point along the inlet boundary rather than distributed across the width of the wetland. Additionally, vegetation density may not be spatially uniform. Under such circumstances, the internal flow patterns are variable in at least two dimensions, lateral and longitudinal. As with all other models, vertical averaging has always been utilized, although that assumption is questionable under many wetland circumstances. Flow modeling involves solution of mass and momentum balances on a two-dimensional grid, together with a tracer mass balance. This procedure has been utilized, for example, by Walker (1998), Persson *et al.* (1999), Koskiaho (2003), Wörman and Kronnäs (2005), Jenkins and Greenway (2005), Kjellin *et al.* (2006), and Lightbody *et al.* (2007). Several software packages are available for this purpose, including RMA4, MIKE-21, HYDRA3, and HYDRUS2D.

The advantage of CFD modeling is the ability to explore the consequences of alterations in the basin morphology on the volumetric and DTD efficiencies of a system. For instance, the use of point inlets and outlets has the conceptual potential to create dead zones in the corners of a rectangular wetland. This may be tested with CFD, and indeed, the corners are predicted to be excluded from the flow (Figure B.21). The CFD simulations of Koskiaho (2003) indicate that the flow can miss both the inlet and exit corners in rectangular wetlands. For this case, there are field data to confirm the predicted effect (Stairs, 1993; Stairs and Moore, 1994). The FWS wetland had an aspect ratio of L:W = 3:1 and was

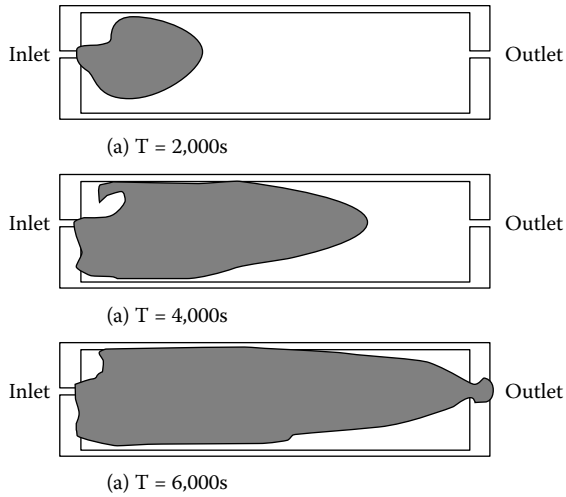



**FIGURE B.21** The penetration of new water into a FWS wetland according to a 2-D simulation model. Note that there may be recirculation and that corners do not "see" the new water. (From Walker (1998) *Ecological Engineering* 10(3): 247–262. Reprinted with permission.)

operated with and without a distribution header across the inlet. Tracer testing indicated a volumetric efficiency of 84% with the header, and 66% without. With the header, *N*TIS = 4.5, and without, *N*TIS = 3.3. The field data indicate an overall hydraulic efficiency $\lambda = 46\%$ for no header and $\lambda = 65\%$ with a header. The simulation results of Persson *et al*. (1999) and Jenkins and Greenway (2005) provide reasonable forecasts of overall hydraulic efficiency $\lambda$ of about 40–45% for unvegetated ponds of similar aspect ratio.

## PREDICTIONS FROM TIS AND CFD FLOW MODELING

The goal for treatment wetland design is to be able to predict the efficiencies of wetlands based on their morphology. Simulation results have been used to provide a few first indications of such relationships.

### Unvegetated Systems

Flat-bottom systems without plants have often been investigated. The effect of length-to-width ratios for ponds on dispersion has been reported by several researchers, and a summary of results is presented by Shilton and Mara (2005). Although the various pond regressions are for deep water environments compared to FWS wetlands, there is a trend, in all investigations, for the dispersion number ($D/uL$) to decrease as L:W increases. In other words, longer and narrower ponds display characteristics closer to plug flow, although still displaying behaviors characterized by a small number of TIS. Jenkins and Greenway (2005) present simulation results for unvegetated FWS wetlands with point inlets and outlets, which may be expressed as

$$e_V = 1 - e^{(-0.3L/W)} \tag{B.49}$$

Thackson *et al*. (1987) presented a similar correlation for data from shallow ponds that substantially agrees with these simulation results (Walker, 1998; Jenkins and Greenway, 2005). Likewise, the *N*TIS value is high for high values of L:W. Therefore, for unvegetated FWS wetlands, there is essentially no advantage in L:W ratios being greater than ten. However, it must be noted that these results are for point inlets and outlets, which are not efficient and probably should not be used.

### Vegetated Systems

The presence of vegetation will markedly change the hydraulics of the wetland by creating less dispersion and higher efficiencies (Kadlec, 1994). For example, the simulation results of Persson *et al*. (1999) showed that the full vegetation of the South Gippsland, Australia, wetland would have increased $e_V$ from 0.39 to 0.69 and *N*TIS from 5.7 to 15, compared to its existing condition. That existing condition was a channel from inlet to outlet, with only fringing vegetation.

The three principal variables associated with vegetation are: (1) percent cover, (2) stem density, and (3) orientation of patches. When vegetation is oriented in the flow direction, typically along the wetland edges, it is termed *fringing vegetation* (Persson *et al*., 1999; Jenkins and Greenway, 2005). When the orientation of patches is across the flow direction, it is termed *banded vegetation*. The simulation results of Jenkins and Greenway indicate that the use of banded vegetation is more efficient than fringing vegetation because of the shortcircuiting issue caused by the flow resistance in the fringes. Results from field studies in Florida (periphyton field cell tracer tests, unpublished data) and California (New River wetlands test results, unpublished data) tend to validate these simulation studies (Figure B.22).

If the vegetation density of a fringe is increased, then more flow is forced down the central channel, and efficiency drops. This was the simulation result of Jenkins and Greenway (2005), and the field result of Dal Cin and Persson (2000) and Stern *et al*. (2001).

### Bathymetry

Deep channels in the direction of flow cause shortcircuiting, whereas deep zones across the flow direction have little effect on flow patterns. The channelization effect is intuitively bad, and has been borne out in both simulation and field studies. It does little or no good to create the channel with meanders, that is, a serpentine deep channel from inlet to outlet (Persson *et al*., 1999).

The use of transverse deep zones in treatment wetlands has been advocated as promoting internal water redistribution and thus the improvement of volumetric and DTD efficiencies (Knight and Iverson, 1990; Knight, 1992; Knight *et al*., 1994; Knight *et al*., 2004). However, simulation results suggest that $e_V$ is little changed by deep zones (0.80 versus 0.81), and $e_{DTD}$ only modestly improved (0.96 versus 0.92) for the Gippsland,



**FIGURE B.22 (A color version of this figure follows page 550)** Rhodamine dye dispersing through an inlet deep zone, with a spikerush band providing resistance to shortcircuiting. (From CH2M Hill (2003a) *PSTA Research and Demonstration Project, Phases 1, 2, and 3 Summary Report*. prepared for the South Florida Water Management District (March 2003).)

Australia, wetland (Persson *et al*., 1999). At the Tres Rios, Arizona, site, 12 FWS wetlands were operated as triplicates of 0, 1, 2, and 3 internal deep zones, and tracer tested (Whitmer, 1998). Analysis of that data via least-squares parameter estimation provided values for $\tau$ and $N$. The resulting efficiencies are shown in Table B.4 for the 12 research wetlands. There was essentially no difference in volumetric efficiency or $N$TIS due to the number of deep zones, and hence none in hydraulic efficiency ($\alpha = 0.05$) (Kadlec, 2007).

Knight *et al*. (1994) operated a set of six treatment wetlands receiving treated paper-pulp-mill effluent for two years. Two internal deep zones comprising 25, 35, and 45% of the area were included in three of the six (Table B.5). Although there was variability in flow (different HLRs) and aspect ratios (2.5, 5, and 10), the authors concluded that there was improvement in nitrogen removal performance due to deep zones. Tracer data from that project are here reanalyzed, and characterized by a tanks-in-series (TIS) detention time distribution. The effect of deep zones was to decrease the number of TIS. However, for the two larger cell pairs, the volumetric efficiency was improved by deep-zone addition (Table B.5).

## Hummocks and Islands

Any wetland feature that protrudes from the water causes a reduction in wetland water volume and surface area compared to the wetland in its absence. Therefore, there is a penalty for such features in terms of the detention time or hydraulic loading. The argument has been made that islands (hummocks are small islands) promote better flow distribution and thus increase the DTD efficiency of the system (e.g., Thullen *et al*., 2005); however, field tests at Tres Rios do not support that contention (city of Phoenix, unpublished data).

