Management Technologies for Metal Mining Influenced Water

Mine Pit Lakes
Characteristics, Predictive Modeling, and Sustainability

Volume 3

Edited by Devin N. Castendyk and L. Edmond Eary

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On the Cover: Mesel Mine Pit Lake, Indonesia by Ted Eary.

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Preface

This *Mine Pit Lakes: Characteristics, Predictive Modeling, and Sustainability* handbook is a project of the Acid Drainage Technology Initiative–Metal Mining Sector (ADTI-MMS), based in Colorado, United States. The mission of the ADTI-MMS is to identify, evaluate, develop, and disseminate information about cost-effective, environmentally sound methods and technologies to manage mine wastes and related metallurgical materials for abandoned, inactive, active, and future mining and associated operations, and to promote understanding of these technologies. ADTI-MMS is a technically focused consensus group of volunteer representatives from state and federal government, academia, the mining industry, consulting firms, and other interested parties who are involved in the environmentally sound management of metal-mine wastes and drainage quality issues. This volume is third in a series of six handbooks written by ADTI-MMS. The other five handbooks in the Management Technologies for Metal Mining Influenced Water series are *Basics of Metal Mining Influenced Water, Mitigation of Metal Mining Influenced Water, Geochemical Modeling for Mine Site Characterization and Remediation, Techniques for Predicting Metal Mining Influenced Water*, and *Sampling and Monitoring for the Mine Life Cycle*.

What began as a document focusing on pit lakes in the state of Nevada evolved into an international collaboration involving dozens of researchers from Canada, Germany, Australia, New Zealand, and the United States. ADTI-MMS is grateful to the people and organizations who donated valuable time and resources toward this project. In addition to the contributing authors, we thank Lisa Stillings at the U.S. Geological Survey and Jim Jonas at NewFields who initially chaired the Pit Lake Committee and provided valuable direction and insight at the inception of the project. In 2003, David Bird at the Colorado Geological Survey helped to refine the objectives of the committee, which led to a $10,000 grant from the International Network for Acid Prevention in January 2004. Later that year, Boojum Research Ltd., Toronto, Canada, pledged to contribute $4,000 toward the production and dissemination of the handbook. Brian Park at Mansfield Technologies served as co-chairman from 2006 to 2007 and greatly assisted in defining the scope and content of the manuscript. Special thanks go to Chris Gammons at Montana Tech, S. Ursula Salmon at the University of Western Australia, and Virginia McLemore at New Mexico Tech for their thorough reviews of select submissions.

PART I: INTRODUCTION

CHAPTER 1

The Nature and Global Distribution of Pit Lakes

D.N. Castendyk and L.E. Eary

One of the most significant environmental issues facing the global mining industry is the water quality of pit lakes. Pit lakes form after the closure of open pit mines that extend below the premining water table and reflect permanent modifications to hydrologic systems resulting from mining. Because of decreases in ore grades and the need to optimize efficiency, the number of active open pit mines has increased in recent years compared to underground mining. Consequently, the global abundance of pit lakes will increase in coming decades. With sufficient planning in advance of mining, pit lakes with high water quality have the potential to be utilized as postmining surface water resources such as public recreation areas and wildlife habitats. At the opposite end of the spectrum, pit lakes with poor water quality pose potential risks to ecosystems and humans. It is unrealistic to expect that all pit lakes can be rehabilitated into postmining water resources given sufficient resources, particularly porphyry copper deposits in arid climates, yet strategies can be implemented that significantly reduce their impact. For these reasons, the goals of sustainable development for open pit mining operations require evaluation of the type of water quality that is likely to occur in pit lakes after mine closures and the long-term mitigation efforts, if any, needed to minimize the environmental effects of pit lakes and to develop lakes into water resources if possible. To this end, the role of mine operators and environmental regulators is to make sure that water resources are protected for future generations while enabling mining companies to meet society's present demands for raw materials and economic growth. The success of these challenges depends on the ability to accurately predict the water quality for pit lakes in advance of open pit mining. This volume of the Management Technologies for Metal Mining Influenced Water series by the Acid Drainage Technology Initiative–Metal Mining Sector (ADTI-MMS), addresses the theory and science behind the prediction of pit lake water quality and reviews best-practice pit lake management. The information in this handbook is applicable to permitting studies for new open pit mines, developing closure strategies for existing open pit mines, and designing monitoring programs for existing pit lakes.

OPEN PIT MINING

Pit lakes are a product of open pit mining for precious metals, base metals, coal, uranium, iron, diamonds, industrial minerals, aggregates, and other commodities. This handbook focuses primarily on pit lakes formed in open pit metal mines, acknowledging that similar processes may occur in other pit lake environments. The study of pit lakes is also of benefit to those concerned with flooded underground mine collapses, which have the appearance of, and many similarities to, an open pit surface mine.

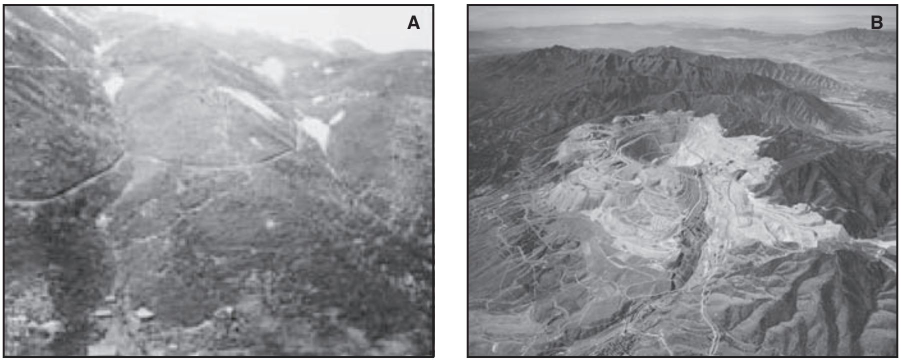
Open pit mining is used for ore deposits that can be mined profitably by surface excavation methods to reach the mineralized zones. Excavation is done by removing successive layers of rock to create a bowl-shaped pit that has benches along the pit walls that serve as transport routes for equipment and to catch loose rock as the pit becomes progressively deeper. The layers of removed rock may contain some proportion of both nonmineralized and mineralized rock that need to be segregated based on metal content and potentially placed in engineered disposal facilities, depending on potential environmental impacts. Waste rock is any mined rock that cannot be economically processed and is sent to waste rock storage areas, whereas ore is the mineralized rock that can be economically processed for extraction of some portion of the metal content. Rock units overlying the ore, called the overburden, can also contain significant amounts of pyrite and result in poor pit water quality even though all of the ore is removed. The ultimate depth and shape of an open pit are dependent on the type of ore deposit, the shape of the ore deposit, mechanical integrity of the rocks in the pit walls, and relative costs of ore recovery versus costs of waste rock removal. Environmental costs for disposal of waste rock and process tailings, monitoring requirements, and closure also factor in to the length of mine life and ultimate pit size (McLemore 2008). Some amount of unmined ore may be left in the sides or bottom of the pit at closure because the relative cost of removing waste rock exceeds the value of recovering additional ore at depth (McLemore 2008), although underground mining may proceed to deeper ore bodies if economically viable.

Open pits can become very large landscape features on account of the mammoth scale of material movement. For example, the Bingham Canyon copper pit in Utah (United States) was originally a hill when open pit mining began in 1904 and is now more than 4 km wide and 1.2 km deep with about 450,000 t (metric tons) of rock and ore removed from the pit daily (Figure 1.1). The Bingham pit is reportedly the largest human-made excavation in the world (www.kennecott.com). Other exceptionally large open pit mines include the Chuquicamata copper mine in Chile, which is approximated 4.5 km in width and 0.8 km deep, and the Escondida open pit copper mine in Chile, which produces more than 750,000 t of material per day.

PIT LAKE FORMATION

The size and depth of open pit mines cause disruptions to the natural hydrologic system surrounding the mine site. Disruptions occur because of changes in topography and infiltration characteristics of the surface soils and rocks. In addition, many open pit mines eventually reach depths below the natural water table. To prevent groundwater flow from inundating the pit and destabilizing the pit walls, most open pit mining operations use a system of pumping wells to dewater the groundwater system around the pit. They may also use diversions to minimize the amount of surface water draining into the pit and employ sump pumps to remove water collected at the bottom of the pit from precipitation, runoff, and inflow from perched aquifers (Figure 1.2a). Water from the pumping wells may be used in ore processing or dust control, or disposed to surface drainages after treatment if necessary, depending on the water balance for the mine and also the water quality.

After mining has ended, the operation of dewatering wells is also ended. As a result, groundwater and surface runoff flow into the open pit as the water table rebounds to a new state of hydrologic equilibrium, producing a pit lake (Figure 1.2b). Depending on rates of groundwater flow and rainfall, the time for a pit lake to reach a steady-state depth may range from a few years to many decades. At mines in arid regions of the western United States and other mining districts where mine dewatering operations may have taken place for numerous years and the storage of the rocks are low, rebound of the hydrologic system of a pit lake may take



Courtesy of Kennecott Utah Copper.

FIGURE 1.1 Bingham Canyon copper mine in Utah, United States, in 1863 (a) and in 2003 (b), showing the excavation of a large open pit over a 140-year period

from 50 to 300 years. In contrast, pit lakes in humid regions where extra surface water is available may take less than a decade to achieve hydrologic equilibrium if surface water is used to flood the pit. For example, the Island Copper pit lake in British Columbia, Canada, was flooded with seawater to a depth of 380 m in approximately 8 days (Wilton and Lawrence 1998).

For most pit lakes in arid regions, groundwater is the primary source of water, and its rate of inflow is dependent on the hydraulic conductivity of the surrounding aquifers and the hydraulic gradient between the water level in the pit and the water elevation in the groundwater system within the capture zone of the pit (Atkinson 2002). In theory, the rate of inflow is initially rapid because the hydraulic gradient is at a maximum and decreases over time as the gradient becomes smaller. At the same time, the rising water table increases the groundwater discharge area as the lake fills, and, in some cases, this increase can offset the decrease in hydraulic gradient resulting in a constant filling rate for a period of time. One of the longest-studied pit lakes in the world, the Berkeley pit lake in Butte, Montana (United States), filled at a constant rate for the first 15 years of flooding owing to this balance (Duaime et al. 1998).

Ultimately, the final depth of the pit lake will be a function of the balance between water inflows and outflows. In arid regions with deep groundwater systems, low hydraulic conductivity, and low rates of infiltration, evaporation is often the major outflow process for water. These pit lakes are often referred to as terminal lakes or hydrologic sinks. Pit lakes located in areas of higher rainfall, hydraulic conductivity, and infiltration may have groundwater entering on an upgradient side and leaving on the downgradient side or overflowing to an exit channel. These types of pit lakes are often referred to as flow-through pit lakes.

FACTORS INFLUENCING PIT LAKE WATER QUALITY

Our conceptual understanding of pit lakes can be likened to a pyramid constructed of layers of interrelated hydrologic, biogeochemical, geologic, and limnologic processes (Figure 1.3). The foundation of the pyramid consists of the climate, geology, and hydrology of the mine area. In the second layer, the geology of the mineral deposit defines the final shape and depth of the open pit, whereas the hydrology and climate define the pit lake water balance. The morphology of the open pit, the water balance, and the local climate drive the physical limnology of the pit lake, represented by the third layer of the pyramid. The fourth layer, lake geochemistry, includes

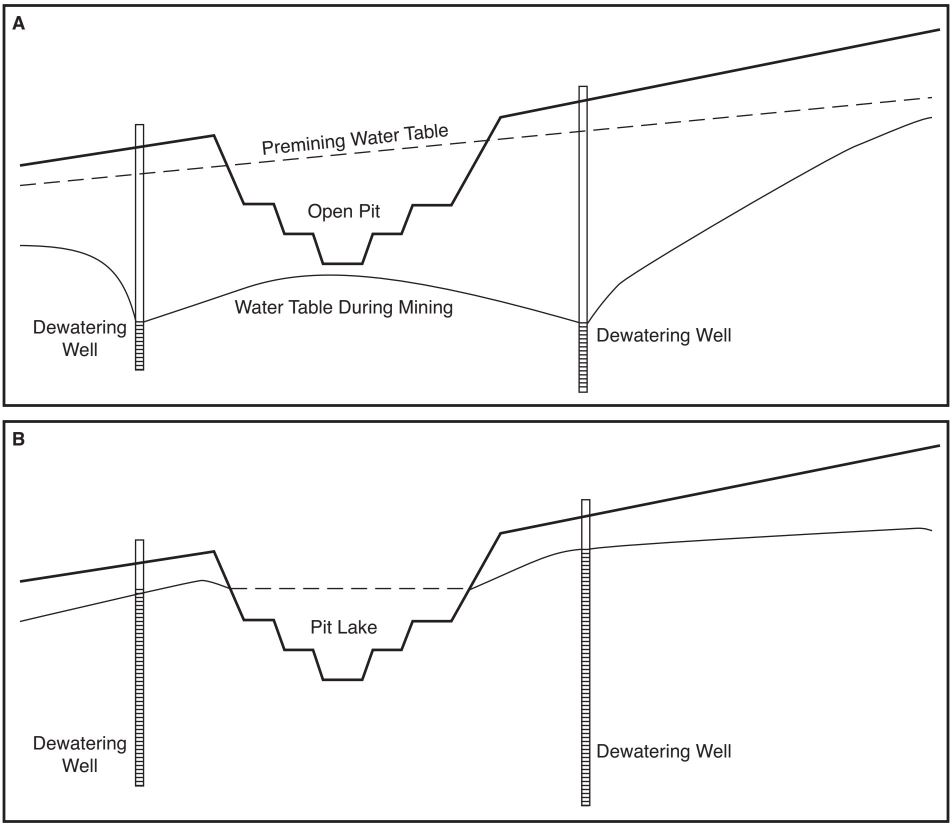


FIGURE 1.2 Hydrologic system for an open pit (a) during mining when dewatering wells are operating and (b) after closure when dewatering wells are shut down

inorganic water–rock reactions and biologically mediated chemical processes occurring in the walls of the pit both above and below the water level, within the water column, and within the pit sediments. These processes are important for affecting the acid–base balance of the water and rates of release of metals from the oxidation of metal sulfide minerals that are commonly present in metal deposits. The physical limnology of the lake controls mixing between shallow and deep lake water and the cycling of nutrients, dissolved metals, anions, suspended particulates, oxygen, and carbon dioxide within the water column. Both ecosystems that take hold in the water column and sediments may also affect the acid–base balance and metal concentrations in lake water. Overlying these layers are management decisions that determine whether the final pit shape changes during mining, the strategy for mine closure, the use of the pit lake after mine closure, and related mitigation strategies that may be employed to improve lake water quality over the long term after site closure. The chemical composition of the water in the pit lake is the product of the combination of all of these processes and is represented by the top of the pyramid.