According to CFD simulations for shallow ponds, this effect is present in wetlands without vegetation (Persson *et al*., 1999). However, CFD results for the Ekeby, Sweden, FWS wetland led to the conclusion that "According to the simulations, the constructed deep zones and islands had only

**TABLE B.4**
**Hydraulic Efficiency Measures for the Tres Rios Research Cells**

| Internal Deep Zones | Percent Deep | $N$TIS | Volumetric Efficiency ($e_V$) | DTD Efficiency ($e_{DTD}$) | Hydraulic Efficiency ($\lambda$) |
|---|---|---|---|---|---|
| 0 | 12.5 | 3.30 ± 0.79 | 0.56 ± 0.19 | 0.69 ± 0.07 | 0.38 ± 0.11 |
| 1 | 20 | 3.93 ± 0.31 | 0.64 ± 0.20 | 0.74 ± 0.02 | 0.47 ± 0.14 |
| 2 | 28 | 4.00 ± 0.40 | 0.67 ± 0.08 | 0.75 ± 0.03 | 0.50 ± 0.04 |
| 3 | 35 | 3.63 ± 0.06 | 0.65 ± 0.08 | 0.72 ± 0.01 | 0.47 ± 0.06 |
| | **Mean** | **3.70** | **0.63** | **0.73** | **0.46** |

*Note:* Means ± standard deviations.

**TABLE B.5**
**Performance of the Champion Wetlands, July 1991–June 1993**

| Wetland | A | B | C | D | E | F |
|---|---|---|---|---|---|---|
| **Area ($m^2$)** | **4,048** | **4,048** | **2,024** | **2,024** | **1,012** | **1,012** |
| Aspect | 2.47 | 2.47 | 4.94 | 4.94 | 9.88 | 9.88 |
| DZ % | 5 | 25 | 5 | 35 | 5 | 45 |
| Depth (m) | 0.270 | 0.380 | 0.250 | 0.480 | 0.270 | 0.580 |
| HLR (cm/d) | 3.16 | 3.38 | 4.83 | 3.99 | 7.82 | 8.13 |
| Nominal HRT (d) | 8.54 | 11.23 | 5.17 | 12.04 | 3.45 | 7.13 |
| *N*TIS | 4.60 | 3.60 | 4.00 | 3.50 | 10.70 | 2.00 |
| Volume efficiency | 0.63 | 0.81 | 0.43 | 0.82 | 0.91 | 0.74 |
| Hydraulic efficiency | 0.49 | 0.59 | 0.32 | 0.59 | 0.83 | 0.37 |

*Source:* Hydraulic and concentration data are from Knight *et al*. (1994) *TAPPI Journal* 77(5): 240–245. The DTD and *k*-value analyses are from this work.

a minor effect on the residence time distribution" (Kjellin *et al*., 2006). The effect of islands in unvegetated systems is to divert flow toward corner zones that would otherwise not be accessed by the flowing water. However, in vegetated systems, the flow is predominantly directed by the vegetation, and not the bathymetry.

Underwater features, such as cross-flow benches or "speed bumps," have been utilized for the expressed purpose of redistributing flow. According to CFD simulations, this is effective for unvegatated systems (Persson *et al*., 1999). The field experience of Simi and Mitchell (1999) was that speed bumps and deep zones promoted mixing, such that a FWS wetland of L:W = 2 behaved approximately as *N*TIS = 2.

### Peninsulas and Sinuous Flow

Because of the strong effect of the length-to-width ratio that prevails for unvegetated FWS wetlands, it has been surmised that it is very advantageous to utilize high L:W for vegetated systems. Very often, the site boundaries do not allow linear wetlands of great length and, consequently, the strategy of folding the wetland several times is used to lengthen the L:W ratio. Such "back-and-forth" flow paths require the use of internal divider berms separating the passes of the flow path. This configuration is sometimes referred to as a sinuous path system in wetlands, but the equivalent in ponds is the use of baffles. Baffles improve the efficiencies of ponds, according to both CFD and field data. Vega *et al*. (2003) used simulation to predict that a pond of 2:1 aspect would show an improvement of $e_V$ from 74 to 84%, and *N*TIS from 3.94 to 5.5, for the addition of two baffles (18:1 with baffles). Lloyd *et al*. (2003) measured an increase in $e_V$ from 42 to 50%, and *N*TIS from 1.46 to 1.57, for two baffles in a pond of L:W = 4:1 (35:1 with baffles). Thus, improvement is found for the use of sinuous paths for unvegetated systems. However, pond systems often have low *N*TIS and low volumetric efficiency compared to vegetated wetlands.

The L:W ratio increases as the square of the number of passes. Sinuous and nonsinuous vegetated wetlands were run side by side at Listowel, Ontario, but unfortunately, no tracer tests were run (Herskowitz, 1986). However, there were improvements in pollutant removal performance for the sinuous systems. A tracer test was run on a FWS wetland of overall aspect L:W = 1:1, with three divider berms creating a four-pass back-and-forth flow pattern at Hillsdale, Michigan. The nominal flow aspect ratio was thus 16:1. Recovery was 70%, and *N*TIS = 4.3, but the volumetric efficiency was only 20–30%. Evidently flow followed small channels within each pass and failed to reach the majority of side areas.

In contrast, a side-by-side tracer test of a one-pass and a three-pass system of the same overall aspect (5:1) showed major differences (CH2M Hill, 2003a). Recoveries were similar but low (45 and 46%). The nonsinuous system (L:W = 5:1) had *N*TIS = 9 and $e_V = 0.88$, whereas the sinuous system (L:W = 45) had *N*TIS = 25 and $e_V = 0.58$. As a consequence, the hydraulic efficiency of the sinuous system was lower ($\lambda = 0.56$) than that of the nonsinuous system ($\lambda = 0.78$).

These studies are insufficient to draw conclusions about the efficacy of sinuous designs, but do provide preliminary indications that the length-to-width advantage may be overridden by the creation of numerous corner dead zones. Additionally, the divider berms occupy a portion of the footprint and so contribute to a loss of wetland area.

### Compartmentalization

Efficiencies may be increased by subdividing the wetland into a number of cells in series. This has the effect of lowering wind fetch and thus preventing large-scale recirculation. Compartmentalization also causes the water to be collected at each cell outlet, thus blending the high- and low-speed flow elements and then redistributing them.

On a theoretical basis, the effect on detention time is simply to break up the total detention time into pieces proportional to the volumes contained in the component cells:

$$\tau = \sum_{j=1}^{n} \tau_j \tag{B.50}$$

where
$n$ = number of compartments
$j$ = detention time in the *jth* compartment, d

However, the effect on the DTD efficiency is a bit more complicated. For cells in series, the variance of the tracer response for the system is the sum of variances for the individual cells (Levenspiel, 1972):

$$\sigma^2 = \sum_{j=1}^{n} \sigma_j^2 \tag{B.51}$$

where
$\sigma_j^2$ = variance change in the *jth* compartment, $d^2$

This may be related to the detention time and number of TIS for each compartment. From Equations B.22 and B.24,

$$\sigma_j^2 = \frac{\tau_j^2}{N_j} \tag{B.52}$$

where then the system is represented by:
$N_j$ = TIS for the *jth* compartment

$$\frac{\tau^2}{N} = \sum_{j=1}^{n} \frac{\tau_j^2}{N_j} \tag{B.53}$$

or

$$\frac{1}{N} = \sum_{j=1}^{n} \left[ \frac{\tau_j^2}{\tau^2} \right] \frac{1}{N_j} \tag{B.54}$$

In other words, the reciprocal *N*TIS values are additive in proportion to the square of the detention time fractions. If the compartments are all of equal size, then the *N*TIS values are simply additive, but not otherwise. For example, if 90% of the wetland is a compartment that behaves as 4 TIS, and the remaining 10% is a compartment that behaves as 10 TIS, then the system will act like 4.9 TIS.

It may easily be shown that pollutant removal is improved by compartmentalization because it moves the system closer to plug flow. However, for a fixed footprint, the wetland area is decreased because of the interior divider berms, and therefore multiple compartmentalization is not necessarily advantageous.

At the present time, it is necessary to rely on simulation results because there are not side-by-side tracer tests of compartmentalized and noncompartmentalized systems.

## TRACER TESTING PROTOCOLS

### Tracer Selection

Many different tracers have been used for hydraulic analysis of treatment wetlands. Desirable properties are that it is soluble in water, does not react with the pollutants or wetland constituents, occurs at low background levels within the wetland, may be easily analyzed, has low toxicity, and does not influence the flow pattern in a significant way (Headley and Kadlec, 2007). The three most common choices for wetland tracers have been fluorescent dyes and lithium or bromide salts. In FWS wetlands, both dyes and salts have been successfully used. Studies in HSSF and VF wetlands are generally performed using salt tracers.

Radioactive tracers, such as tritium, have very good tracer properties and have been used in constructed wetlands (Crohn *et al*., 2005; Xu *et al*., 2005; Kjellin *et al*., 2006). However, the use of radioactive substances in wetlands is often precluded by regulatory requirements and specific analytical requirements.

### Dye Tracers

Dyes have advantages of low detection limits, zero natural background, and low relative cost. However, they are susceptible to both photodegradation and biodegradation (Dierberg and DeBusk, 2005). This can complicate the analysis of tracer data, as discussed above.

Rhodamine WT is the most commonly used dye, mainly because of availability. Dissolved solids have no effect below about 600 mg/L, and pH has no effect above pH = 6, although the level of fluorescence is temperature sensitive, changing about 2% per degree Celsius (Smart and Laidlaw, 1977). Rhodamine WT is susceptible to biodegradation, photolysis, and adsorption onto organic solids, detritus, and some plastics (Smart and Laidlaw, 1977; Lin *et al*., 2003a; Dierberg and DeBusk, 2005). Due to the above characteristics, rhodamine WT should only be used in short-term tests ($\tau_n$ less than about three days) and within environments that are not highly organic.

Rhodamine WT is typically measured using fluorescence spectrophotometry at wavelength 558 nm and an emission wavelength of 580 nm (Smart and Laidlaw, 1977; Simi and Mitchell, 1999). Other dyes that have been used with varying degrees of success include eriochrome acid red (Bowmer, 1987), and uranine and eosine (Netter and Bischofsberger, 1990; Netter, 1994).

At high concentrations, fluorescent dyes are visible in FWS wetlands. For instance, just a cupful of RWT can easily be tracked through a 1-ha system. It is also feasible to use RWT for visible studies on large wetlands, such as the 147-ha cell 4 of the ENRP (see Figure B.8) (Dierberg *et al*., 2005). Visible studies are useful for establishing blatant shortcircuits, and cross-levee leakage.

### Salt Tracers

Bromide and lithium are the most extensively used ionic tracers, mainly due to their perceived noninteractive character

and ease of analysis. They are added as solutions of sodium or potassium bromide, or lithium chloride, and have yielded reliable results in numerous wetland studies (Bowmer, 1987; Netter, 1994; King *et al*., 1997; Rash and Liehr, 1999; Drizo *et al*., 2000; Grismer *et al*., 2001; Keller and Bays, 2001; Tanner *et al*., 2002a; Lin *et al*., 2003; García *et al*., 2004b; Smith *et al*., 2005). Lithium and bromide are not susceptible to degradation but are capable of being taken up by wetland plants and other organisms to a limited extent (Kung, 1990; Whitmer *et al*., 2000; Xu *et al*., 2004; Parsons *et al*., 2004). Perhaps more serious is the loss of any tracer, inert or not, caused by the movement of water from the water column to the rhizosphere due to the transpiration flux. Tracer is drawn into the root zone, where it is effectively lost during the relatively brief duration of a tracer experiment.

Background concentrations of lithium are typically very low; for instance 30 ± 14 µg/L (mean ± s.d.; range 4–50 for 20 FWS systems). Bromide may be present in natural waters at concentrations well above detection; for instance 420 ± 760 µg/L (mean ± s.d.; range 5–3,500 for 20 FWS systems). Bromide is typically analyzed via ion chromatography, although specific ion electrodes are available. Lithium can be measured using atomic absorption spectrometry (AAS) or inductively coupled plasma–optical emission spectrometry (ICP-OES).

Chazarenc *et al*. (2003) used high concentrations (67,000 mg/L) of sodium chloride solution as a tracer in a HSSF and measured conductivity in order to derive DTDs. They reported tracer recovery rates exceeding 78% from eight individual tracer studies. Sodium chloride can represent a relatively inexpensive option that can be easily monitored and logged in the field using electrical conductivity probes. However, high salt concentrations may have a negative effect on wetland biota and possibly treatment. Wörman and Kronnäs (2005) used potassium iodide as the tracer at the Alhagen, Sweden, wetland.

## Amount of Tracer

The mass of tracer in the impulse addition is based on the expected dilution, the method detection limit of the analytical techniques, and the background concentration of the tracer in the wetland system after dilution. During the introduction of the tracer impulse into the influent stream, the concentration should be considerably higher than both the analytical method detection limit and background concentration of tracer in the wetland. A target for the peak effluent concentration in the response might therefore be at least 20–50 times the background concentration, and at least 50 times the detection limit.

It is perhaps easier to plan in terms of the average concentration desired in the response curve. The peak-to-mean ratio (ratio of the peak tracer concentration to the mean tracer concentration over a period of three detention times) is typically 2–3 for FWS wetlands, and 4–5 for HSSF wetlands. The benchmark concentration can be defined as the mass of tracer added divided by the nominal volume of water in the wetland:

$$C_{\text{bench}} = \frac{M}{V_{\text{n}}} \tag{B.55}$$

where

$C_{\text{bench}}$ = benchmark concentration, g/m$^3$ = mg/L
$M$ = mass of tracer added, g
$V_{\text{n}}$ = nominal wetland water volume, m$^3$

The reader is reminded that the nominal wetland water volume for subsurface flow wetlands must take into account the porosity of the granular medium in the wetland bed. This benchmark concentration will be approximately 30–70% of the peak concentration. Keller and Bays (2001) suggest that the quantity of tracer added, if assumed to mix uniformly throughout the volume of the wetland, should achieve a benchmark concentration at least 10 to 20 times the background concentration.

## Density Stratification Problems with Salt Tracers

Tracer materials are most often available as either solids (e.g., NaBr, LiCl) or as concentrated solutions. These as-received materials are usually dissolved or diluted to form the tracer solution to be added, which is here designated to have concentration $C_{\text{dose}}$. Care should be exercised in preparing a solution to be dosed into the wetland because there can be large heat-of-solution effects upon dissolution. Most tests are conducted as impulse tests, which implies that the tracer should be dumped into the wetland inlet structure all at once. However, at high $C_{\text{dose}}$ concentrations, the density of the dosing solution may be substantially higher than that of the ambient water in the wetland to which it is added, causing the tracer pulse to sink to the bottom of the wetland.

In FWS wetlands, this situation is exacerbated by the presence of an inlet deep zone that serves as a repository for the dense slug of tracer. This effect is worsened by the transpiration flux in a FWS wetland, which draws water selectively from the wetland bottom and consequently may "pull" the dense tracer from the bottom of the water column into the soil matrix. Under such circumstances, tracer recovery will be poor. Although the typical interpretation of FWS wetland water flow is vertically averaged, water moves more slowly across the wetland bottom due to drag forces. This can cause the flow of the tracer through the wetland to be retarded as it slowly creeps along the bottom and stagnates in depressions. Consequently, density stratification can lead to a significant distortion in the DTD (Schmid *et al*., 2004a; 2004b).

To avoid the density stratification issue, the tracer concentration should not be so high as to result in a significant difference in density between the ambient wetland water and the tracer impulse when it is introduced into the wetland. For typical tracers salts, at 25°C, density equations are as follows (Söhnel and Novotny, 1985); all $R^2 \geq 0.9995$:

Lithium chloride $$\rho = 1.0000 + (5.0327 \times 10^{-7}) C_{\text{dose}} \tag{B.56}$$

Sodium bromide $$\rho = 1.0000 + (7.2869 \times 10^{-7}) C_{dose} \quad \text{(B.57)}$$

Potassium bromide $$\rho = 1.0132 + (6.9267 \times 10^{-7}) C_{dose} \quad \text{(B.58)}$$

Sodium chloride $$\rho = 0.9986 + (2.6123 \times 10^{-7}) C_{dose} \quad \text{(B.59)}$$

where

$C_{dose}$ = concentration of tracer dose solution, $g/m^3$ = mg/L

$\rho$ = density of tracer dose solution, $g/cm^3$

It is known that, in lakes, a density difference of about 1% is enough to maintain stratification. In their teaching lab experiment to demonstrate this effect, Wetzel and Likens (1991) use a density difference of 1.5%. It is therefore not surprising that Chazarenc *et al*. (2003) observed tracer stratification in full-scale HSSF systems at a density difference of 3.4%.

There is also a body of anecdotal evidence that implicates density effects in anomalous tracer tests. For example, several LiCl tests were run by the South Florida Water Management District on test cell wetlands with instantaneous dosing with 78,000 mg/L lithium chloride. The density difference would have been on the order of 3.9%. Recoveries were quite variable, with some as low as 23%. Rousseau *et al*. (2006) reported "losing" a dose of high-concentration lithium in an SSF wetland pilot but finding it residing on the wetland bottom. The studies of Whitmer *et al*. (2000) displayed very low recoveries of sodium bromide, impulse-dosed into deep zones at about 200,000 mg/L. The estimated density difference would have been approximately 14.6%. Although the loss was attributed to plant uptake, it seems highly probable that the dense bromide accumulated in the inlet deep zones and stayed there. Schmid *et al*. (2004a; 2004b) performed detailed experiments using salt tracers in shallow-water flumes, intended to emulate FWS wetlands. The presence of plastic dowels in the water column (which were used to represent plants) assisted in the prevention of stratification. Nonetheless, stratification in these idealized laboratory channels was found to exist for concentrations on the order of 150 mg/L of sodium chloride. Drizo *et al*. (2000) dosed HSSF mesocosms with potassium bromide at 20,000 mg/L, which has a density 1.2% greater than water. The tracer was found on the bottom of all mesocosms, whether planted or unplanted.