This pyramid structure illustrates the significance of processes occurring within a pit lake and their dependence on one another (Figure 1.3). Because of the interdependencies of the processes, there are numerous feedback loops that are difficult to predict without numerical models. Moreover, from a prediction standpoint, this structure shows how changes to the foundational

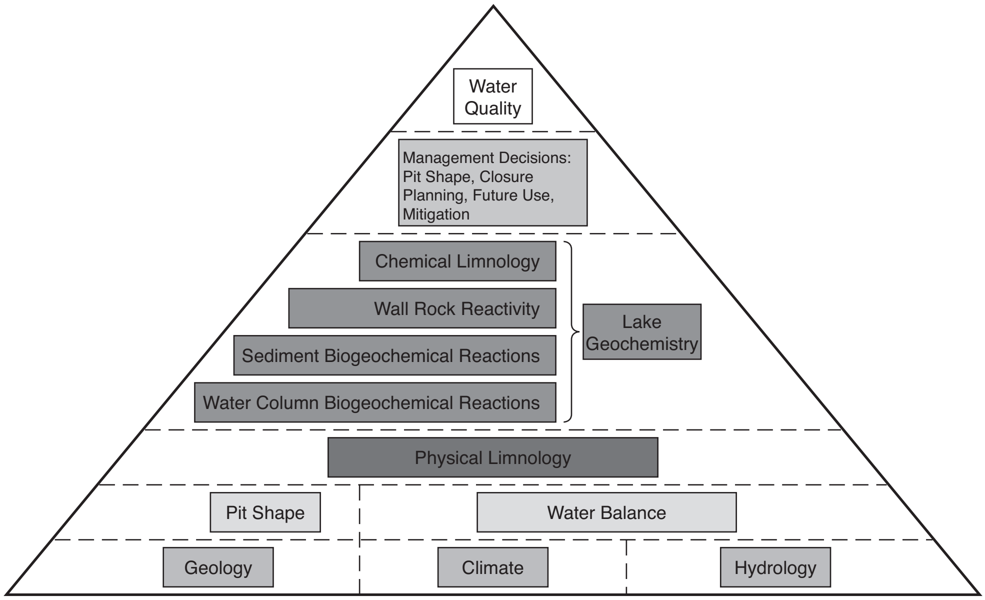


FIGURE 1.3 Conceptual relationship between the principal factors that affect pit lake water quality considered in the scientific literature

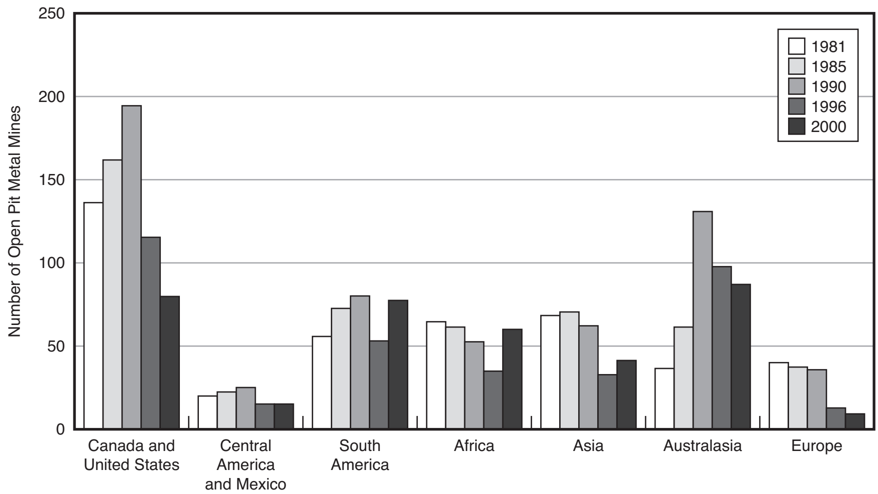
assumptions, that is, conceptualizations of pit lake processes, can strongly affect the results produced by a pit lake water quality model. This handbook includes chapters dedicated to each of the factors shown in Figure 1.3.

GLOBAL DISTRIBUTION OF PIT LAKES

The global distribution of pit lakes is expected to be similar to the distribution of open pit mines because a high proportion of open pit mines will become pit lakes after closure. Open pit mines are located in all of the major mining districts of the world with the highest numbers in North America, Australia, and South America, according to mining activity surveys for 1981 to 2000 (Figure 1.4). In 2000, Australia, the United States, Chile, South Africa, Brazil, and Canada had the highest numbers of open pit mines (Table 1.1). More recent data on mining activity were not available. However, in consideration of past exploitation of many prospects in the established mining districts of Canada, the United States, and South Africa, it is anticipated that there will be increasing numbers of open pit mines and, therefore, pit lakes in West, East, and Central Africa, Central Asia, and Central America in the future (Castro and Moore 2000).

ENVIRONMENTAL EFFECTS OF PIT LAKES

Pit lakes have the potential for causing long-term changes to hydrologic systems and water quality (NRC 1999). Changes to the hydrologic systems occur because of the creation of a lake where, prior to mining, no lake existed. The collection of water in pit lakes rather than being released to surface rivers and natural lakes can result in net losses of water, especially in arid regions. For example, Miller (2002) has estimated that 35 pit lakes in Nevada (United States) have the potential to store 1.9 billion m^3 (1.5 million acre-feet) of water compared to 740 million m^3 (0.6 million



Source: Data from Mining Activity Survey 1981, 1985, 1990, 1996, 2000.

FIGURE 1.4 Global distribution of open pit mines (Ag, Au, Cr, Co, Cu, Fe, Mo, Ni, Pb, Pd, Pt, U, Zn) from 1981 to 2000 for mines with ore production in excess of 150,000 t/y, which accounts for about 90% of Western World ore output

TABLE 1.1 Numbers of open pit mines (Ag, Au, Cr, Co, Cu, Fe, Mo, Ni, Pb, Pd, Pt, U, Zn) by country in 2000

Country	Number of Open Pit Mines in Year 2000	Country	Number of Open Pit Mines in Year 2000
Australia	78	Iran	2
United States	63	Philippines	2
Chile	33	Uzbekistan	2
South Africa	21	New Zealand	2
Brazil	20	Columbia	0
Canada	17	Guyana	0
Zimbabwe	13	Suriname	1
Ghana	10	Uruguay	1
Indonesia	10	Costa Rica	1
Peru	8	Honduras	1
Venezuela	7	Botswana	1
Mexico	7	Namibia	1
India	7	Sudan	1
China	5	Armenia	1
Kazakhstan	4	Georgia	1
Papua New Guinea	4	Kyrgyzstan	1
Bolivia	3	Malaysia	1
Democratic Republic of Congo	3	Mongolia	1
Zambia	3	Myanmar	1
Spain	3	Russia	1
Sweden	3	Saudi Arabia	1
Argentina	2	Tajikistan	1
Cuba	2	Fiji	1
Dominican Republic	2	New Caledonia	1
Nicaragua	2	Austria	1
Guinea	2	Finland	1
Ivory Coast	2	Greece	1
Mali	2		

Source: Data from Mining Activity Survey, 2000.

acre-feet) of water stored in all of the reservoirs in the state. The storage of water in pit lakes is a potentially important consideration in the southwestern United States, a region that hosts many precious metal open pit mines and the first and second fastest growing state populations in the country (i.e., Utah and Arizona; U.S. Census Bureau, 2008). Moreover, Hoerling and Eischeid (2007) predict that this region is entering a new, long-term drought era on account of increased surface air temperatures and evapotranspiration rates.

In arid and mountainous areas of the world where many mining districts are located, pit lakes have the potential to be important water resources if they contain high-quality water. For regions such as Nevada, where data are available to examine water quality trends, most pit lakes are neutral to alkaline in character and have relatively low concentrations of metals and reasonably good water quality (Price et al. 1995). Given their potential volumes and high water quality, mine pit lakes in Nevada comprise a potentially large water resource. Some of these lakes have the potential to be used as recreation areas, wildlife reserves, aquaculture facilities, and irrigation water reservoirs. The Yerington pit lake in western Nevada has also been used to reduce flood damage along the Walker River. However, the resource value of pit lakes is greatly diminished if the water is of poor quality. Moreover, low-quality groundwater or surface water discharging from pit lakes may contaminate present or future drinking water supplies. This has occurred near Jamestown, California (United States), where arsenic-rich groundwater discharging from the Harvard pit lake increased arsenic concentrations in an adjacent aquifer to levels above national drinking water standards (Savage et al. 2000). The region surrounding the Harvard lake has experienced a recent population increase, yet new water supplies are limited because of contamination from historic mine sites.

Poor water quality could affect both the ecological communities that might come into contact with the surface water of the pit lake and the downgradient groundwater system at flow-through pit lakes. Perhaps the best known example of a pit lake that poses environmental risks is the Berkeley lake in Butte, Montana (United States), which has a pH near 2.5 and potentially toxic concentrations of Ag, As, Cd, Cr, Cu, Ni, Pb, V, and Zn (Davis and Ashenberg 1989; Pellicori et al. 2005). The Berkeley pit lake is part of the largest Superfund site in the United States and is occasionally responsible for migratory bird fatalities (Hagler Bailly Consulting 1996). Other acidic pit lakes resulting from metal mining in the United States occur at the Elizabeth copper mine Superfund site (pH 3.5–3.9) in Vermont (Seal et al. 2003), the Red Hill pit (pH 2.6–2.7) in South Carolina (Davis and Eary 1997), and the Liberty (pH 3.2) and Ruth (pH 3.9) mines in Nevada (Davis and Eary 1997). Outside the United States, several acidic pit lakes have resulted from metal mining such as the East Sullivan mines (pH 3.2) in Quebec, Canada (Tassé 2003); Parys Mountain (pH 2.3) in northwest Wales (Bowell 2002); Udden pit lake (pH 4.8 to 5.5) in northern Sweden (Ramstedt et al. 2003); and 21 pit lakes ranging from pH 2.2–3.6 plus Corta Atalaya (pH 1.2) in the Iberian Pyrite Belt of southwest Spain (España et al. 2008). Low-pH pit lakes can also develop from coal mining. Six pit lakes in the Anthracite District of eastern Pennsylvania (United States) range from pH 3.7 to 4.3 (Mase et al. 2008), whereas several pit lakes in abandoned lignite mines in eastern Germany range from pH 2.6 to 3.8 (Klapper and Schultze 1995; Stottmeister et al. 1999; Boehrer et al. 2003; Karakas et al. 2003).

For terminal pit lakes in arid climates, evapoconcentration has the potential to reduce water quality over time as the removal of fresh water by evaporation causes concentrations of dissolved metals to increase (Eary 1998). At least two pit lakes in Nevada, Getchell-North pit (pH 7.0–8.0) and Boss (pH 8.0–8.5), have arsenic concentrations that exceed state drinking water standards in part due to this process (Price et al. 1995; Miller et al 1996; Shevenell et al. 1999; Shevenell 2000; Shevenell and Connors 2000; Tempel et al. 2000).

SUSTAINABLE DEVELOPMENT

The burgeoning global population will not only increase the demand for mineral resources, it will also increase the need for freshwater resources. To meet the needs of future populations, a growing number of mining corporations, governments, and regulatory agencies require the sustainable development of natural resources. For example, the New Zealand Resource Management Act (RMA 1991) requires the sustainable management of resources in a way that provides for social, economic, and cultural well-being in addition to sustaining the potential of the land and water to meet the needs of future generations; safeguarding the life-supporting capacity of water, soil, and ecosystems; and avoiding, remedying, or mitigating any adverse effects on the environment.

Clearly the sustainable management of nonrenewable resources, such as minerals, creates problems on account of the longevity of the resource. For example, given the geologic rate of gold deposition, it is unrealistic to expect gold extraction to occur at rates that are equal to, or slower than, deposition rates. Instead, gold mining operations continue until all economically accessible gold is exhausted at a particular deposit. Because mineral resources are finite, the measure of sustainable development with respect to mineral resources cannot be based on the perpetual availability of the mineral, but rather the long-term quality of the water, soil, and land that naturally occur at the mine site and within the surrounding community. Therefore, to accomplish the sustainability requirements listed above, mining companies need to preserve the integrity of the water, soil, and land resources such that the postclosure mine site and community have an economic value and potential for reuse of social, economic, and/or environmental purposes after the mining company leaves.

In recognition of this need, nine of the world's largest mining companies and more than 30 smaller companies established sustainable development objectives for the mining industry as part of the Mining, Minerals, and Sustainable Development Project (MMSD 2002). With regard to open pit mines, the MMSD final report recommends the creation of site closure plans at the onset of mining, and the rehabilitation of mine sites into postmining resources with environmental, social, and/or economic benefits. For pit lakes, this could mean the creation of a surface water resource at the conclusion of mining, such as a recreation area, an ecological habitat, or an irrigation reservoir. To meet societal expectations, it is important for mining companies to discuss postmining resource options with local communities, financial lenders, governments, and other stakeholders when designing closure strategies for open pit mines (Howard 2006).

PURPOSE AND OVERVIEW

One of the main objectives of ADTI-MMS is to communicate knowledge that has been learned from past and present research into the management of mine wastes, mine processing wastes, mined lands, and air and water affected by metal-mining and metal-processing operations. This book is the third volume of the ADTI-MMS Management Technologies for Metal Mining Influenced Water series. The purpose of this handbook is to present the current global understanding of pit lake characterization, modeling, and mitigation/remediation approaches, as well as the postmining rehabilitation of pit lakes into sustainable land or water resources. This handbook also organizes and summarizes the major tools, methodologies, and knowledge base necessary to forecast future pit lakes and appropriately manage existing pit lakes. It is hoped that some of the major causes of uncertainty in existing pit lake water chemistry predictions will be identified and areas where future research is needed will be highlighted. Our intended audience includes mine managers, environmental consultants, environmental regulators, academics, and students. The volume is organized into eight sections:

- Part I provides an introduction to the subject of pit lakes and a review of regulatory issues.
- Part II reviews the climatic, hydrologic, and limnologic characteristics of pit lakes, and describes how lakes may be classified based on these characteristics.
- Part III provides conceptual models used to illustrate hydrologic, limnologic, and geochemical processes that influence pit lake water quality.
- Part IV recommends data to be collected when sampling and monitoring existing pit lakes.
- Part V addresses the prediction of future conditions that will influence the water quality of a future pit lake, including local climate, surface water inputs, groundwater inputs, physical limnology, lake geochemistry and water quality, biogeochemistry and ecology, subaqueous water–rock reactions, and biological reactions. This section also addresses the integration of different models used in the development of a water quality prediction, the validation of prediction models, and the assessment of environmental risk.
- Part VI reviews current technologies to remediate existing pit lakes with poor water quality, such as neutralization, flooding pit lakes with surface water, eutrophication, microbial and organic treatment, backfilling, and induced permanent stratification.
- Part VII explores potential postmining uses for pit lakes toward sustainable resource development, and a technical issue that may limit use, specifically slope stability.
- Part VIII summarizes the state of contemporary pit lake research based on the material addressed in this document. Present data gaps, areas of uncertainty, and future research topics are identified, and a best-practice guide for open pit closure and pit lake management is presented.

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PART I: INTRODUCTION

CHAPTER 2

Regulatory Issues in the United States

R.D. Williams

INTRODUCTION

Publicity regarding pit lakes at abandoned, disused, or unreclaimed hard-rock and coal mine sites has generally been negative. Extensive coverage of sites like "Lake Berkeley" at the abandoned open pit copper mine in Butte, Montana (United States), where pH values are below 3 and heavy metals contaminate the water, has dramatically impacted any discussion of mine development and expansion if there is potential for the formation of pit lakes (Figure 2.1).