Density stratification may be ameliorated by aeration of the wetland. An aerated HSSF wetland in Anamosa, Iowa, was dosed with 142,000 mg/L potassium bromide (11.2% more dense than water) (Nivala *et al*., 2004). The measured recovery of 85% and volumetric efficiency of 91% were indicative of lack of stratification (Nivala, 2005). Nonetheless, at 25% of the bed length, the concentration peak in the top layer (15 cm) was five times higher than that in the deep layer (45 cm).

Based on these findings, it is suggested that the density of the tracer impulse should be a great deal less than 1% of the density of the ambient wetland water in order to minimize the risk of density effects. Clearly, it is also useful to promote as much turbulence as possible at the point of water introduction so that the tracer is further diluted by the primary inlet water flow.

Fortunately, the density difference problem can also be alleviated by prolonging the period of tracer dose introduction. This effectively dilutes the dose concentration with incoming water prior to the wetland. In effect, the impulse is replaced by a "square wave" tracer input. The clock for the test is set to zero at the midpoint of the inlet tracer delivery time series. There will then be no error in the computation of the tracer detention time. The variance of the tracer outlet response is adjusted by subtracting the variance of the inlet pulse (Levenspiel, 1972). In this way, the dose concentration may be diluted by a very large factor.

### Sampling Frequency and Duration

A quantitative tracer test requires definition of the full response curve, from the pretest background through the rising and falling limbs of the distribution and back down to a posttest plateau. This requires sampling for a long-enough period and at a suitable frequency to adequately describe the tracer response curve. Thus, an estimate of the likely DTD of the wetland should be made based on nominal parameters to design a sampling regime. Important aspects of the DTD to be considered when determining sampling frequency are the steeply rising limb of the peak concentration profile and the long, declining tail. However, tracer responses often cannot be accurately forecast. It is therefore prudent to collect and store more samples than anticipated to be needed because lithium and bromide do not degrade during storage. These supplementary samples can then be analyzed retroactively if they are deemed to be necessary based on analysis of the primary sample set.

About 30–40 sample points are normally adequate to define the response curve spread over about three nominal detention times. A minimal sampling frequency of 24 points is illustrated in Figure B.23. These are more densely spaced in the early portion of the response to capture the rise in concentration. A lesser frequency is needed far out on the tail. Where the flow rate is somewhat variable (stormwater wetlands, for instance), flow-weighted sampling will generally be the best way to ensure that accurate sampling of the tracer response curve is achieved. In any case, it is necessary to record both the time of sampling and volume of water that has passed through the sample point, as these are required in interpreting the variable-flow tracer-response data. The use of automatic samplers is of great benefit. They can be configured to collect samples at regular time intervals, or alternatively to collect samples on a flow-weighted basis.

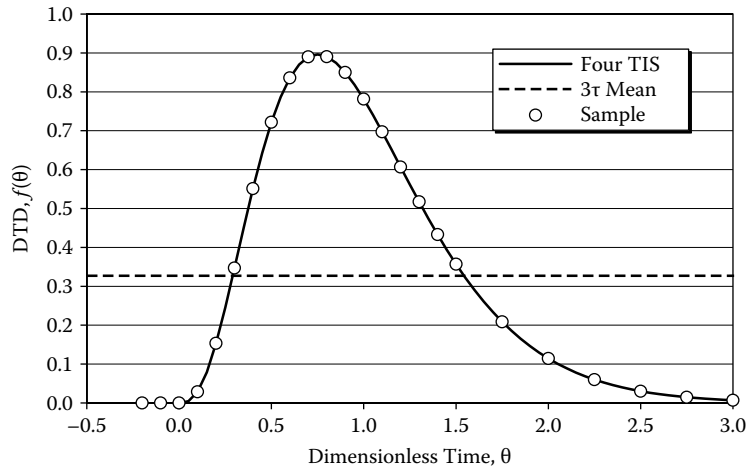



**FIGURE B.23** The concentration of the peak of a 4-TIS response is 2.74 times the mean concentration over three detention times.

Sampling should begin immediately prior to tracer injection to identify the background concentration at the time of the tracer study. The greatest sampling frequency is required at times when the concentration is changing most rapidly (that is, around the time of the peak in concentration). The rise typically begins at 10–30% of $\tau_n$. The peak will typically occur at 50–90% of $\tau_n$.

Following the expected peak in tracer concentration, the sampling frequency can be progressively decreased. To adequately capture the declining tail of the DTD sampling is typically required for an extended period. Examination of tracer responses from the numerous available experiments indicates that the impulse response is typically complete after four $\tau_n$. In general, care should be taken not to allow too much volume to exit without being sampled. Otherwise, a spike or dip in the concentration could be partially or even completely missed, particularly at high flow rates (Werner and Kadlec, 2000c).

## Hydrologic Measurements

To quantify the results of a tracer test, the complete hydrologic conditions must be known. The flow rates entering and exiting the wetland should be continuously logged over the duration of the tracer study. Rainfall can be important, and should be measured at the site. Evapotranspiration and seepage may be estimated. These combine to give the wetland water budget. In the context of a tracer test, the flows are necessary to compute the tracer mass balance and the recovery of tracer in the outflow. Effects of rain and *ET* are usually small, but not always (Chazarenc *et al*., 2004).

It is important to know the wetland water volume because it is required to compute the nominal detention time ($\tau_n$). In turn, $\tau_n$ is used to calculate the volumetric efficiency of the wetland $e_V = \tau/\tau_n$. For FWS wetlands, the bathymetry is used to obtain the water volume. As noted in Chapter 2, these measurements are somewhat ambiguous because, typically, the wetland bottom is not well defined. For FWS wetlands, the water volume requires knowledge of the upper water surface and the porosity of the media. That porosity may be accurately known for the bare media, but the extent of pore blockage by roots, biofilms, and accreted solids is typically not accurately known.

## Internal Sampling

In addition to inflow/outflow points, the tracer response at interior wetland points can be of some use in assessing the spatial characteristics of the wetland. There are two distinct cases to be considered: the individual cells in a multicompartment wetland, and points interior to an individual cell. The difference is considerable because, in the former case, the water volumetric flow rate past the measurement is known, whereas in the latter case it is not. Thus, it is possible to get a flow-weighted concentration at a cell outlet, whereas it is not possible at an interior point.

For cell networks with single inlets and outlets, there is no ambiguity, and efficient cells may be distinguished from inefficient cells. If there are multiple inlets and outlets, then averaging assumptions must be made, but analysis is still unambiguous. Perhaps the most involved cellwise tracer testing was performed at the Orlando Easterly treatment wetland, where the first 15 of 17 cells were tracer tested (Martinez and Wise, 2001) (see Figure B.24). These cells varied widely in their hydraulic behavior (see Table B.1). Results of the testing confirmed the need to re-establish the bathymetry of several cells (Sees, 2005).

Another multicell wetland tracer study was conducted at Lakeland, Florida (CH2M Hill, 1999; Keller and Bays, 2001). Four cells in series, of 81, 77, 166, and 30 ha, were tested separately, with recoveries of 67–106% and volumetric efficiencies of 0.04–0.45. Cell 2 had $\tau = 0.7$ d, and a $\tau_n = 19.1$ d, indicating very serious shortcircuiting. In such cell studies, it is feasible to measure the individual structure flows, and test protocols are the same as for single-cell wetlands. However, if a single pulse is to be followed through multiple cells in series, analysis must recognize that detention times (first moments) and variances (second moments) are additive (Levenspiel, 1972).

Both FWS and HSSF wetlands may also be sampled for tracer response at internal points, along and across the flow

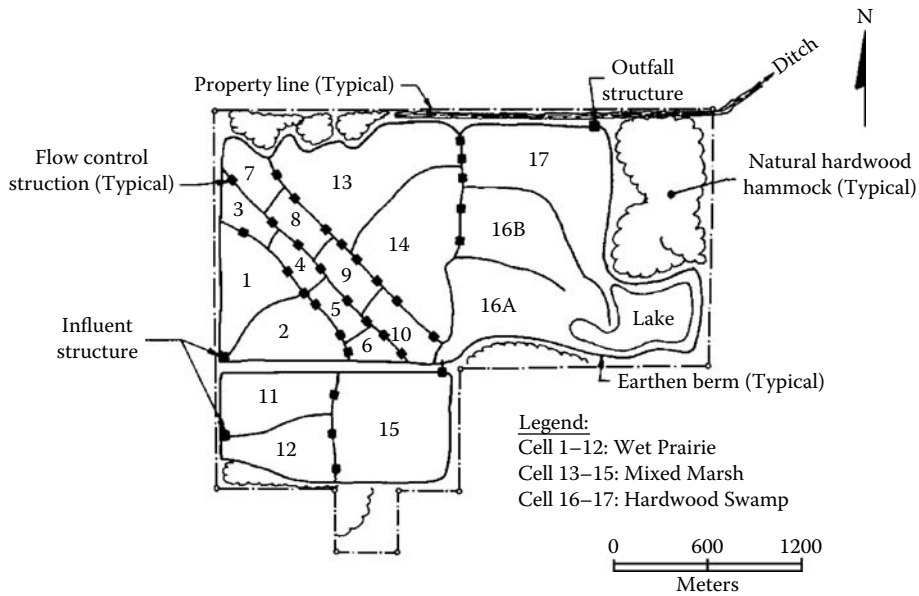



**FIGURE B.24** The layout of the Orlando Easterly treatment wetlands. The first 15 cells were tracer tested. (From Martinez and Wise (2003b) *Journal of Environmental Engineering* (*ASCE*) 129(6): 553–560. Reprinted with permission.)

direction, and at varying depths. In contrast to cell studies, the flow fraction passing such internal points is unknown.