This extensive negative publicity regarding pit lakes contributes greatly to the public's perception that pit lakes at any stage in a mine's life are bad and should be avoided. Concerns about pit lakes center on pit water quality, impacts to groundwater, wildlife, and sometimes safety and public utilities. These concerns must be highlighted and addressed in the mine design, permitting, and environmental review process, and updated during operations, closure, and postclosure. This section will address some of the legal framework that governs potential pit lakes in the United States.

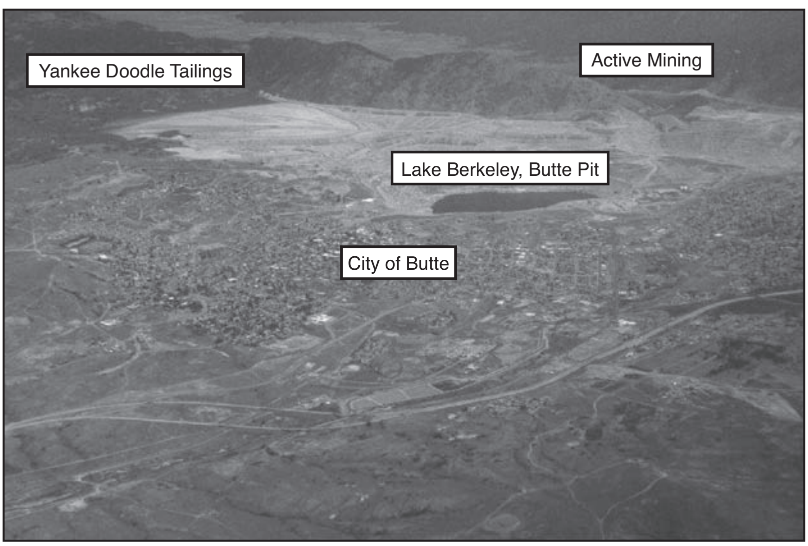

FIGURE 2.1 Aerial view of Lake Berkeley, the Butte pit, and the Clark Fork Superfund Complex, Butte, Montana

LEGISLATION IN THE UNITED STATES

Federal Legislation

The legal requirements for mines proposing to establish pit lakes are linked to several federal laws and may also be constrained by state laws as well.

General Mining Act of 1872. This law comes under United States Code (USC) Title 30, Chapter 22, generally codified as 30 USC 22 et seq. The 1872 mining law is the federal law that authorizes and governs prospecting and mining for most metals and industrial minerals on federal public lands in the United States. The original intent of the law was to grant free access to individuals and corporations to prospect for minerals on public domain lands, and upon making a discovery, stake a claim on the deposit. The law has been extensively amended over time and Congress removed many lands and minerals from operation of the mining law. The most significant changes were the removal of the U.S. National Park System and U.S. National Historic Sites, Native American and military reservations, wilderness areas, and water and power projects from the mining law. Oil, natural gas, oil shale, phosphate, and sodium resources were removed from the mining law and are covered by the Mineral Leasing Act of 1920. Numerous attempts have been made over time to effect significant changes to the statute itself, but the ruling remains unchanged since President Ulysses S. Grant signed it into law in 1872 (Dragonetti 2000).

National Environmental Policy Act of 1969. This comes under USC Title 42, Chapter 55—National Environmental Policy, generally codified as 42 USC 4321–4347. This law established a U.S. national policy promoting the enhancement of the environment. The most significant effect of the law was to establish the requirement for a public environmental review process, which includes environmental impact statements or environmental assessments for U.S. federal government actions.

Clean Water Act of 1972 (CWA). This law comes under USC Title 33, Chapter 26—Federal Water Pollution Prevention and Control Act, generally codified as 33 USC 1251–1387. The CWA is the primary federal law that regulates water pollution in the United States. This law was significantly expanded by the Clean Water Act of 1977 and the Clean Water Act of 1987. The statute employs a variety of regulatory and nonregulatory tools to sharply reduce direct pollutant discharges into waterways, finance municipal wastewater treatment facilities, and manage polluted runoff. These tools are employed to achieve the broader goal of restoring and maintaining the chemical, physical, and biological integrity of the nation's waters so that they can support "the protection and propagation of fish, shellfish, and wildlife and recreation in and on the water." The CWA does not deal directly with groundwater or with water quantity issues. These are often dealt with in state law.

Federal Land Policy and Management Act of 1976. Coming under USC Title 43, Chapter 35, this law is generally codified as 43 USC 1701–1785. This law recognized the value of public lands and established a legal framework for the U.S. Bureau of Land Management (BLM) to manage public lands for multiple uses. This law also required, for the first time, the recording of mining claims with the BLM and established that failure to record and maintain the claims with the BLM constituted abandonment.

Surface Mining Control and Reclamation Act of 1977 (SMCRA). This law falls under USC Title 30, Chapter 25, generally codified as 30 USC 1234–1328. This is the primary federal law that regulates the environmental effects of coal mining in the United States. SMCRA created two programs: one for regulating active coal mines and a second for reclaiming abandoned mine lands. SMCRA also created the Office of Surface Mining Reclamation and Enforcement (OSM), an agency within the U.S. Department of the Interior, to promulgate regulations, to fund state regulatory and reclamation efforts, and to ensure consistency among state regulatory programs.

Other laws, such as the Clean Air Act, and so forth, may also apply to mining operations, but the above laws are those that most specifically affect these types of operations. The laws for metal mining operations are distinctly different than those for coal. Coal mining reclamation requirements are spelled out in considerable detail in the SMCRA.

State Legislation

State laws in the western United States are generally silent about whether or not pit lakes are acceptable at reclaimed pit mines. The state laws generally rely on a variety of other applicable laws regarding ground- and surface water quality, and/or the utility of the lake for wildlife or humans, to determine whether a pit lake would be acceptable or not. In most western states, the regulation of mining activity, including potential pit lakes, is shared between an agency that regulates mining activity and reclamation and a sister agency that regulates water quality. Virtually all western states require environmental performance bonding, whereby a special fund or insurance policy is established prior to mining that is intended to cover unexpected costs associated with site rehabilitation. It is unlikely that a pit lake with predicted poor water quality could be permitted in any of the western states without an approved plan for remediation at closure (e.g., external or in situ water treatment, water diversion, backfilling) (Bolen 2002).

METAL MINE RECLAMATION REQUIREMENTS

There is no overarching single metal mine reclamation law but rather a group of federal and state laws covering various aspects of reclamation. These laws and regulations are administered by a variety of state and federal agencies, often acting together under various formal or informal agreements to simplify mine permitting. Since mines may adversely impact ground or surface waters, most mine closure plans focus on reclamation as a means to limit potential impacts to surface or groundwater quality or quantity. Pit lakes can become a reclamation issue if surface or groundwater quality is threatened or if the quality of the pit lake itself poses an environmental risk. There are no laws that specifically prohibit including pit lakes as a possible reclamation alternative at closure, provided that relevant ground- and surface water quality regulations can be met.

U.S. Bureau of Land Management

Federal regulations covering surface management for BLM lands are found at 43 Code of Federal Regulations (CFR) 3809. The regulations require the BLM to "prevent unnecessary or undue degradation." The regulations are silent with respect to pit lakes, and the manual guidance developed from the regulations is general and recognizes that pit lakes may or may not be desirable. The BLM's regulations also require that a reclamation plan include "information on the feasibility of pit backfilling that details economic, environmental and safety factors..." (43 CFR 3809.401(b)(3)(iii). The regulations do not require pit backfilling, and BLM manual guidance details some of the considerations for determining when pit backfilling may be appropriate to consider as an alternative. Refer to Chapter 20 of this handbook for additional details on pit backfilling. The BLM requires a financial guarantee for the proposed reclamation.

U.S. Forest Service

U.S. Forest Service (USFS) regulations governing use of the surface of National Forest System lands for mining operations authorized by the 1872 mining law are found under 36 CFR 228, Subpart A. The regulations require that mining operations be conducted so as to minimize adverse environmental effects to surface resources. The regulations do not require pit backfilling and are in fact silent on both pit backfill and pit lakes. The USFS requires a financial guarantee for the proposed reclamation.

STATE REQUIREMENTS

In the western United States, most states do not specifically require open cut or pit backfilling, nor do they preclude reclamation alternatives that include pit lakes. Regulations most typically include references to returning landscapes to some level of productive use, reducing hazards, and establishing stable landforms. These requirements generally neither preclude nor require backfilling or reclamation alternatives that include pit lakes. Most western states require financial guarantees for proposed reclamation and generally coordinate this requirement with either the BLM or USFS so that only a single financial guarantee is required. Where pit lakes are proposed, state and federal regulatory agencies generally require detailed geochemical modeling of projected water quality as well as modeling of water quantity and pit filling rates. This geochemical modeling may extend for decades into the future to ensure long-term water quality. Regulatory agencies also typically require monitoring of both water quality and quantity as pit lakes fill to ensure that modeling projections are accurate. In the event that water quality of a pit lake does not meet relevant water quality standards, it may be necessary to implement a water treatment program to ensure that surface and groundwater are protected. Implementation of water treatment programs will generally require detailed design studies, construction of treatment facilities, and planning for potentially perpetual operation and disposal of associated waste products. These facilities are expensive, both to engineer and design, and also to provide financial guarantees for, which in some cases may be a trust fund that provides a long-term source of funding for water treatment.

Company Policies

Many metal mining companies have developed their own internal "sustainable development" policies that may provide for alternative reclamation (see MMSD 2002). In some cases, these policies can mean the development of industrial parks at mine sites that are undergoing closure. Mine sites can be attractive for commercial development because of the support infrastructure the mine has developed. Generally these proposals include the standard mine reclamation practices, which could include a pit lake but also provide for continued commercial use of the site. In some cases, this could mean the mine buildings at the site would not be removed or demolished but would remain to support continued commercial development. It is possible that the development of pit lakes with good quality water could enhance opportunities for sustainable development.

COAL MINE RECLAMATION REQUIREMENTS

Coal mine reclamation requirements are codified at 30 CFR 700 and generally preclude the development of pit lakes through the requirement to achieve "approximate original contour." However, a number of coal mine pit lakes exist in the Anthracite Mining District of eastern Pennsylvania (see Mase et al. 2008), and elsewhere in the United States, which predate this law. Occasionally the high wall of one of these mines will be deemed a risk to local residents, and federal funds from the U.S. Department of the Interior OSM will be utilized for rehabilitation. This funding mechanism was established under SMCRA. The Acid Drainage Technology Initiative–Coal Mining Sector has produced two guidebooks that address water quality impacts associated with historic, existing, and future coal mines in the United States (Skousen et al. 1998; Kleinmann 2000). Many coal mine pit lakes also exist in Germany, as do coal and metal mine pit lakes in Western Australia, which are discussed elsewhere in this handbook.

OTHER REGULATORY REQUIREMENTS

Wildlife and/or waterfowl mortalities are common problems that can occur any time a mine or metallurgical processing facility has standing poor-quality water, whether in process tailings, process ponds, or pit lakes. These are generally considered to be illegal "takings" and can result in fines and legal action by state wildlife agencies or the U.S. Fish and Wildlife Service. For pit lakes, this can be a difficult problem to resolve, depending on the location. The standard practice for pit lakes is to develop extensive hazing and monitoring programs to keep waterfowl from landing, and shoo them away if they do land. Pit lakes in areas with little available alternative water sources can be problems, and even aggressive hazing programs can fail when bad weather or fog forces waterfowl to land.

OTHER RECLAMATION REQUIREMENTS

Mining operations other than coal or metal mines, such as quarries and aggregate excavations, also have the possibility of developing pit lakes. The reduced likelihood of adverse water quality is likely to make these pit lakes less of an environmental issue than is often the case with metal or coal mines, and in many cases these lakes have considerable recreational value.

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CHAPTER 3

Climatologic Characteristics

R. Grimaldi

INTRODUCTION

Annual volumes of precipitation and evaporation at a mine site dictate the volumes of water entering and leaving a pit lake through groundwater, pit wall runoff, surface water, and direct exchange across the lake surface. The addition or removal of water affects pit lake chemistry by diluting or concentrating dissolved solids, and by transporting dissolved solids to or from the lake. Hence, an understanding of local meteorological conditions is central to the development of pit lake predictions and to the evaluation of regional, climate-based trends in pit lake geochemistry.

Prior to mining at a new, remote location, on-site meteorological data will most likely not be available. Modelers typically resort to using data from the nearest neighboring weather observing stations for predictions. However, these may be far enough removed from the pit lake such that annual mean values for precipitation and evaporation do not adequately represent on-site conditions. This issue presents a significant challenge for pit lake modelers, which can potentially influence the accuracy of pit lake water quality predictions.

This chapter provides a general understanding of the physics of precipitation and evaporation rates that pit lake modelers should consider when selecting values for these parameters. It also provides a discussion on the impact of global climate change on mean precipitation and evaporation values that should be considered when developing long-range (10- to 100-year) predictions.

FACTORS GOVERNING PRECIPITATION

Precipitation plays a role in buffering the surface layer acidity of the pit lake as well as diluting concentrations of dissolved solids. The neutralizing effect of rainwater exists despite the fact that precipitation is slightly acidic. In the absence of industrial emissions of acid constituents, such as sulfur oxides (SO_x) and nitrogen oxides (NO_x), the typical pH of rain is close to 5.6, primarily on account of the interaction between cloud water and carbon dioxide. In the case of an atmosphere affected by SO_x and NO_x, most notably downwind of large industrial centers, the mean pH of rainwater is closer to 4.5 and can be as low as 3.9 (Parungo et al. 1987). The potential buffering effect of rainfall to a pit lake is therefore sensitive to the geographic location with respect to the average wind directions, urban centers, and coal-based power plants. A further complication lies in the potential for rain and overland flow to scour the mineral-laden catchment area of the pit lake, thereby increasing the sediment load and acidity of pit lake inputs. Such factors depend on the permeability and competence of the catchment surface as well as the precipitation rate and duration.

Precipitation is driven by the conversion of water from its gas phase to its liquid or solid phase. This process is favored by rising motions in the atmosphere that may occur on the convective scale (~10 km in width) or the synoptic scale (~1,000 km in width). Regardless of the horizontal spatial scale, the lifting of atmospheric layers usually occurs within a few kilometers above the earth's surface. Ascent can either be forced by jet stream dynamics or, in the case where the temperature falls quickly with height, result from moist convection driven by buoyancy (i.e., thunderstorms). It is a fundamental goal for a numerical forecast model, whether regional or global in nature, to accurately predict the magnitude and location of such rising motions as well as evaluating the degree to which moisture is present in the atmospheric layer to be lifted.

Much of the rainfall in the United States, especially during the cold season, is stratiform in nature, meaning that the precipitation falls from layered clouds of large horizontal extent that exist within environments that offer some resistance to rising motion. Such stable layers have temperatures that decrease at a lesser rate with height than do saturated air parcels. For these cases, variations in jet stream velocity and orientation ultimately supply the needed work against gravity to produce the slow and steady rising motions required to initiate and sustain precipitation. Numerical models are relatively good at predicting such synoptic-scale forcing. Models have more difficulty in accurately forecasting convective precipitation, which depends more on smaller-scale, nonlinear, atmospheric, geographic, and storm-to-storm interactions.

Convective precipitation occurs in air where the temperature falls more quickly with height than saturated air parcels, thus promoting buoyancy. The resulting vertically developed clouds are commonly called thunderstorms if the updrafts are strong enough to produce a lightning discharge. Though relatively small in horizontal extent, convective updrafts support vertical velocities that greatly exceed those found in stratiform clouds. Therefore, it is not uncommon for thunderstorms to produce localized rainfall rates in excess of 5 inches per hour, most especially in environments where cloud bases are warm (i.e., tropics and subtropics). The area covered and the vigor of convective rainfall is further enhanced when jet stream dynamics also contribute to lift. Moist convection is most common during the warm season when the air close to the surface is the warmest. However, the southeast and south-central United States may have a significant portion of their cold season rainfall take the form of convective showers when strong dynamics are in place.

The spatial distribution of convective rainfall is often heterogeneous and is influenced by interactions between the terrain and low-level air flow. Most notably in regions of complex terrain, it is not uncommon for a given location to receive extreme convective rainfall, whereas a neighboring area just 10 km away receives little if any rainfall. In the United States, the overall stochastic nature of convective rainfall contributes to a significant degree of uncertainty in the forecast of annual precipitation values east of the Rocky Mountains where a significant percentage of the total rainfall is convective. Figure 3.1 illustrates the annual mean precipitation in the United States and the percent of the annual total that is convective.

FACTORS GOVERNING EVAPORATION

Evaporation describes the transformation of liquid water to gaseous water, known as water vapor. The evaporation process is effective in removing fresh water from pit lakes. Evaporation initiates a molecular reconfiguration from a more-ordered liquid phase to a less-ordered gas phase. Such a phase transition requires the addition of energy. The energy required to evaporate a unit mass (1 kg) of water is substantial, roughly equivalent to 2.4 million joules of energy. The majority of this energy is supplied from radiation. This external source of energy is strongest during daylight hours when the sun is shining.

Evaporation is also affected by the ambient concentration of water vapor in the air overlying the lake surface. The percentage of total air molecules that are water vapor are much less than the

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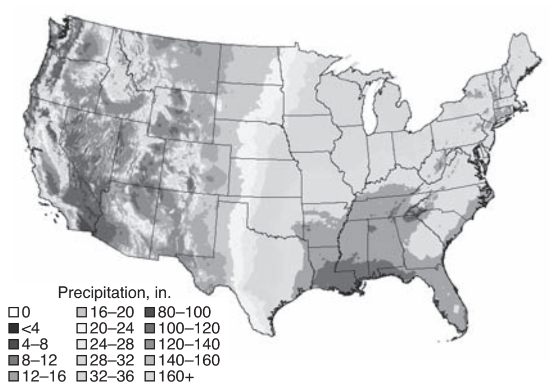
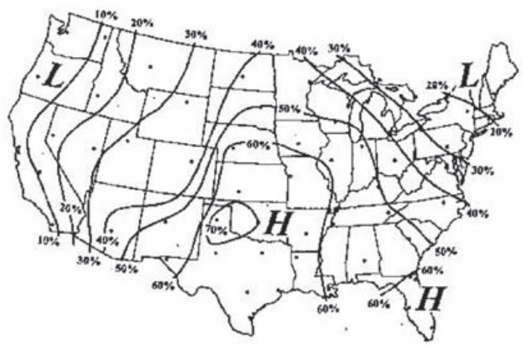


Source: Copyright ©2006, PRISM Group, Oregon State University, www.prismclimate.org. Map created June 16, 2006.

B



Source: Adapted from Changon 2001.

FIGURE 3.1 (a) Mean annual (1971–2000) rainfall observed in the continental United States. (b) Mean percent contribution of annual convective rainfall to total annual rainfall.

dominant components of air, but unlike nitrogen and oxygen, water vapor concentrations are highly variable.

Water vapor percentages range from near 0.5% (cold/dry conditions) to almost 4% (hot/humid conditions) of the total air molecules. Such concentrations are more conveniently expressed in terms of *vapor pressure* (e), which typically ranges from a few millibars (mb) to 40 mb, respectively. The *relative humidity* indicates the ratio of water vapor present in a volume of air to the water vapor holding capacity of that air. One may formulate either a mass-based (mixing ratio) or pressure-based (vapor pressure) representation of the water vapor concentration. The water vapor holding capacity, indicated by *saturation vapor pressure* (e_s), varies exponentially with temperature (Figure 3.2). This is to say that warmer parcels of air may sustain much higher vapor pressures (mixing ratios) than cooler parcels.

For a rising air parcel, condensation will begin when its temperature drops to the *dew point temperature* (T_d). At this temperature, the air can no longer sustain the volume of water vapor initially contained within the parcel. Using the curve in Figure 3.2, the vapor pressure (e) is equivalent to the value on the y-axis that corresponds to the dew point temperature on the x-axis, consistent with the following formula:

$$e = e_s(T_d) \tag{EQ 3.1}$$

Relative humidity (RH) is given by the ratio:

$$\text{RH} = \frac{e}{e_s} \times 100\% \tag{EQ 3.2}$$

The dew point temperature therefore expresses the degree to which air must be cooled for the relative humidity to reach 100%, holding all other parameters constant.

In the unsaturated case, where vapor pressure is less than the saturation vapor pressure, evaporation will proceed. Under this condition, bodies of water will tend to convert liquid water to water vapor at a rate that is proportional to the difference between the saturation vapor pressure of the air immediately above the water surface and the dew point temperature of the ambient air measured 2 m above the surface. This quantity is called the *vapor pressure deficit* (VPD):

$$\text{VPD} = e_s - e \tag{EQ 3.3}$$

Here the saturation vapor pressure is evaluated using the surface water temperature, and vapor pressure is evaluated using the dew point temperature of the overlying air mass.

For example, a water surface at 20°C exposed to air having a dew point temperature of 5°C will have a vapor pressure deficit of 14.7 mb. At 20°C the saturation vapor pressure is 23.4 mb. Using the dew point temperature of 5°C, the vapor pressure is 8.7 mb (Figure 3.2). Using Equation 3.3, it follows that the vapor pressure deficit is equal to the difference.

In the previous example, a constant replenishing of undiluted fresh air is assumed so that the layer of air immediately above the water surface is constantly in contact with, or exposed

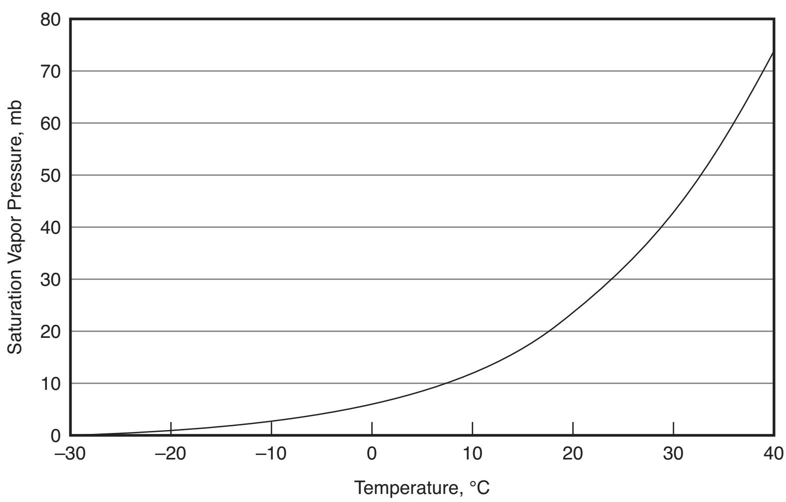


FIGURE 3.2 The graph depicts the saturation vapor pressure over a flat surface of pure water as a function of temperature

to, air having a 5°C dew point. In reality, however, the evaporation process produces a boundary layer of air a few centimeters thick immediately above the evaporating surface, which, over time, becomes loaded with water vapor. In the absence of circulation currents (i.e., wind-induced turbulence), the evaporating surface is exposed to a thin, moist layer of air that has a higher vapor pressure than the ambient air at 2 m. Such moisture stratification tends to reduce the evaporation rate. The ability to mix ambient, relatively dry air down to the evaporative surface is largely dependent on the wind speed or turbulence occurring near the water surface. In summary, the three factors that govern evaporation rate are solar radiation, vapor pressure deficit, and wind speed.

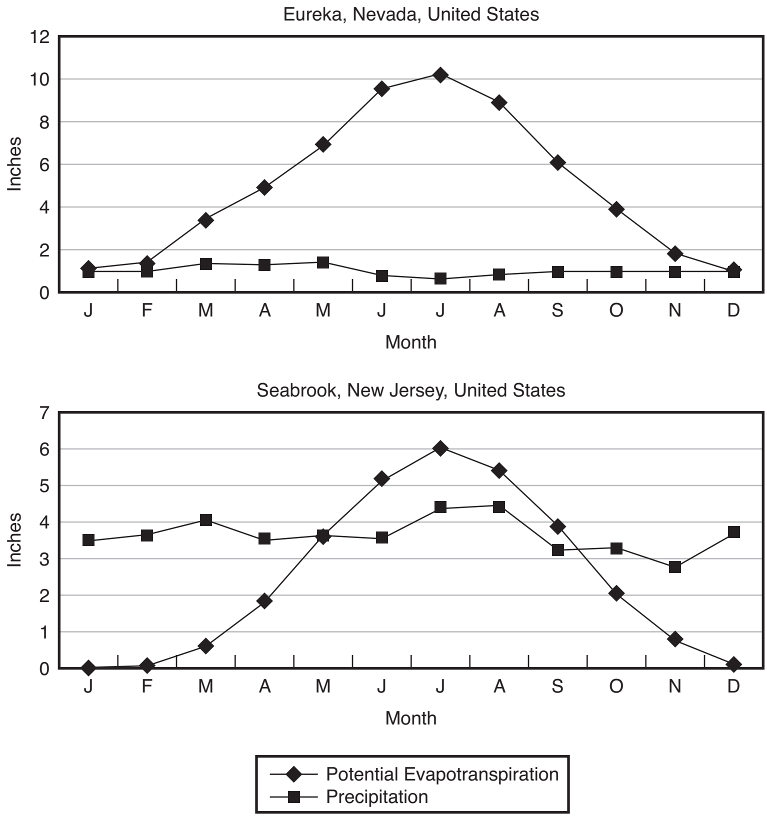
DETERMINING THE METEORIC WATER BALANCE

Potential evapotranspiration is the amount of water that would be lost from the surface of a pit lake due to the combination of evaporation and transpiration by plants, assuming an infinite supply of water is available. For most pit lakes, potential evapotranspiration is the same as actual evaporation. Outside of the tropics, in the Northern Hemisphere, potential evapotranspiration trends upward during the spring, reaching a seasonal maximum just after summer solstice when the days are long and temperatures are warm. Potential evapotranspiration decreases into autumn as the sun retreats further south in the sky, reaching a minimum during the days with shortest daylight hours in winter (Figure 3.3). This is to say that potential evapotranspiration is highly correlated to radiation, both directly and indirectly through the associated warm temperatures that allow for large vapor pressure deficits.

Dunne and Leopold (1978) provide several methods to estimate potential evapotranspiration. To directly measure evaporation, a "class A" evaporation pan can be installed on-site and monitored daily for the amount of water lost from the pan. Alternatively, the Thornthwaite method provides a means for calculating potential evapotranspiration using average monthly air temperature and the latitude of the pit lake. Potential evapotranspiration can also be calculated using an energy balance approach, which requires the air temperature (measured 2 m above the lake surface), dew point temperature, water temperature, precipitation depth, total solar radiation, and wind velocity (NOTE: similar data are required for limnologic modeling, as discussed in Chapter 9). The energy balance calculations show closer agreement with pan evaporation observations than the Thornthwaite method; however, the latter is simpler and requires less input data. Both the Thornthwaite method and the energy balance approach can be used in advance of mining using historic data from nearby weather stations. Initial predictions can be refined using data from an on-site weather station installed as close to the future water body surface as possible. Because the potential evapotranspiration rate is affected by wind velocity, air temperature, and dew point, a sampling interval on time scales of tens of minutes is desirable.

With these data available, the first step in determining the approximate meteoric water balance for a pit lake is through the analysis of a time series of potential evapotranspiration and precipitation. Figure 3.3 compares time series data representative for the arid Basin and Range province of the western United States (measured at Eureka, Nevada) to representative data for the northeast coast (measured at Seabrook, New Jersey). To construct such a plot, it is desirable to select a weather station that has daily weather records that span at least 25 years so that a complete solar cycle as well as decadal cycles in ocean temperatures are represented. Both are known to affect patterns of both cold and warm season precipitation in the United States (Meehl et al. 2003; Yu 2002; Hu et al. 1998; Ting and Wang 1997; Graham 1993; Landscheidt 1983). A 25-year time scale also ensures the inclusion of at least a few El Niño and La Niña events.

It is common to see plots of *net precipitation* (precipitation minus potential evapotranspiration) compiled on monthly time scales. In this fashion, changes in lake water storage (deficits/surpluses)



Source: Adapted from Thornthwaite and Mather 1955.

FIGURE 3.3 Precipitation and potential evapotranspiration time series that are typical for a semiarid western U.S. station (top) and a relatively moist East Coast station

are related to periods of negative and positive excursions from steady-state conditions. Pit lakes are likely to lose or gain fresh water during these periods. When constructing a plot of this nature, it is imperative to use consistent units (i.e., inches or centimeters).

ESTIMATING PRECIPITATION AND EVAPORATION RATES AT NEW MINE SITES

It is not uncommon for the nearest weather station to be tens to hundreds of kilometers away from a proposed open pit mine site. Considering the fact that many mining sites are located in complex terrain, estimates of both evaporation and precipitation for a future pit lake are compromised when using nonlocal data. Complex terrain can alter incoming radiation, wind velocity, and both the frequency and rate of precipitation. For mines in mountainous terrains, caution should be used in retrieving raw data from nearby observation stations located along flat, open plains, such as a small airport or landing field. Should this be the only option, the data should to be adjusted using statistical and/or dynamic downscaling procedures (Kidson and Thompson 1998; Murphy 1999; Benestad et al. 2008).

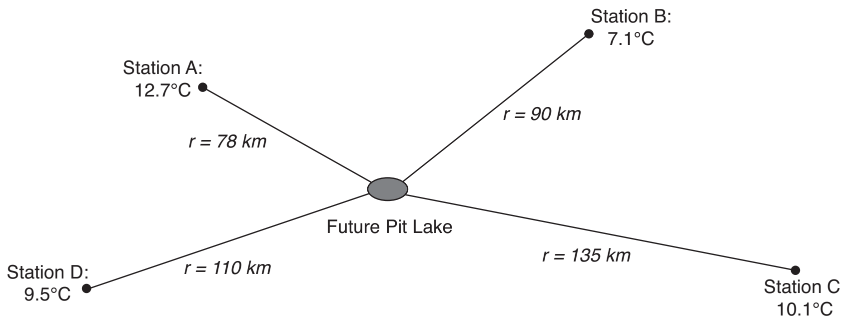


FIGURE 3.4 Diagram showing the setup of a Barnes scheme employed to estimate an unknown Figure 3.4 climate parameter at a future pit lake from data collected at surrounding weather stations. In this example, temperatures measured at four weather stations at distance r are used to estimate the temperature of the future lake.

Barnes Scheme Estimation for Homogeneous Terrain

The concept behind statistical downscaling is to generate statistical relationships between observed station data and larger-scale data using either analogue methods, regression analysis, or neural network methods. Future values of the large-scale variables obtained from global climate model (GCM) projections of future climate are then used to drive the statistical relationships in order to estimate the smaller-scale details of future climate. Dynamical downscaling uses a limited-area, high-resolution model, sometimes called a regional climate model, that is driven by boundary conditions from a GCM to derive smaller-scale information.