Internal tracer studies on FWS have been reported by Kadlec (2001a), Jawitz and White (2002), Crohn *et al*. (2005), Dierberg *et al*. (2005), and Harvey *et al*. (2005). These studies confirm some intuitively obvious phenomena. The Des Plaines River wetlands studies showed that the shallow and backwater areas of irregular FWS systems saw relatively less tracer than the main flow path (Kadlec, 2001a). Jawitz and White (2002) found that vegetated barrier strips perpendicular to flow produced only small variations in tracer arrival times across the flow direction, regardless of high (aspect ratio = 45) or low (aspect ratio = 5) length-to-width ratio. Crohn *et al*. (2005) also saw only small variation in arrival times in a sequence of cross-deep zones at the Hemet, Californiia, wetland. However, remnant ditches and roads in cell 4 of the ENRP caused short-circuiting that was visible via rhodamine (see Figure B.8) but also quantified by quantitative analysis of the dye (Dierberg *et al*., 2005). Harvey *et al*. (2005) measured water velocities and conducted tracer studies to obtain the volume average water velocity. They found that the average velocity was only about two thirds of the spatial average due to the preferential velocity measurements being in open water zones.

Internal tracer sampling for HSSF systems was done in the researches of Spangler *et al*. (1976b), Fisher (1990), Pilgrim *et al*. (1992), Marsteiner *et al*. (1996; 1997), Drizo *et al*. (2000), García *et al*. (2003a; 2003b; 2004b), Melton (2005), and Nivala (2005). These studies provide a general picture of tracer preferentially moving through the bottom layers of gravel, regardless of whether it arrived there due to density or distribution (Figure B.25). When the bed floods, tracer testing indicates that almost all the flow is overland across the top of the gravel (Spangler *et al*., 1976b).

Flow in HSSF beds is typically nonuniform from side to side, as seen in Figure B.2. Time-series sampling at multiple points across the flow path often shows this imbalance in the form of quite dissimilar tracer responses (for instance, see Chapter 6, Figure 6.17). Studies by Pilgrim *et al*. (1992), Marsteiner (1997), Melton (2005), and Nivala (2005) show differing degrees of internal tracer response imbalance. However, internal patterns of tracer response are of no quantitative value in analyzing wetland behavior because of the lack of flow information to accompany them.

The old literature on tracer analysis describes internal average sampling (cross-transects) as "through the wall" sampling (Levenspiel, 1972). In the wetland context, this corresponds to arithmetic averaging of samples taken at a number of points perpendicular to the flow direction (Kadlec, 2000). Averages of tracer concentrations taken across the flow direction are not the same as the flow-weighted concentration at that point in the overall travel path; in fact, they are often very much different (Levenspiel and Turner, 1970). As described in Chapter 2, different sampling points across the flow path led to very different internal tracer response curves at the Benton, Kentucky, HSSF wetland. The same result was found at a Texas on-site facility (Melton, 2005). Thus, spatially averaged internal cross-sampling is likely to be valid only if done in a deep zone that ensures complete mixing of the various flow elements (Crohn *et al*., 2005).

## Analyzing Tracer Data

The goal of tracer studies is to understand the volumetric and DTD efficiencies of a particular wetland. Although the methods of processing data are straightforward, there are some nuances of data utilization that require explanation.

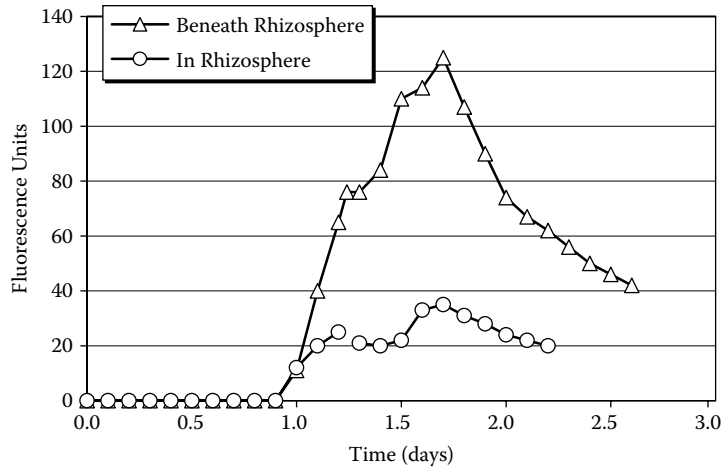



**FIGURE B.25** Tracer responses for locations in and beneath the rhizosphere in a HSSF wetland. (From Fisher (1990) In *Constructed Wetlands in Water Pollution Control*. Cooper and Findlater (Eds.), Pergamon Press, Oxford, United Kingdom. Reprinted with permission.)

## Baselines and Tails

In all cases, there will be a background concentration of the tracer in the wetland water. The value is not necessarily constant, but may vary during the test either due to random processes or to a (slight) time trend. Thus, the background concentration after the test may not be the same as before the test, either due to random effects or extremely slow bleed-back of some small fraction of the test material. In other words, the tail of the response may not come back down to the starting background concentration. An example is shown in Figure B.4 for Cell 2 of the ENRP in Florida. Over 2 tons of lithium chloride were added to this 167-ha FWS wetland, with care taken to prevent density effects. The initial lithium in the wetland outflow was about 8–9 μg/L. After the test, some 35 to 45 days later, the outflow lithium appeared to no longer have a trend, and was 12–13 μg/L. It is not known if the higher background is due to added tracer remaining after very long times, or due to other causes.

It is intuitive that the initial background should be subtracted from the measured concentrations to keep track of the added lithium. For the ENRP tracer study, the corrected recovery of tracer for this test was 98%. However, the fact that the response did not return to the original baseline has a fairly large effect on the computed tracer detention time. The small concentrations of tracer that occur at long times are disproportionately important in the numerical evaluation of this summation, as seen in Table B.2. In the ENRP example, the tracer detention time is 13 days, calculated over the entire 47 d of the test, and about 12% of that is due to the material leaving in the plateau at 35–47 days. This "tail error" has long been known to possibly seriously affect tracer test results (Curl and McMillan, 1966).

There are two procedures that have been used to de-emphasize the long-term plateau effect. First, the tail of the response ought to be exponential, according to the generally accepted modeling results (Vereecken *et al*., 1999). Therefore, the tail of the response maybe fitted with an exponential function, relying upon the higher concentrations prior to the tail to set the rate of decline. This procedure has been followed by some wetland researchers (Dal Cin and Persson, 2000; Wang, 2006). Often, the "tail" of the measured exit concentration distribution is poorly defined. Sometimes the sampling is terminated too soon; sometimes the final baseline does not return to the starting zero. Under these circumstances, the tail may be determined as an exponentially decreasing function, extrapolated from the data points past the second inflection of the response.

The alternative procedure is to perform a least-squares fit of the selected DTD to the data, which places equal emphasis on the errors across the entire test. The result of this procedure for the ENRP Cell 2 example is shown in Figure B.4.

## Tracer Mass Recovery

Closure of the tracer mass balance is a prerequisite to a good tracer test. Large losses or apparent gains of tracer, compared to the amount dosed, are a signal that something is awry with the water flows, or that the chosen tracer is undergoing extensive nonconservative behavior (Kadlec, 1994). The measure of the tracer test reliability is the recovery percentage. Outflow rates and tracer concentrations are combined and summed to compute the total mass in the exit tracer pulse:

$$M_o = \int_0^\infty Q_o C(t) dt = ? = M_i \qquad \text{(B.60)}$$

where

$C(t)$ = exit tracer concentration, g/m$^3$ = mg/L
$M_i$ = mass of tracer introduced with inflow, g
$M_o$ = mass of tracer exiting with outflow, g
$Q_o$ = average outflow rate, m$^3$/d

This permits a tracer mass balance and computation of the percentage tracer recovery. The concentrations used in this summation are first corrected for background. Secondly, if there are anomalies with the measured concentrations in the tail zone, or if the tail has been missed due to premature cessation of the test, then an exponential "patch" may be applied. Alternatively, a least-squares fit may be conducted. Typically, treatment wetland tracer recoveries in the range of 80–120% are considered acceptable.

## POLLUTANT REMOVAL EFFECTS

A primary reason for tracer analysis is to be able to account for hydraulic effects and thus reduce that part of the variability in pollutant reductions. First and foremost, it is desirable to know the real retention time, or equivalently the volumetric efficiency. Secondly, the distribution of detention times is important via the DTD efficiency.