Assuming data is available at a number of neighboring stations, a Barnes scheme is a good starting point to approximate data for a future pit lake (Figure 3.4). It should be emphasized that this approach neglects the effects of complex terrain. In this scheme, the surrounding weather stations are each assigned a weight based on their distance (r) to the future pit lake and a smoothing parameter (a). The smoothing parameter is a user-defined term that is approximately equal to the radius of influence surrounding the pit lake. Typically, this is on the order of a few hundred kilometers. After appropriate values for r and a are determined, the weighting function for each weather station (w_i) can be calculated:

$$w_i = \exp\left(\frac{-r_i^2}{2e^2}\right) \tag{EQ 3.4}$$

The exp function is 2.718 to the bracketed power. After the weight of each weather station is determined, then the parameter of interest (Z_x) can be estimated based on the observed parameter (Z_i) measured at n, the number of weather stations within the radius of influence:

$$Z_x = \frac{\sum_{i=1}^{n\,obs} Z_i w_i}{\sum_{i=1}^{n\,obs}} \tag{EQ 3.5}$$

For example, suppose that the average air temperature for the month of April needs to be determined for a future pit lake in Utah (United States). Figure 3.4 shows average April air temperatures recorded at four weather stations near the future pit lake along with the distance between the lake and each station. The pit lake has a radius of influence of 150 km. Using distances and the smoothing factor, one first computes the weighting functions. Next, the product of the temperatures and the weighting functions specific to each station are computed and added to justify the numerator in Z_x. The denominator is computed by summing the weighting functions alone. The quotient will return the estimated air temperature value for April for the future pit lake, in this case 9.88°C:

$$w_1 = \exp\left(-\frac{78^2}{2(150)^2}\right) = 0.8735 \times 12.7 = 11.09$$

$$w_2 = \exp\left(-\frac{90^2}{2(150)^2}\right) = 0.8353 \times 7.1 = 5.93$$

$$w_3 = \exp\left(-\frac{135^2}{2(150)^2}\right) = 0.6669 \times 10.1 = 6.74$$

$$w_4 = \exp\left(-\frac{110^2}{2(150)^2}\right) = 0.7642 \times 9.5 = 7.26$$

$$\sum_{i=1}^{4} w_i = 3.14 \quad \text{and} \quad \sum_{i=1}^{4} Z_i w_i = 31.02$$

$$Z_x = \frac{\sum Z_i w_i}{\sum Z_i} = \frac{31.02}{3.14} = \boxed{9.88}$$

Issues Associated with Complex Terrain

Pit walls will also affect the amount of solar radiation received by the lake surface and the wind speed across the lake. An elevated location is likely to experience stronger winds than would exist at more protected lower elevations. However, pit lakes lie in human-made excavations, which tend to shelter the lake surface from the effects of stronger winds existing on the surrounding mountainsides. Light shading by pit walls will also reduce exposure to solar radiation and further reduce the evaporation rate.

Variability in the degree of nonconvective precipitation in complex terrain is dependent on the climate of the geographic region and on the location of the pit lake relative to the typical winds observed during precipitation events. In the eastern United States, for example, the direction of the "moist airflow" during storm events may differ from the prevailing wind. In a typical East Coast winter storm event, the eastern side of the Appalachians, which face the moisture source (the Atlantic and/or Gulf of Mexico), tend to receive enhanced precipitation by providing for orographic lift, by focusing low-level convergence, and through the generation of gravity waves. Such local enhancements are typically offset by decreased rainfall on the leeward side of the barrier. An example of such a precipitation dipole typically occurs between the piedmont of the Carolinas (rainfall enhancement) and the coal fields of eastern Tennessee (rainfall reductions). Although this process is at times significant along the Appalachians, the effects of topography upon precipitation are even more dramatic along the Cascades of Washington and Oregon, the Front Range of the Rockies, the Black Hills of South Dakota, and the Sierra Nevada of California/Nevada.

Pit lakes that are exposed to winter weather are subjected to the complication of snow storage. Snowfall on top of an ice sheet protects the pit lake from the relatively small amount of sublimation that would otherwise occur during the cold season. The storage of snow will tend to delay but at the same time enhance the melting process in a way that upon melting, a rapid influx of relatively fresh water will be deposited into the pit lake, altering the chemistry of the surface water. For this reason, information about the mean thickness and depth, aerial coverage, lifetime, and ultimately the volume and chemistry of the snowpack and ice may be of interest.

CLIMATE CHANGE IMPACTS ON CLIMATE PREDICTIONS

The mean temperature of the lower atmosphere has warmed 0.6° to 0.7°C over the course of the 20th century. The Intergovernmental Panel on Climate Change (IPCC 2007) believes this warming is the result of enhanced down-welling of terrestrial infrared radiation on account of anthropogenic emissions of greenhouse gases. Most general circulation models (GCM) forecast an additional 1.5° to 3.5°C increase in global mean temperature before the end of the 21st century. The greatest warming (in excess of the global mean) is expected to occur low in the atmosphere, over land, at middle to high latitudes where water vapor concentrations are relatively low. Sections of the interior United States and Canada are among the areas that are expected to warm in excess of the global mean. These changes are likely to affect mean precipitation and evaporation values used in pit lake prediction models; hence, predicted values may diverge from observed values over time, affecting the accuracy of long-range pit lake predictions.

Global climate models predict smoothed climate parameters over a large domain resulting from climate change. For future climate predictions applied to pit lakes, statistical or dynamic downscaling techniques represent the best approach in determining precipitation and evaporation at a point within the larger domain (see the "Barnes Scheme Estimation for Homogeneous Terrain" section previously in this chapter). This is an area for future collaboration between meteorologists and mine managers. It is useful for individuals generating and reviewing pit lake predictions to be aware of the limitations of climate models on which pit lake predictions are based.

The atmospheric response to a greenhouse warming scenario can be subdivided into dynamic and thermodynamic components. The former concerns itself with altered jet stream and global circulation patterns, while the thermodynamic contribution is a consequence of altered air properties that affect the nature of the condensation and evaporation processes. What follows is a brief overview of the extremely complex earth–atmosphere response to a greenhouse warming scenario.

Thermodynamic Response to Climate Change

A warming of the surface temperature increases the holding capacity for water vapor, thereby increasing the evaporation rate. Keeping in mind that the rate of evaporation is proportional to the vapor pressure deficit, the exponential dependence of saturation vapor pressure upon temperature illustrates the fact that small changes in temperature yield relatively large gains in the ability for the air to sustain higher water vapor concentrations. The slope of the vapor pressure and mixing ratio curves, in the vicinity of 15°C, show that for every 1°C increase in global temperature, the atmosphere's ability to sustain water vapor increases by about 7% (Figure 3.2).

Stronger evaporation will tend to increase water vapor concentrations near the surface, which will curtail the stronger evaporation rates until a balanced state is reached. The GCMs strongly suggest that such equilibrium condition is represented by a fixing of the mean relative humidity in the troposphere. Such a constraint is expected to yield a 2% increase in global mean evaporation per 1°C of warming (Held and Soden 2006). However, pure relative humidity conservation is most likely to be observed in maritime climates and least likely to occur in areas where surface water sources are limited (Seager et al. 2007). Yet, even in these drier regions, mainly toward the interior of continents, a commensurate rise in mean dew point temperature is still expected as the lower atmosphere struggles to meet this criterion.

The GCMs suggest that increases in precipitation will not be enough to account for the relatively stronger increase in evaporation. One consequence of evaporation increasing faster than precipitation is that the residence time for water vapor in the atmosphere will increase. Held and Soden (2006) contend that more water vapor will be transported through the mid-latitudes instead of condensing there. Such a pattern suggests that low-level convergence and subsequent condensation will be repositioned, on average, north of 40° latitude, likely contributing to precipitation increases along the northern tier of the United States and southern Canada.

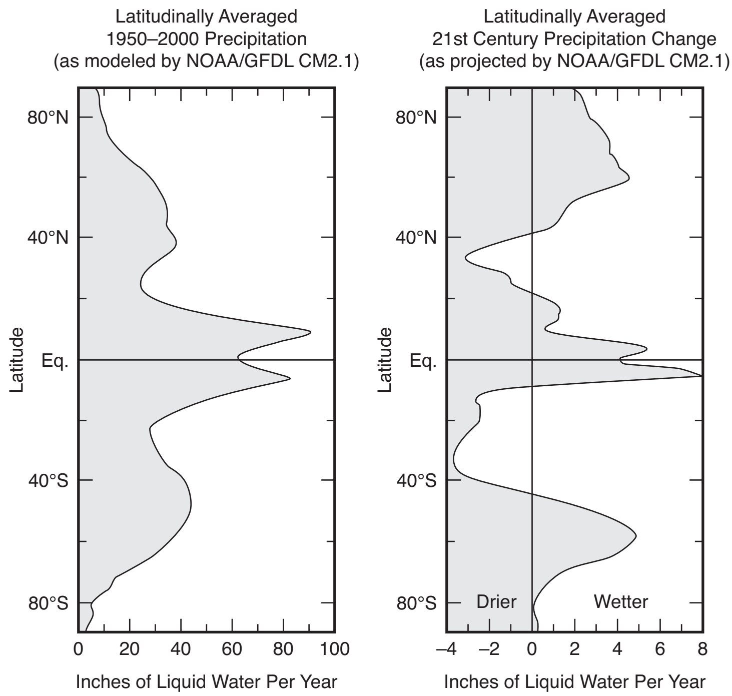
The expected increase in both surface air temperature and dew point temperature will have the effect of causing air parcels to achieve saturation at warmer temperatures. A warming of cloud bases can significantly increase rainfall rates. For example, if a thunderstorm's cloud base is normally at 10°C (8.6 g of water vapor per kilogram of dry air) and under a climate change scenario the cloud base temperature increases to 12°C (9.6 g/kg of vapor per kilogram of dry air), simple thermodynamic arguments imply nearly a 12% increase in rainfall rate. Such thermodynamic alterations do not necessarily imply more thunderstorms, rather they suggest that those storms that do manage to form have an increased potential of producing extreme rainfall rates.

Dynamic Response to Climate Change

The GCMs applied to greenhouse warming scenarios strongly suggest a poleward expansion and slight weakening of the Hadley cell circulation (Lu et al. 2007). The sinking air along the poleward extent of this transverse circulation is a fundamental cause of the subtropical dry zones that normally exist between 20° and 30° latitude. These widespread areas of mean sinking atmospheric motion are expected to migrate northward from their historic position so that portions of the southern tier of the United States are expected to become drier. Figure 3.5 displays the subtropical minimums in precipitation as well as the expected trends over the 21st century.

In a related fashion, the overwhelming majority of the GCMs suggests a poleward shift in the mean relative position of the jet streams and associated storm tracks. As both the subtropical and polar jet streams migrate poleward, the dynamics needed to produce stratiform precipitation shifts with them. Relative to the northern hemisphere, a northward displacement of the track of major low-pressure centers will contribute to general increases in precipitation north of 45° latitude and precipitation decreases south of 35° latitude. This is consistent with the aforementioned increased water vapor transport across mid-latitudes. It follows that the net effect of a poleward-displaced Hadley cell circulation contributes to reducing precipitation minus evaporation (drier conditions) south of 35°N latitude while providing for increases (wetter conditions) north of 45°N latitude (Lu et al. 2007).

In summary, there is a general modeling consensus that the already semiarid climate of the interior western United States will trend toward an even drier climate as a consequence of climate change. The GCMs indicate a particularly strong likelihood of evaporation rates exceeding precipitation rates (i.e., net drying) in the southwestern United States, exacerbating existing



Source: NOAA/GFDL 2007; reproduced with permission of the National Oceanic and Atmospheric Administration/Geophysical Fluid Dynamics Laboratory.

FIGURE 3.5 The observed 1950–2000 latitudinal mean precipitation (left) and the expected trend in latitudinal mean precipitation during the 21st century (right), based on a climate model that is in close agreement with the GCM consensus

moisture deficits. Some modeling studies cite increases in heat-driven evaporation as the primary force behind drying (Hoerling and Eischeid 2007), whereas other studies cite a reduction in annual precipitation as the primary mechanism (Seager et al. 2007). Because notable warm-season precipitation deficits are already in place in the interior western and southwestern United States (Figure 3.3), the expected drying trend will magnify this preexisting pattern. East of the Rockies, however, stronger evaporation in conjunction with a greater likelihood of drought and flooding episodes will further complicate the forecast for the annual water balance.

It is worthy of mentioning that variations around the mean climatic conditions are not reflected by average values. Such variability is perhaps more significant than changes in the mean itself. This is an important point, considering that the GCMs predict an increased likelihood for hydrologic extremes.

CONCLUSIONS

Precipitation and evaporation are important input parameters for the development of pit lake models. In order to formulate a water budget specific to an existing pit lake, it is desirable for an instrumentation platform to be placed as close to the pit lake as possible. This is especially true in light of the fact that pit lakes are often located in remote areas of complex terrain and are surrounded by pit walls. This type of environment can significantly alter precipitation and evaporation patterns and may necessitate on-site instrumentation to record precipitation and estimate evaporation.

Under the expected climate change scenario, the mean temperature of the planet is expected to rise 1.5° to 3.5°C during the course of the 21st century. An even greater increase in the global mean dew point temperature should accompany the warming, which will increase water vapor concentrations in the atmosphere. Climate change also is expected to cause the jet streams to move poleward. In the United States, a northward migration of the storm track is expected to contribute to precipitation increases north of 45°N latitude and precipitation decreases south of 35°S latitude. Combined dynamic and thermodynamic arguments suggest increases in the magnitude and frequency of flooding events. At the same time, episodic drought conditions are expected to become more frequent, most especially in the southern one-third of the United States. The greatest likelihood for drought conditions are expected for the desert southwest areas of the United States.

Modelers should use a high degree of scrutiny when determining local precipitation and evaporation rates for use in pit lake prediction models. As shown in Figure 1.3 of Chapter 1, these climate parameters form the basis for water balance, hydrology, limnology, and ultimately, geochemistry predictions, such that small errors between predicted and observed climate values can decrease the accuracy of water quality predictions. In the absence of on-site weather observations, regional data should be adjusted using downscaling techniques in order to account for differences in topography, wind sheltering, solar radiation, and so forth. Models should account for monthly variations in precipitation and evaporation, as shown in Figure 3.3.

Ideally, geochemical predictions that span 10 to 100 years or more should account for expected variations in precipitation and evaporation rates resulting from global climate change. The simple assumption that mean annual evaporation and precipitation rates will remain constant over century-scale time spans is likely to generate errors between observed and predicted water quality values. Unfortunately, climate model output is subject to some error. Pit lake modelers should be aware that this data gap exists, and that it may reduce the accuracy of long-range predictions whereby the magnitude of the error is proportional to the duration of the prediction. For this reason, uncertainty associated with global climate change may limit the practical duration of pit lake predictions.

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CHAPTER 4

Hydrologic Characteristics and Classifications of Pit Lakes

W.L. Niccoli

INTRODUCTION

The mining process—the act of extracting geologic materials from the earth's crust—always ends with a "hole in the ground." The nature of this hole can range from large, cavernous openings, kilometers in length beneath the ground surface, used to extract base and precious metals to a small pit outside of an urban area where gravel was once extracted. The latter technique, open pit mining, is likely the most common material extraction methodology utilized in mining today. Open pit mining techniques are used for extracting commodities such as sand and gravel, base metals (e.g., copper and zinc), and precious metals (e.g., silver and gold). Figures 4.1 and 4.2 provide typical examples of open pit mines. Open pit mining often intersects the groundwater table, such that when the extraction process is complete, the open pit will remain and, in many cases, may form a pit lake. The focus of this chapter is to discuss a handful of aspects related to the hydrology of open pits or pit lakes after mining ceases, such as

- How do "flow-through" pit lakes differ from "terminal" pit lakes?
- What is known about the water quality of these lakes from observations of existing pit lakes?
- What impact will climate change have on hydrology?
- How does artificial flooding of pit lakes with surface water affect groundwater input?
- What can be expected in future pit lakes?
- What are the current data gaps?

Woodhouse (2002) provides a grouping of articles that describe a wide range of topics associated with mine pit lakes.

HYDROLOGIC STATUS OF PIT LAKES

Two types of hydrologic conditions exist in pit lakes:

1. Flow-through conditions—surface and/or groundwater flows into and out of this type of lake (Figures 4.3 and 4.4).
2. Terminal conditions—groundwater flows into the pit and outflow occurs only as evaporation (Figure 4.5).

Flow-through pit lakes are common in areas where rainfall exceeds evaporation, in highly productive aquifers where groundwater inflows exceed evaporation rates (e.g., alluvial aquifers), and any time that the net water balance surrounding the pit is positive. Another type of a flow-through

FIGURE 4.1 Gravel quarry pit lake, Grand Junction, Colorado, United States

FIGURE 4.2 Small mine pit lake with adits, arid region of southwestern United States

pit lake is one that exists above the water table and is filled by surface water. Outflows consist of vertical leakage and evaporation.

Terminal pit lakes are common in the arid areas of the world where evaporation exceeds rainfall/precipitation or any time that the net water balance surrounding the pit is negative. With seasonal or long-term climatic changes, the hydrologic status of a pit lake may fluctuate between terminal and flow-through. The Martha mine in New Zealand (Ingle 2002) is an example of a uniquely engineered flow-through pit lake. After closure, the plan is to place a drainage pipe below the premining groundwater elevation that will fix the elevation of the postmining pit lake. Whereas all local groundwater will flow into the lake, the discharge of surface water will make the pit lake exhibit flow-through conditions. A terminal pit lake would have evaporation as the only discharge.

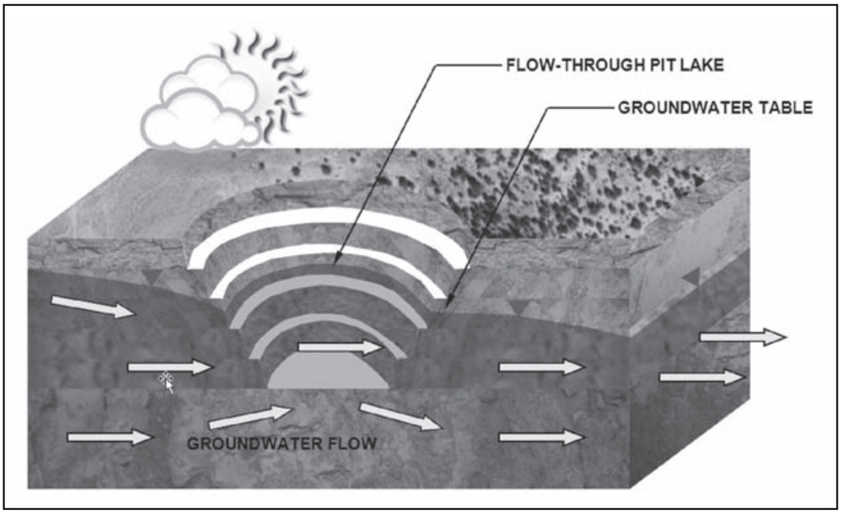

FIGURE 4.3 Flow-through pit lake below the groundwater table

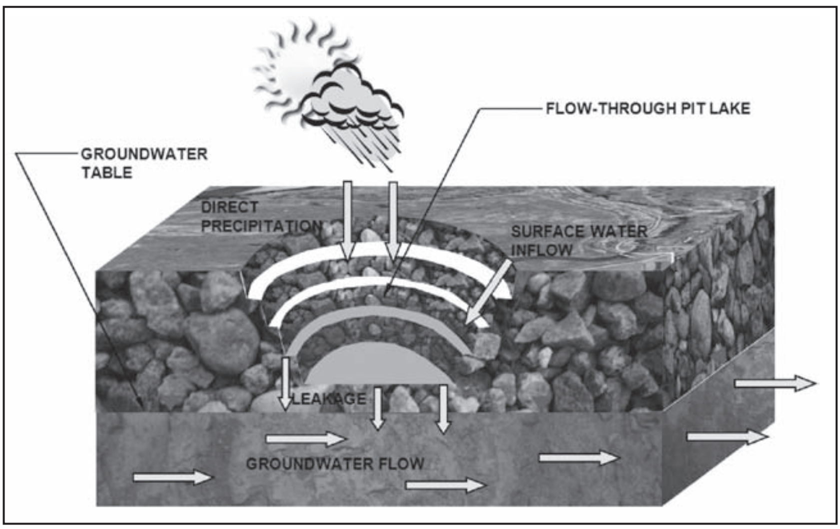

FIGURE 4.4 Flow-through pit lake above the groundwater table

In the United States, the Sweetwater pit in Wyoming is an example of a terminal pit lake, whereas the Berkley pit in Montana, if left unmanaged, is expected to become a flow-through pit lake.

To determine which condition exists for any particular pit lake, measurements and observations of the hydrologic components of the pit lake are key. Groundwater elevation data surrounding the pit lake, pit lake elevations, precipitation, and surface water inflows and outflows are parameters that should be measured. Maps of groundwater heads, expressed as groundwater elevations (i.e., potentiometric maps), and flow nets built from the collected groundwater head data can show the hydrologic status of a pit lake (Figure 4.6). Predicting the hydrologic condition of a pit prior to its filling and reaching steady state is a different endeavor and is covered

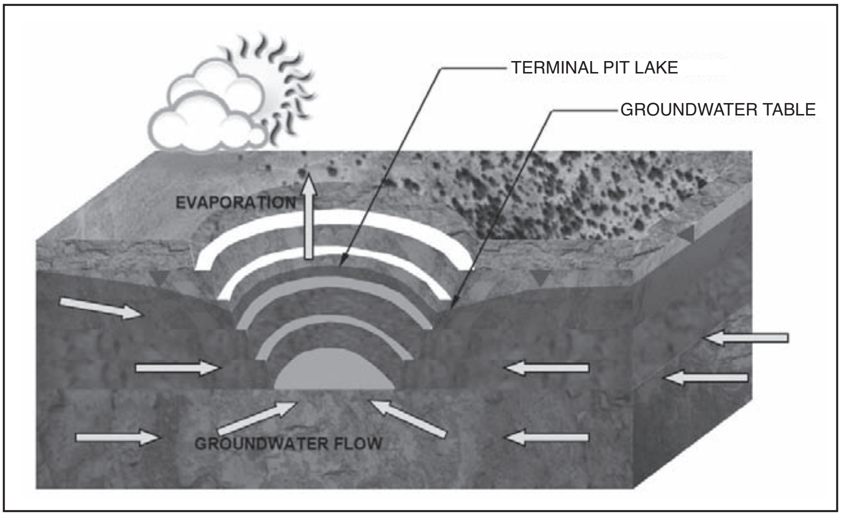

FIGURE 4.5 Terminal pit lake—all outflow is from evaporation

in Chapter 8. However, the same measurements required to determine the hydrologic status of a lake after filling are helpful in aiding predictions before filling.

Two possible scenarios are encountered when an evaluation to determine the hydrologic status of a pit lake is undertaken: (1) Information exists for the premining hydrologic conditions at the site and good records of dewatering rates at different mine elevations exist, and (2) little to no historic data are available. Scenario 1 provides the most robust method of evaluating the hydrologic status of a pit lake. Historical information can be used to quantify water balance components (Figure 4.7) and an estimate of the pit lake water balance (i.e., summing the inflows and comparing them to the outflows) completed. By evaluating available groundwater elevation data, comparing them to known pit lake water elevations, and examining the water balance, the hydrologic status of the pit lake can be estimated. For example, if the dewatering rate at a given elevation matches the net evaporation (i.e., the balance of direct precipitation, runoff, and lake evaporation) at the same elevation, the lake is likely at a steady state. If groundwater elevation data surrounding the pit lake are all higher than the lake surface elevation, the lake is terminal.

Scenario 2 presents a more difficult challenge; however, both a water balance and an estimate of the groundwater conditions must be made. Water balances without measurements require that estimates be made from available (or newly collected) data. It is beyond the scope of this chapter to describe all of the methods of estimating the various components of the hydrologic balance associated with a pit lake. The reader is therefore pointed to classic textbooks such as McWhorter and Sunada (1977), Freeze and Cherry (1979), Watson and Burnett (1993), and Schwab et al. (1981) to gain a fundamental understanding of these components. Provided herein is an example of evaluating the hydrologic status of a pit lake using a hypothetical pit lake.

Presuming that the owner of the hypothetical pit lake shown in Figure 4.6 purchased the property recently and that any hydrologic records from the previous owner had burned in a fire and were no longer available. The new owner wanted to estimate the amount of water flowing through the pit for regulatory purposes (e.g., to satisfy water rights regulations or meet discharge permit requirements). Given that there are wells in the area, he sent out a team from his

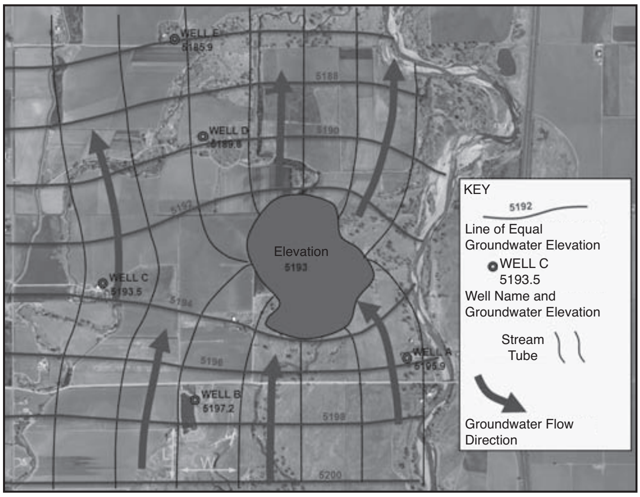

FIGURE 4.6 Potentiometric map and flow net associated with a flow-through pit lake. Water elevations are given in feet (1 foot = 0.3048 meters).

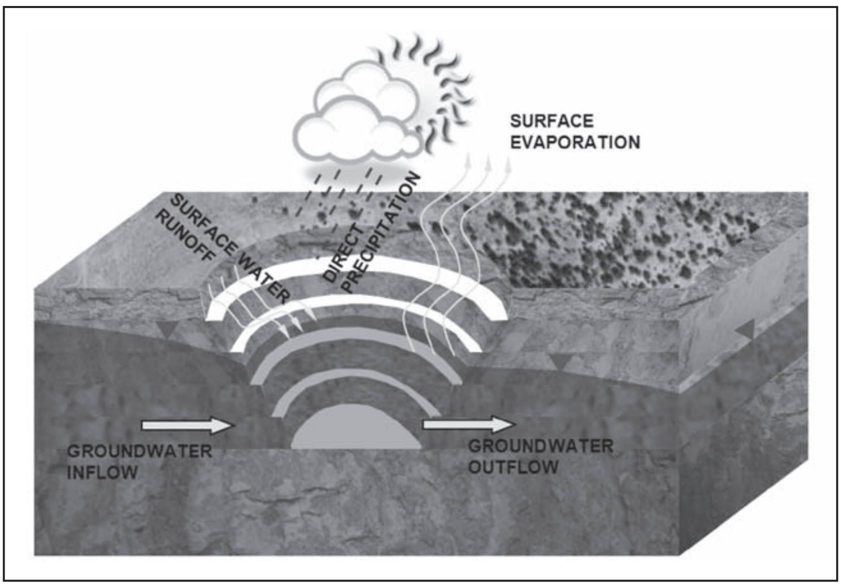

FIGURE 4.7 Hydrologic processes associated with a pit lake

environmental department to measure water elevations in the wells and the pit lake. He also had them perform aquifer tests in all the wells. Concurrently, the team searched the Internet and found the mean annual precipitation and pan evaporation rates from a nearby governmental weather station. Table 4.1 shows the data that were gathered during the investigation.

The team created the groundwater elevation and flow map shown in Figure 4.6, which shows that the pit is likely flow-through. To calculate the groundwater flow to the pit, they used Darcy's law and the flow net shown on Figure 4.6. These calculations are shown in Table 4.2. Table 4.2

TABLE 4.1 Example of field hydrologic investigation results

Well Name	Water Elevation, ft	Water Elevation, m	Hydraulic Conductivity, ft/d	Hydraulic Conductivity, m/d
Well A	5195.5	1583.5	375	114.3
Well B	5197.2	1584.0	700	213.3
Well C	5193.5	1582.9	350	106.7
Well D	5189.8	1581.8	600	182.9
Well E	5185.9	1580.6	150	45.7
Pit	5193.0	1582.7		
Geometric mean			383	116.7
Geometric mean, wells A–C			451	137.5
Mean annual precipitation	8 in./yr	20.3 cm/yr		
Mean annual pan evaporation	72 in./yr	182.9 cm/yr		

also shows that the outflow is approximately 85% of the groundwater inflow. This reduction in flow is represented by the narrower width of the stream tubes downgradient of the pit in Figure 4.6.

This straightforward approach is just an example of one method to estimate the hydrologic status of a pit lake, but it can be quite robust and provides a good idea of the magnitude of the flows associated with the pit lake. However, one should exercise diligence when performing any kind of analysis to make sure that other factors are not influencing the water balance associated with the pit. For example, unknown heterogeneity in the aquifer such as fractures, faults, and historic works may not be apparent in the investigation, yet they could have important effects on the actual flow. If these features are suspected to occur, further investigation is warranted.

HYDROLOGIC IMPACTS ON PIT LAKE WATER QUALITY

The water quality of a pit lake is a result of many contributing factors including but not limited to

- Geology and mineralogy of the host formations,
- Influent water quality,
- Concentrating effects of evaporation,
- Geochemical, biological, and limnologic processes within the lake, and
- Anthropogenic impacts.

The hydrology of a pit lake affects the water quality of the pit lake by changing the chemical mass balance associated with the lake.