The models discussed in this appendix are all concerned with correctly describing the behavior of an inert material as it passes through the wetland. However, the important target is correctly describing reactive materials. To understand the implications for DTD model selection, a hypothetical case is considered. Suppose that the wetland behaves like four well-mixed units in series, and suppose that the contaminant in question follows first-order kinetics and has a zero background concentration. For this illustration, the amount (concentration) of pollutant remaining at the outflow will be considered as the measure of performance.

The first questions are: Does it matter if there are four TIS? Isn't plug flow good enough? To answer those, a variety of rate coefficients and detention times may be considered for the two cases. It is the product $Da = k\tau/h$ that controls both cases. Figure B.26 shows that, for small $Da$, which corresponds to small removal of the contaminant, there is not much difference between models. If $Da < 1$, for which there is more than 40% of the material remaining, there is only a maximum of 10% difference. Either model could be effectively used if performance was to be in this low range.

However, if removals in excess of 90% are contemplated, then there are very big differences between the plug flow and 4 TIS forecasts. The ratio of fractions remaining grows from a factor of two at 90% 4 TIS conversion to a factor of twenty at 98% 4 TIS conversion. Therefore, if high removals are contemplated, it is very important to correctly characterize wetland internal hydraulics.

### The Tail of the DTD

The next issue is: How careful do we have to be in describing the details of the tracer response? Does it matter if the tail end of the DTD is accurately modeled? To answer these questions, we consider the origins of the unreacted material in the wetland outflow. If there are shortcircuits, they will carry the contaminant to the wetland outlet before they have had much time to react. If there are dead zones, water will remain there a long time and be fully reacted. These ideas may be quantified by considering the detention times for the unreacted materials. We therefore continue the illustration and calculate the fraction of the unreacted effluent material corresponding to different detention times. This is the same calculation as for degrading tracer that was considered earlier. Figure B.27 shows the travel time distribution of the unreacted material leaving for $Da = 2$. The material leaving primarily followed short detention paths: 98% of the leftovers came from detentions less than 1.5 times the average detention time, and 85% from detentions less than 1.0 times the average detention time. The tail is irrelevant to the determination of the amount of unreacted material.

The example chosen is representative of only one wetland hydraulics and one operating condition. However, exploration of the TIS model shows that the conclusion of tail irrelevancy is strengthened for lower numbers of TIS (TIS < 4), and

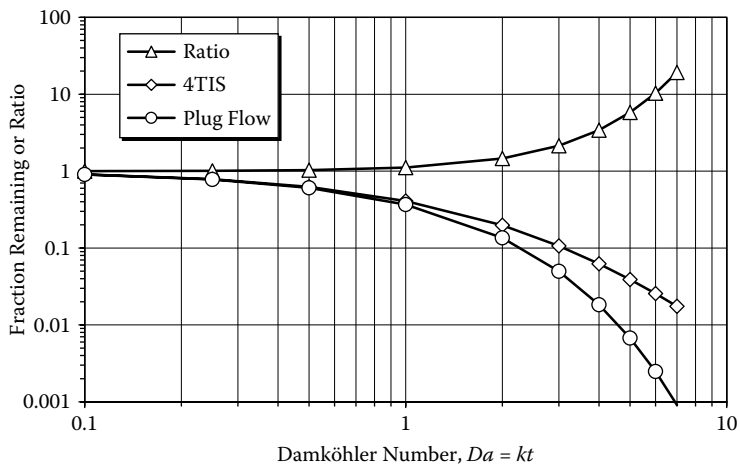



**FIGURE B.26** Variation of plug flow and 4 TIS model results for various rate constants.

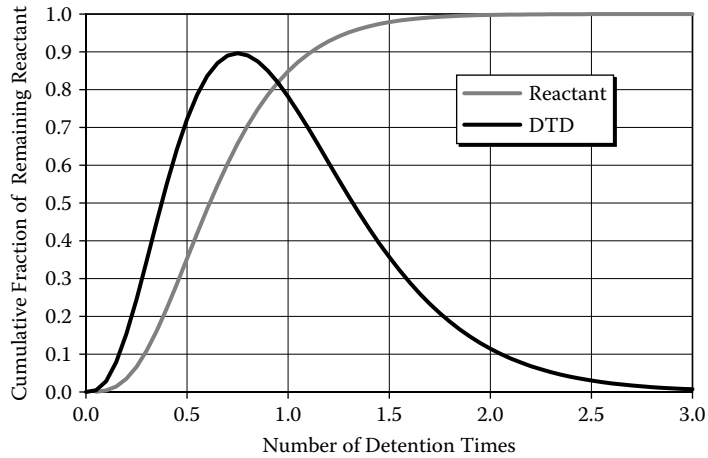



**FIGURE B.27** Origins of the unreacted material leaving a hypothetical 4-TIS wetland for $Da = 2$. The removal is 86.5% under these conditions, and the remaining reactant cumulative distribution refers to the remaining 13.5%. The DTD is scaled by a factor of ten.

also for non-TIS DTDs with longer and bigger tails. Higher Damköhler numbers, equivalent to higher removal percentages, shift the origin of unreacted material to even shorter detention times. For $Da = 1$, or 60% unreacted material, 99% of the leftovers come from detentions less than 2.0 times the average detention time, and 75% from detentions less than 1.0 times the average detention time.

## The Head of the DTD

The front end of the tracer response represents what is generically called *shortcircuiting*. In pathological situations, a large peak occurs far in advance of the nominal detention time. However, even in "normal" situations, it is often necessary to correctly account for the water and associated pollutants that leave the wetland early.

Given the insensitivity of pollutant removal to the tail of the DTD, it is logical to ask whether the head of the DTD is critical to pollutant removal. There is usually a delay of some magnitude before any tracer arrives at the wetland outlet, which is larger for HSSF wetlands than for FWS wetlands (see Figures B.5 and B.6). For 35 FWS wetland tracer tests, the first tracer breakthrough occurred at 23 ± 11% of the tracer detention time. For 32 HSSF wetland tracer tests, the first tracer breakthrough occurred at 37 ± 14% of the tracer detention time.

For some guidance, we return to the 4-TIS illustration. It is found that, for low rates of reaction, the early-departing material carries only a small fraction of the unreacted total. For FWS behavior, and for $Da = 1$, or 60% unreacted material, only 2% of the leftovers come from detentions less than 0.25 times the average detention time. For HSSF behavior, and for $Da = 5$, or 4% unreacted material, 48% of the leftovers come from detentions less than 0.4 times the average detention time. It is clear that care must be taken to provide a proper representation of the leading portion of the DTD if it is necessary to characterize high removals.

The two simplest choices for describing the breakthrough lag are the TIS model, and the TIS model modified to have a leading plug flow component (see Equation B.27).

In general, this three-parameter ($N$, $\tau$, $t_D$) fit of the DTD will give a better fit to data than the two-parameter TIS fit, especially if there is a substantial delay. However, it adds some complexity to pollutant removal modeling, and so it must be examined for necessity. As an illustration, the basis for comparison is taken to be a wetland that behaves like 4 TIS after a time delay. The "head effect" is known to be most important for high removal percentages, so the Damköhler number is selected to be $Da = 3$, corresponding to 95% removal in plug flow, or 5.0% remaining. For 4 TIS, the percent remaining is 10.65%. The TIS model appears to give a reasonable least-squares fit to the delayed tracer response (Figure B.28), but it does turn up a bit sooner than the delay. Because of the delay, the number of TIS grows with the delay, from $N = 4$ for no delay to $N = 17.9$ for a delay of 50% of the detention time.

The amounts of pollutant remaining also differ for the delayed "data" and the least-squares *N*TIS fit. The *N*TIS fit produces greater amounts remaining: 7% more for a delay of 23% of the detention time (FWS), and 10% more for a delay of 37% of the detention time (HSSF). Therefore, if the wetland is contemplated to remove less than 95% of a contaminant, it is probably acceptable to use the two-parameter *N*TIS fit rather than the three-parameter delayed *N*TIS fit of the tracer response.

It is also possible to use the moment method for fitting the *N*TIS model. Earlier it was seen that this method emphasized the tail of the DTD, but it also emphasizes the head of the DTD. Of course, that is exactly what is needed to better account for the delay.

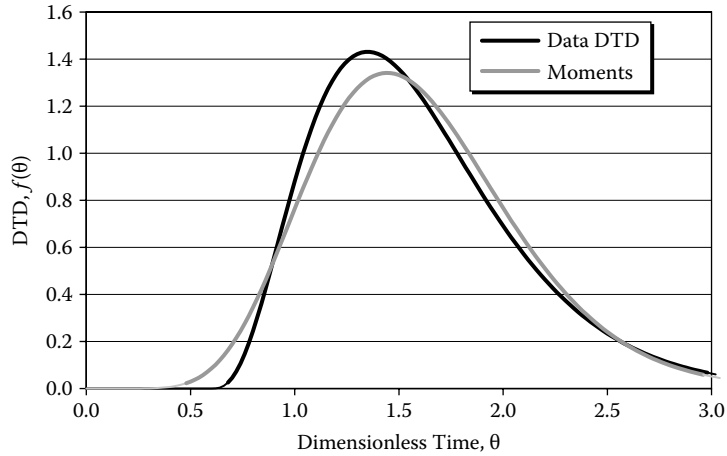



**FIGURE B.28** Tracer response with a delay. The points were generated from a delay of θ = 0.5 together with a 4-TIS response. The tracer detention time is θ = 1.5. A least-squares fit of the TIS model gives $N = 9.7$ for this case.