To make a general statement that one type of hydrologic condition results in better water quality than another would be a misstatement on account of the influence of nonhydrologic processes. The ultimate disposition of pit lake water quality is a balance of all the effective processes. However, general statements regarding the impacts of individual hydrologic processes (Figure 4.7) on water quality can be made:

- Groundwater inflows carry dissolved constituents at background concentrations into the lake. Natural, upgradient groundwater will have a certain water quality and will contribute constituents to the pit lake. Inflowing groundwater can also pick up constituents from

TABLE 4.2 Calculation results

Darcy s Law	
$$Q = \text{Area} \times K \times \frac{\Delta H}{L}$$	
where Q = groundwater flow Area = cross-sectional area of flow K = hydraulic conductivity ΔH = change in hydraulic head over the flow length, L	
Data	
L = 2,500 ft	Distance between hydraulic head contours on one stream tube (see Figure 4.6).
W = 2,500 ft	Width of an upgradient stream tube (see Figure 4.6).
D = 350 ft	Depth of pit and aquifer
K = 451 ft/d	Geometric mean of measured hydraulic conductivity in the area upgradient and near the pit lake. Note: The geometric mean typically yields that best estimate of the effective hydraulic conductivity that represents the overall aquifer.
ΔH = 2 ft	Change in hydraulic head over the calculation length L
A = 451 acres	Surface area of the pit lake
$evap_{rate}$ = 72 in./yr	Pan evaporation rate
$precip_{rate}$ = 8 in./yr	Mean annual precipitation
Calculations	
$q_{streamtube} = W\ D\ K\ \Delta H/L$ $q_{streamtube}$ = 1,640 gpm	Flow in upgradient stream tube using Darcy s law
$No_{streamtube}$ = 4	Number of stream tubes flowing to pit lake
$Q_{gw\_in} = No_{streamtube}\quad q_{streamtube}$ Q_{gw_in} = 6,560 gpm	Groundwater inflow to the pit
Evap = $evap_{rate}\quad A\quad 0.7$ Evap = 1,173 gpm	Evaporation from the pit lake surface (0.7 is a typical factor to convert pan evaporation to lake evaporation rates)
Precip = $precip_{rate}\quad A$ Precip = 186 gpm	Direct precipitation to the pit lake
$Q_{gw\_out} = Q_{gw\_in}$ + Precip – Evap Q_{gw_out} = 5,573 gpm	Groundwater outflow from the pit based on a water balance—assuming no surface water run-on
Conversions	
1 gpm = 3.7854 Lpm	U.S. gallons per minute converted to liters per minute
1 acre = 4,047 m^2	Acres converted to cubic meters
1 in. = 2.54 cm	Inches converted to centimeters
1 ft = 0.3048 m	Feet converted to meters

Source: Adapted from Marinelli and Niccoli 2000.

weathered rock immediately surrounding the pit lake or from recharging meteoric water passing through the dewatered zone.

- Surface water runoff from pit walls may transport constituents and sediments that will affect the chemistry of the lake.
- Direct precipitation generally is a diluting factor on pit lake quality.

- Because evaporation takes out the water and leaves behind any dissolved constituents, evaporation tends to have a concentrating effect on pit lake water quality.
- Ground- and surface water outflows from pit lakes typically have the effect of removing constituents from the lake. However, if the lake is stratified, these outflows may remove water of different quality, which may result in either improved or worsened water quality in the pit lake.

Generally, it can also be said that the hydrologic status of a pit lake can affect whether or not the lake water quality will reach a condition of hydrochemical steady state (i.e., dissolved constituent concentrations in the pit lake are relatively constant over time). For example, a flow-through lake has a better chance of reaching a hydrochemical steady-state condition because chemical inflows and outflows from the lake water column can eventually balance if no other processes are active. It is impossible for hydrologic processes alone to keep a terminal pit lake from reaching chemical equilibrium (i.e., chemicals flow into the pit lake water column but none flow out). These statements are by necessity generalities, and the impacts of hydrologic and other chemical processes are unique to each individual pit lake.

CLIMATE CHANGE AND PIT LAKE HYDROLOGY

Climate is the single most important factor on the hydrologic processes associated with a pit lake. Changes in climate (e.g., temperature, rainfall, wind, precipitation amount and distribution) will affect the individual hydrologic components differently. In general, surface hydrologic processes (e.g., direct precipitation, evaporation, surface water runoff) are impacted immediately upon a change in climate. Groundwater inflows are generally and ultimately generated from precipitation recharge. The groundwater system tends to buffer short-term climatic changes, but long-term climatic changes will be reflected in groundwater inflows over the long term.

The Intergovernmental Panel on Climate Change (IPCC 2007) indicated that there is a strong probability that temperatures will continue to increase into the future given that current conditions affecting atmospheric processes remain constant. This increase in temperatures will affect surface hydrologic processes differently in different parts of the world. Some areas will become wetter while other will become drier. For the western United States, Hoerling and Eischeid (2007) used climate models to predict the Palmer Drought Severity Index in the future. Their work indicates that drought conditions will be worse than at any time in the recent past and drier conditions will prevail. Thus, evaporation rates are anticipated to increase, resulting in an increased loss of surface water from rivers and lakes.

For pit lakes, a dryer climate will most certainly result in lower pit lake elevations. Contrarily, a wetter climate will most certainly result in higher pit lake elevations. However, it is difficult to make broad statements about how climate changes will affect the status of a pit lake (i.e., if it will change from a flow-through to a terminal pit lake or vice versa), because climate changes will affect all the components of the hydrologic system. Because each individual pit lake is different, the resulting water balance from climate change must be evaluated on a case-by-case basis to determine climate change effects on pit lake status. Assessing the effects of potential climate changes was described in Chapter 3.

ARTIFICIAL FLOODING EFFECTS ON GROUNDWATER INFLOW

One method for affecting pit lake water quality is to flood the empty pit at the end of mining with surface water, artificially raising the water elevation in the lake to long-term equilibrium levels over a relatively short time period. The idea being that if there are oxidizing conditions in the pit walls that release acidity and other constituents, a quick submersion reduces the time available (i.e., hydrologic steady-state conditions are reached more quickly) for oxidation to take place and thus reduces the ultimate chemical loading to the pit lake. Also, the water quality of waters used for flooding can be of high quality, resulting in an improvement of initial pit lake water chemistry over that which can result from natural inputs.

If a pit is below the water table and it is artificially inundated, then the hydraulic head in the pit lake will be immediately higher than all the surrounding groundwater elevations (Figure 4.8). Because groundwater *always* flows downgradient (not necessarily along geologic formations or features as is sometimes commonly thought), inflow to the pit will cease and flow will be out of the pit lake into the groundwater system, filling the void space in the surrounding wall rock. This process can push constituents released from the wall rock farther into the groundwater aquifer, effectively increasing the zone of impact surrounding the pit. As such, the potential environmental impacts of artificial filling on regional groundwater resources need to be considered prior to adopting this strategy. The size of the draw-down cone surrounding the pit, the time that the pit has been dewatered, the geology and mineralogy of the pit walls, and the ultimate hydrologic status of the pit lake will dictate the magnitude of impacts to the surrounding groundwater system.

Under artificial filling conditions, groundwater inflows to the pit will cease until the heads in the surrounding aquifer increase to elevations higher than the pit lake surface and the pore spaces surrounding the pit are filled. Groundwater inflows will then increase to the pit lake until a near-steady-state flow rate is achieved.

FUTURE OF PIT LAKES

The hydrologic status of future pit lakes will be dependent on a multitude of factors ranging from climate to regulatory policy changes. Short- and long-term climate changes will inevitably affect the hydrologic status of pit lakes. Continuing research in the prediction and monitoring of climate change will be important to understanding the future status of pit lakes, as climate will

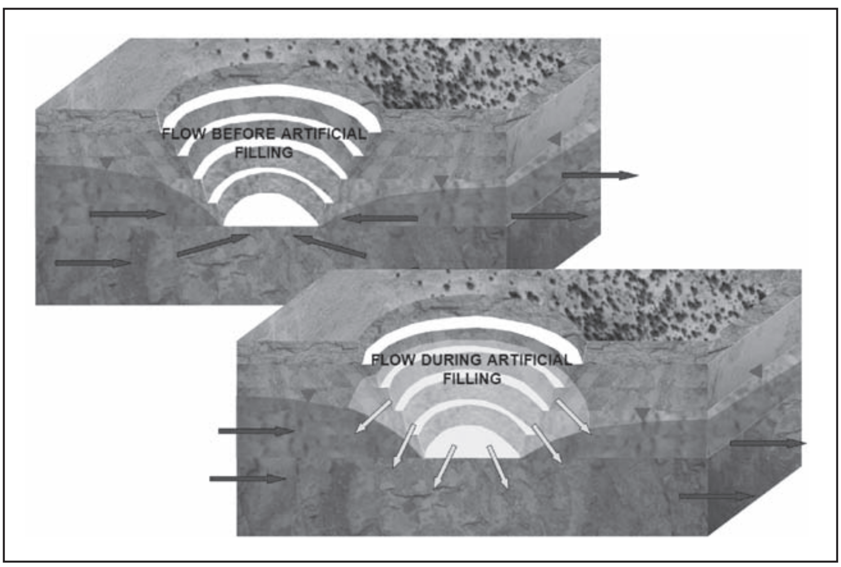

FIGURE 4.8 Artificial filling impacts on groundwater inflow

affect every component of the water balance associated with pit lakes. Future regulatory policies will likely be more protective of the environment. Regardless of climate or policy changes, each pit lake should be evaluated on its own individual merits and as a part of the entire hydrologic system. The advantages and disadvantages of allowing pit lakes to form should be evaluated and objectives developed (e.g., a pit lake may be allowed to form in order to treat water or to be a storage feature in a supply system) prior to pit lake formation. Then, if the lake is allowed to form, a robust monitoring and management program can be implemented to ensure that the pit lake is meeting its objectives.

HYDROLOGIC CHARACTERIZATION DATA GAPS

From a hydrologic standpoint, the inevitable question that is asked regarding an existing pit lake is, "What is the water balance associated with a pit lake?" Ancillary questions are also asked, such as when will it reach a steady water level, will it be terminal or flow-through, and what will its final depth be? By accurately defining the water balance associated with the pit lake, most associated hydrologic questions can be answered.

The most accurate method for determining a water balance would be a direct measure either at the time the question is asked or measurements taken during mining of the pit (e.g., groundwater inflow rates as a function of pit depth). Direct precipitation can be measured with a fair degree of accuracy, and lake surface evaporation can be estimated from pan evaporation measurements taken on-site near the pit lake in question. If complete and accurate dewatering flow rates and groundwater elevation records were taken during mining, groundwater inflow can be robustly estimated. The remaining components (high wall evaporation and groundwater outflow) are typically found through difference. Niccoli et al. (2004) describes a hydrologic balance approach to estimating hydrologic components with a mine pit in Montana. In arid regions, high wall evaporation (i.e., direct evaporation from groundwater exiting the high wall) can be a large component of the water balance, especially if groundwater elevations rebound such that a seepage face develops (Figure 4.9). Continuing research into high wall evaporation would be worthwhile.

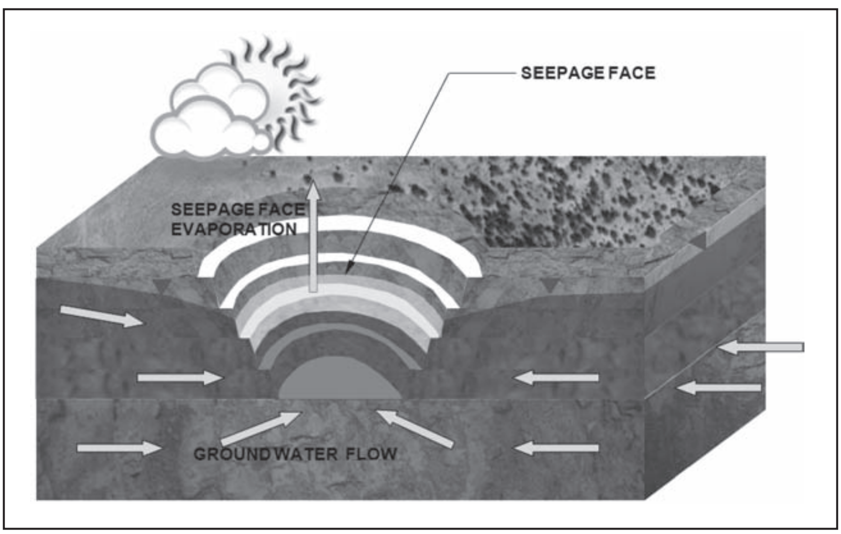

FIGURE 4.9 Seepage face evaporation—groundwater flows to the pit above the pit lake water level

Perhaps direct flux measurements along known seepage faces to verify previous estimates could further understanding of this component. A concerted effort in compiling a database of existing pit lakes and associated hydrologic predictions would be worthwhile. This database could then be used to understand how accurate predictions were, and whether they are close enough to answer the questions asked and to evaluate where weaknesses in predictions occur.

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CHAPTER 5

Stratification and Circulation of Pit Lakes

B. Boehrer and M. Schultze

INTRODUCTION

Like natural lakes, most mine lakes show a vertical stratification of their water masses at least for some extended time periods. Vertical circulation of the water masses is controlled by heat exchange with the atmosphere and gradients of dissolved substances. Even small density differences, in the range of a fraction of a percent, can prevent a lake from overturning and refreshing the deep waters with oxygen. This has decisive impact on the evolution of water quality and, as a consequence, on the community of living organisms in the lake.

In this chapter, the annual cycle of stagnation periods and circulation periods is described. The relevance of the circulation is indicated, and special features of lakes that do not experience a full overturn are introduced. Important factors contributing to density stratification, such as temperature and dissolved substances, are discussed. Also included is the derivation of the appropriate physical quantities for evaluating the stability of a water column—that is, electrical conductance and potential temperature—from in situ measurements. Finally, mitigative measures are referred to, which can be implemented if natural circulation of the lake does not provide enough oxygen to the deep water.

CIRCULATION PATTERNS

In most climate zones on Earth, surface temperatures of lakes show a pronounced temperature cycle over the year (Figure 5.1). This is a consequence of thermal contact with the atmosphere and the seasonal variation of meteorological parameters, such as incoming solar radiation. The temperatures in the deep water follow the surface temperatures only for a time when the lake is homothermal during winter (Figure 5.1). Throughout summer, temperatures differ between surface and deeper layers. Warmer water floats on top of colder, denser water such that the lake remains stratified. As overturning water parcels would require energy, Lake Goitsche, Germany, shown in Figure 5.1, is called stably stratified during summer.

On the contrary during winter, no density differences obstruct the vertical transport of water parcels. Hence, the annual cycle is divided into a stagnation period and a circulation period. During the circulation period, dissolved substances, such as oxygen or nutrients, are distributed over the entire water body. Hence, the circulation pattern is a decisive factor for the evolution of water quality and in consequence for living organisms in the lake. The following classification of lakes according to their circulation patterns is in common use:

- *Holomictic lakes* overturn and homogenize at least once a year. According to the number of circulation periods, monomictic, dimictic (Figure 5.2), and polymictic lakes are distinguished.