### Strategies for Data Fitting

The preceding model exploration exercises lead to suggestions regarding the strategy of fitting and using tracer data. These are

- The "tail" of the tracer DTD is not generally an important contributor to the amount of a pollutant that will pass through the wetland unreacted. It is probably not necessary to add parameters to tracer models to better account for long "tails."
- The "head" of the tracer DTD is important for situations where high contaminant removals are contemplated. The important feature is the existence of a delay before any tracer is observed in the wetland outflow. The presence of such a delay contributes to a better pattern of flow by virtue of adding a plug flow component to the response. Typically, for high removals, the contaminant residual is reduced 25% in FWS wetlands, and 35% in HSSF wetlands, compared to systems with no delay.

The delay may be accounted by adding a third parameter to the *N*TIS model. Alternatively, it is probably acceptable to use the *N*TIS model.

In the data processing in this book, we have used the simpler two-parameter approach.

### Reactions along Flow Paths for FWS Wetlands

In addition to distributions of detention times (across flow fractions), other factors exist that can influence overall Damköhler numbers. One class of possibilities arises from spatial distributions of wetland water depths and vegetation density. Distributions of Damköhler numbers across flow fractions may be created due to both $k$ and $t$ being related to vegetation density and depth.

Kadlec (2000) postulated that patterns of vegetation can strongly influence the perceived reaction rates in treatment wetlands, as well as affect the velocity and amount of water that passes through a given region of a FWS wetland. This presumption was explored in more detail by Carleton (2002).

As an example, consider a wetland with parallel flow paths of unequal vegetation density. High-velocity flow paths result from lack of vegetative resistance; low-velocity paths from large vegetative resistance (Hosokawa and Horie, 1992; Jenkins and Greenway, 2005). In addition, wetland data show that water velocity is strongly depth dependent (Kadlec and Knight, 1996). For laminar flow under a fixed water surface gradient,

$$u = \frac{a' h^b S^c}{X} = \frac{a h^b}{X} \tag{B.61}$$

$$t = \frac{L}{u} = \frac{LX}{a h^b} \tag{B.62}$$

where

$a, b, c$ = constants, $d^{-1}$
$h$ = water depth, m
$L$ = length of wetland cell, m
$X$ = submerged vegetation surface area density, $m^2/m^3$
$S$ = slope of the water surface, m/m
$t$ = time, d
$u$ = superficial water velocity, m/d

Different values of the exponent $b$ apply to different vegetation types. Here, for purposes of illustration, $b = 1$ is chosen. These relations state that, for a specified overall water slope, the water velocity is inversely proportional to the immersed vegetation surface area and directly proportional to the depth. Therefore, the detention time along any flow path is directly proportional to the vegetation density and inversely proportional to the depth. Consequently, cross-flow spatial distributions of $h$ and $X$ create a distribution of detention times, as detailed by Carleton (2002).

Depth and vegetation density effects on water velocity have been investigated, and an improved understanding is emerging (e.g., Hall and Freeman, 1994). Those same variables exert influence on pollutant-reduction rates, but the quantification of those

effects is still in early stages. There have been some definitive studies that establish the influence of water depth on volumetric rate coefficients; for instance, the data of Gearheart *et al.* (1989) that clearly demonstrated $k$ for BOD was inversely proportional to depth. In other cases, the relationship is less clear. Stem density has been found to be an important determinant of rate coefficients in some cases (Lakhsman, 1982) but not in others.

The wetland vegetation density influences many wetland pollutant removal processes. This may be due to the biofilms that coat submerged litter, roots, and stems, or to the biogeochemical cycling that creates net uptake to new sediments (Khatiwada and Polprasert, 1999a). Many removal processes are associated with surface area, such as mass transfer or microbial reactions. The active reaction zone may therefore be on the bottom, in the water column, or a combination. If the active zone is exclusively associated with immersed surfaces, the rate coefficient is presumptively proportional to $X$, as observed by Smith *et al.* (2000) for nitrate reduction. Conversely, many wetlands display constancy of areal rate, and thus the volumetric rate coefficients used in previous sections are inversely proportional to depth:

$$k = \frac{\beta X}{h} \tag{B.63}$$

There may be other influences on the rate coefficient, such as enhancement of mass transfer via increasing velocity. Here, such velocity dependence is assumed to be absent, as a limiting case.

The effect of parallel spatial patterns of vegetation density and water depth is to create spatial patterns of the removal rate constant. The effects of both types of distribution on pollutant removal are thus two-fold. Flows through low vegetation density regions are fast and ineffective, whereas flows through high vegetation density regions are slow and effective. Flows through high-depth regions are fast and ineffective, while flows through low-depth regions are slow and effective. Combination of Equations B.62 and B.63 for $b = 1$ gives

$$kt = \left(\frac{a\beta}{L}\right)t^2 = \gamma t^2 \tag{B.64}$$

where

$\gamma = (a\beta/L) =$ constant, $d^2$

This is the same result presented by Carleton (2002). The fraction remaining (for $C^* = 0$) along a parallel flow path of vegetation resistance $X$ and detention time $t$ is

$$\frac{C}{C_i} = e^{(-\gamma t^2)} \tag{B.65}$$

Under this potential scenario, the fraction of pollutant remaining along a given flow path decreases exponentially with the square of the detention time along that path. It is the result of a direct interrelation of $k$, $X$, $h$, and, $t$ and the corresponding wetland circumstance is termed the "*Xhkt*" condition. Note that there is a serious consequence of the *Xhkt* condition, in that travel time in a flow system is no longer equivalent to batch time in a non-flow-through system. Any batch experiment is necessarily done under conditions of constant $X$ and variable time, which is not possible in a flow system if travel time is dependent on $X$. This nonequivalence has been experimentally explored by Stein *et al.* (2003).

Vegetation density and depth distributions can produce a false impression of plug flow conditions. Poor treatment along fast flow paths produces a faster-than-average rate of pollutant removal at longer detention times, which precludes plateaus in concentration profiles. Kadlec (2000) points out that the interdependence of $k$ and detention time leads to an imputed PFR rate constant (from a PFR fit of the *Xhkt* data) that depends upon the hydraulic loading rate. Here, it is additionally noted that the *Xhkt* condition may yield the appearance of PFR performance for an *N*TIS wetland. To illustrate, the behavior of a 3-TIS wetland is examined under the *Xhkt* assumption for a zero-background concentration. The coefficient $\gamma = 0.25\ d^{-2}$ yields flow-weighted outflow concentrations that decrease with increasing mean detention time (Figure B.29). This hypothetical data may be fitted with a relaxed TIS model, where the variable $P$ represents the

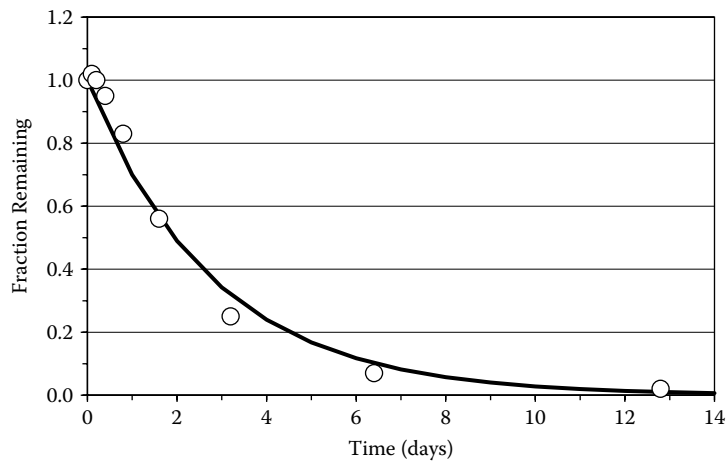



**FIGURE B.29** Fraction remaining for a single compound undergoing first-order degradation or removal in a wetland with variable vegetation density on parallel flow paths. The points are computed from Equation B.38, coupled to a 3-TIS DTD. The line represents the *N*TIS model, as well as the PFR model.

combined effect of the number of tanks, $N$, and weathering of the contaminant, as discussed in Chapter 6. The results for this model are $P = 1{,}834$, $k_{app} = 0.358\ d^{-1}$, and $R^2 = 0.982$. This high $P$-value corresponds to an apparent plug flow condition.

Thus, both increases and decreases in the apparent TIS number can result from internally distributed parameters. Interestingly, no combination of distribution effects has been found here that caused performance behavior to stray far from an envelope bracketed by apparent plug flow and apparent TIS.

## SUMMARY

Tracer testing offers a means of better understanding of the processes controlling the performance of treatment wetlands. Results are now available from hundreds of tracer tests of FWS and HSSF systems. Despite the appealing simplicity of the plug flow assumption that has repeatedly been made, and continues to be made, in the literature, it is abundantly clear that neither FWS nor HSSF systems behave as plug flow. Furthermore, many wetlands have significant fractions of the water column that are dead zones, which are not directly flushed by flowing water, as evidenced by volumetric efficiencies less than unity. The interpretation of wetland pollutant removal data, and the design for pollutant removal, are facilitated by a knowledge of the volumetric efficiency, as well as the DTD efficiency of the wetland.

The proper technique of conducting a tracer test has not always been adhered to, thus causing difficulties in interpretation. Density stratification, degradable and sorbable tracers, and lack of hydrologic control have obscured results in some cases. Nonetheless, a great deal of progress has been made in the last ten years toward better understanding of the features of treatment wetlands and how they affect hydraulic behavior. It has been possible to apply quite sophisticated models to represent tracer responses, but that level of detail is not always necessary for interpretation of contaminant removals. Importantly, this new understanding moves the science of wetland performance toward greater confidence levels. Although the old plug flow approach may have been sufficient for interpolating key process data in for design purposes, extrapolation beyond the existing data set was quite often disastrous. Tracer results enable a new generation of models that are better able to describe systems over wider ranges of operation conditions.

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**COLOR FIGURE 1.13** Single-home HSSF wetland in Comfort Lake, Minnesota.



**COLOR FIGURE 3.33** Trees growing in the Vermontville, Michigan, constructed treatment wetland after 15 years of operation.



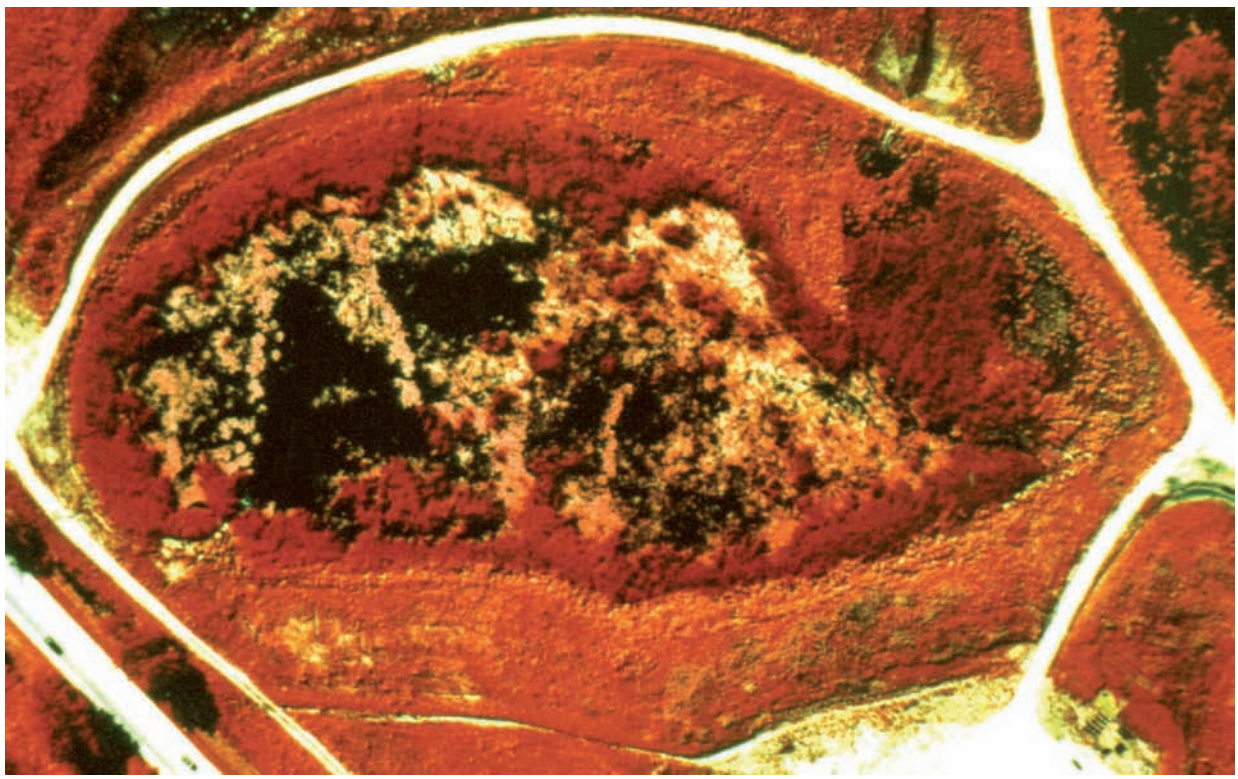
**COLOR FIGURE 7.10** Venting groundwater at this Wellsville, New York, site contains iron, which oxidizes upon contact with air.



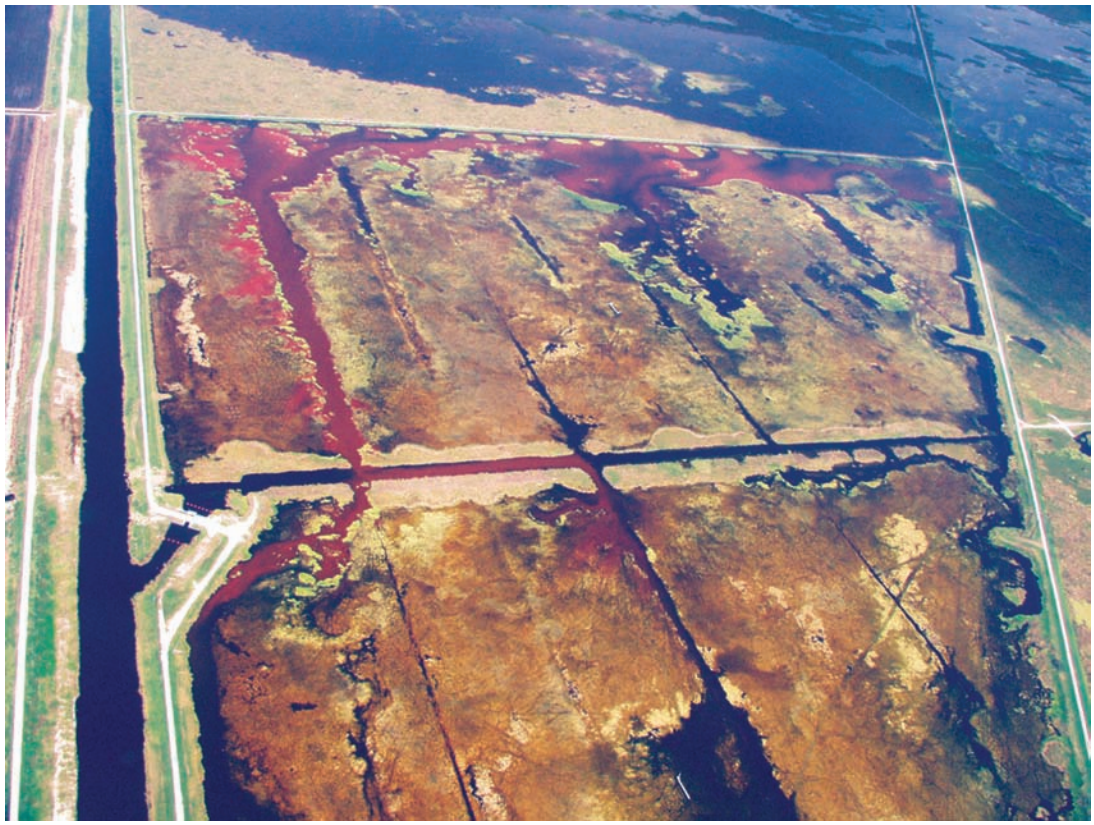
**COLOR FIGURE 11.6** This HSSF wetland outlet structure at Tamarack, Minnesota, has become coated with elemental sulfur.



**COLOR FIGURE 14.7** An urban stormwater wetland in the city of Blue Mountains, Australia.



**COLOR FIGURE 22.7** Wetland EW3 at the Des Plaines River site near Wadsworth, Illinois. The fringe zone is vegetated by cattails and bulrushes, whereas the interior has submerged aquatic vegetation and floating-leaved plants. Open water shows as dark areas on this false color infrared photo.



**COLOR FIGURE B.8** Progress of Rhodamine WT dye tracer through a FWS wetland, cell 4 of the ENRP in Florida. The dye was introduced along the upper boundary, and flow proceeds from the top to the bottom. Note the short-circuit along the left-hand side, which is partially redistributed by the central cross canal. (Photo courtesy of T. DeBusk.)



**COLOR FIGURE B.22** Rhodamine dye dispersing through an inlet deep zone, with a spikerush band providing resistance to shortcircuiting. (From CH2M Hill (2003a) *PSTA Research and Demonstration Project, Phases 1, 2, and 3 Summary Report*, Prepared for the South Florida Water Management District.)



# TREATMENT WETLANDS

## SECOND EDITION

*Treatment Wetlands, Second Edition* represents the most comprehensive and up-to-date resource for the design, construction, and operation of wetland treatment systems.

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