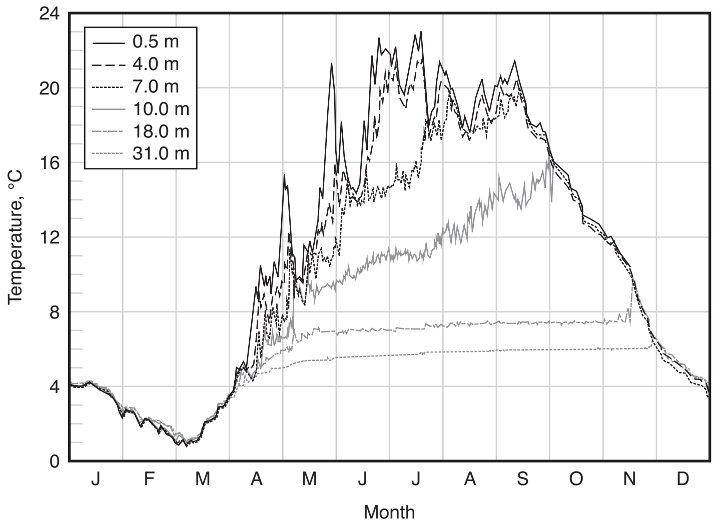


FIGURE 5.1 Temperatures, averaged over 24 hours, at several depths in mine lake Goitsche near Bitterfeld, Germany, during the year 2005

- *Meromictic lakes* are those in which the deep recirculation does not reach the deepest point of the lake. A bottom layer, referred to as the monimolimnion, does not participate in the homogenization and shows pronounced chemical differences, such as anoxia, compared to the mixolimnion (described later).
- *Amictic lakes* do not experience deep recirculation. Usually permanently ice-covered lakes are included in this class. Lakes, however, can also circulate underneath an ice sheet by external forcing.

FORMATION OF LAYERS

Temperature Stratification

While the surface water is exposed to solar radiation and thermal contact with the atmosphere, the deeper layers are sheltered from the major sources of heat. Diffusive heat transport on a molecular level is very slow and takes weeks to transport heat over a vertical distance of 1 m. A much more efficient heat transport is accomplished by turbulent transport. The limited budget of available kinetic energy limits the depth to which a certain amount of heat can be forwarded over the stratification period. In sufficiently deep lakes, the thermal stratification holds until cooler autumn and winter temperatures allow a deeper circulation. The warm surface water layer is called *epilimnion*, whereas the colder water layer beneath, which has not been mixed into the epilimnion over the stratification period, is called *hypolimnion*. A sharp temperature gradient (*thermocline*) forms between both layers (Figure 5.3).

The epilimnion and the atmosphere are in thermal contact and exchange volatile substances with each other. In addition, the epilimnion is circulated episodically by wind events or periods of lower temperatures during the stratification period. In contrast, the hypolimnion is isolated from exchange with the atmosphere during the stratification period. Transport of dissolved matter across the vertical density gradient of the thermocline usually is small.

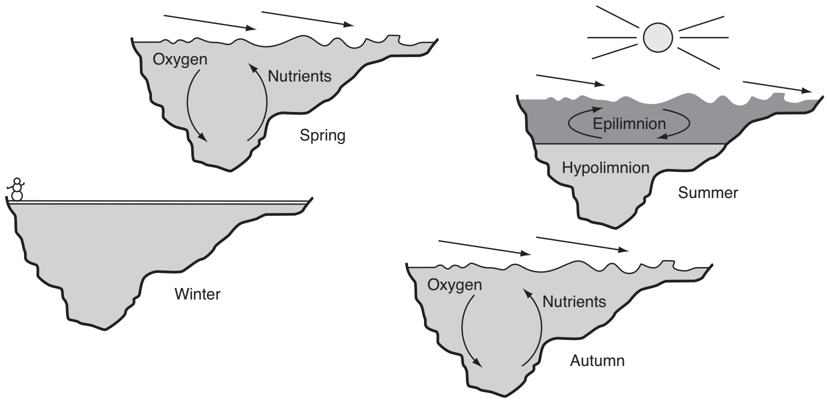


FIGURE 5.2 Annual cycle of a holomictic lake with two circulation periods separated by the presence of an ice cover during winter and thermal stratification during summer (dimictic lake)

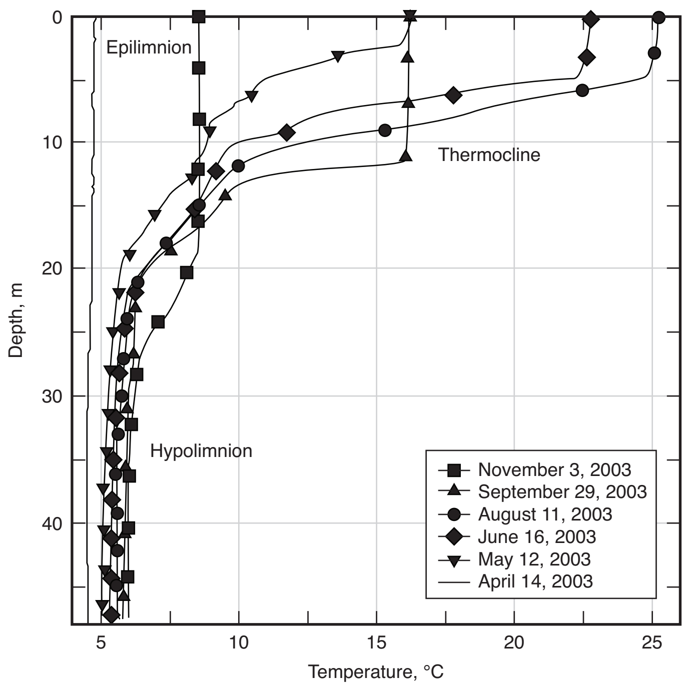


FIGURE 5.3 Temperature profiles of Lake Goitsche, Germany, at station XN5 on six dates in 2003. Symbols are added for every 16th data point to distinguish between acquisition dates.

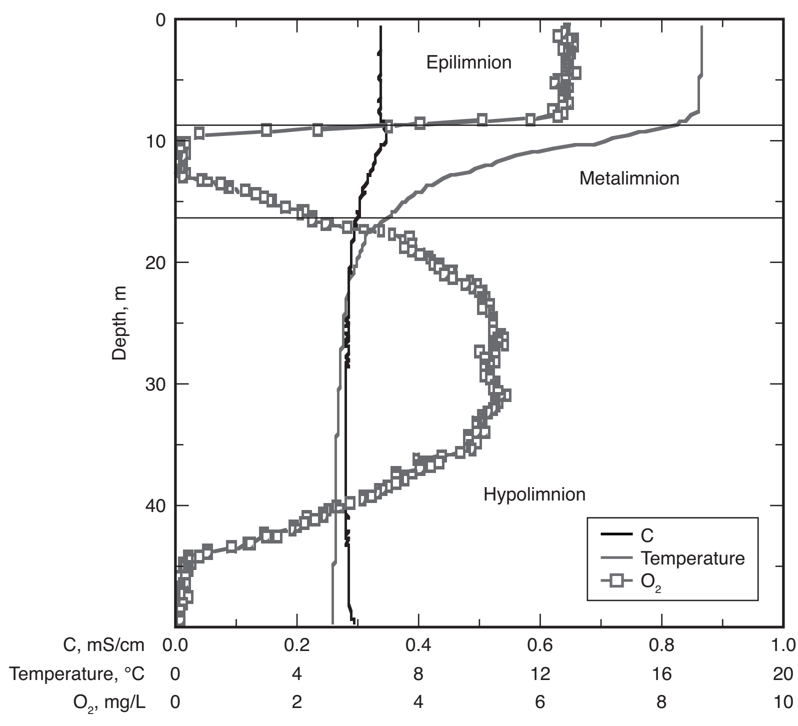
The epilimnion thickness is a crucial factor for living organisms. In general, wind determines the thickness of the epilimnion with few exceptions, such as where light penetrates beyond the mixing depth on account of wind. Empirical studies yield regressions, for example, for the epilimnion thickness (z_{epi}) in units of meters:

$$z_{\text{epi}} = 4.6A^{0.205} \qquad \text{(EQ 5.1)}$$

for natural lakes in temperate regions (Patalas 1984), where higher energy input from stronger winds above larger lakes is included by surface area A in square kilometers. As inferred from Figure 5.3, the thickness of the epilimnion is not constant over the stratification period. In spring, a thin layer is formed, which gradually thickens over the summer on account of the cumulative input of wind energy and heat.

Because of its high gradients, the thermocline forms a special habitat. Inanimate particles can accumulate on their level of neutral buoyancy. Organisms controlling their density can position themselves in the strong-density gradient. In addition, motile organisms dwell in the thermocline, to profit from both the epilimnion and the hypolimnion. As a consequence, a layer of distinctive properties can form. Such a layer is called the *metalimnion*. Especially in nutrient-rich lakes, the decomposition of organic material can cause a depletion of oxygen, resulting in a metalimnetic oxygen minimum (Figure 5.4). On the contrary, if light can penetrate to the thermocline and photosynthesis can overcome the oxygen consumption locally, a metalimnetic oxygen maximum may occur.

In climates where the surface temperature of lakes crosses the temperature of maximum density T_{md} (i.e., 4°C for fresh water) each year, deep mine lakes (> 200 m) will be thermobarically stratified if no other processes have turned them meromictic. Such lakes show the characteristic temperature profile, which follows the T_{md} profile in the vertical over 100 or 150 m, and below a



Source: Adapted from Boehrer and Schultze 2005, with permission.

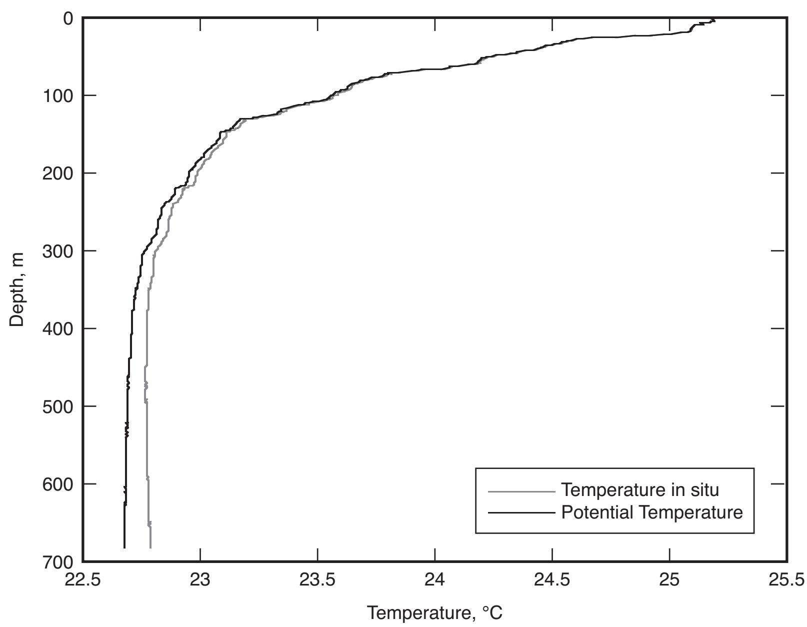
FIGURE 5.4 Profiles of temperature (T), electrical (in situ) conductivity (C), and concentration of dissolved oxygen (O_2) on September 6, 2000, in Arendsee, Germany. The boundaries between layers were drawn along the gradients in the oxygen profiles. Oxygen concentration numerically corrected for response time of 7.5 s of the sensor.

depth of 150 to 200 m temperature gradients disappear. For more details on thermobaric stratification, see Crawford and Collier (1997) and Boehrer and Schultze (2008).

Temperatures recorded in lakes are so-called in situ temperatures. Without any further annotation, temperature data will be understood as such, as it is the physically, chemically, and ecologically relevant parameter. However, for detailed considerations on the stability and vertical temperature gradients, the reference of potential temperature (dT/dz) may be useful. This quantity accounts for the energy required for expansion when a water parcel is transferred to atmospheric pressure adiabatically (*ad*), that is, without exchanging energy with the environment:

$$\left(\frac{dT}{dz}\right)_{ad} = \frac{g\alpha(T + 273.15)}{C_p} \qquad \text{(EQ 5.2)}$$

In Equation 5.2, z represents the vertical coordinate, g = 9.8 m/s^2, the Earth acceleration is the temperature-dependent expansion coefficient, and $C_p \approx 4{,}185$ J/(kg$K$) is the specific heat where K is temperature expressed in Kelvin. In lakes where the deep water is close to the temperatures of maximum density T_{md}, the thermal expansion coefficient is very small, $\alpha \approx 0$. On the contrary, Lake Malawi is located in the tropical zone of Africa with deep water temperature far from 4°C. Consequently, the difference between in situ temperature and potential temperature is noticeable. In this case, potential temperature indicates a stable stratification of the lake by temperature only, whereas a wrong interpretation of the in situ temperature profile would support the opposite conclusion (Figure 5.5).



Source: Data from Vollmer et al. 2002. Reproduction from Boehrer and Schultze 2008, with permission from American Geophysical Union.

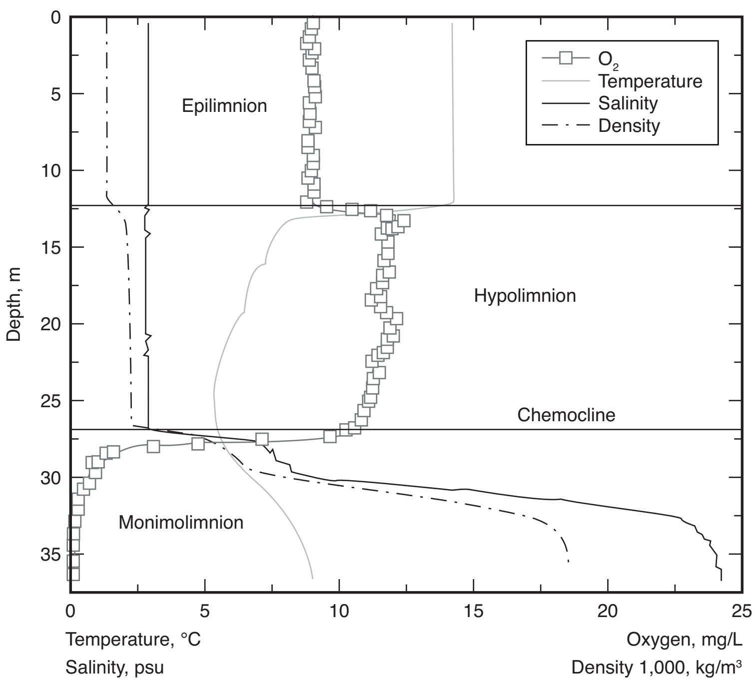
FIGURE 5.5 Profiles of (in situ) temperature T and potential temperature near the deepest location of Lake Malawi, Africa, on September 13, 1997

Salinity Stratification

The ventilated underground in the vicinity of a mine void as well as the overburden, waste rock, unlined process tailings, or ash dumps may release soluble minerals into the aquatic domain. As a consequence, mine waters in general show a higher concentration of total dissolved solids (TDS) than most freshwater systems, and concentrations in the range of 1 g/kg of mine lake water can commonly be encountered. Higher concentrations allow for higher water density gradients. In addition, substances such as iron can dissolve or precipitate because of changing conditions of oxidation–reduction (redox) potential or pH. As a consequence, density gradients can form within the water body.

Dissolved solids modify the properties of lake water. In ocean water, salinity is commonly used to describe the dissolved salt concentration (Figure 5.6). For example, 1 kg of ocean water contains about 35 g of salt; ocean water has a salinity of 35 per mil (parts per thousand). Values given in practical salinity units (psu) are calculated from measurements of electrical conductivity and temperature, and approximate the value per mil for ocean water. The evaluation is straightforward and often used if detailed information on the composition of dissolved substances is missing. Brackish water, that is, water mixed from sea water and fresh water, shows a similar composition of dissolved substances as the ocean, whereas salt composition in lakes can greatly deviate from ocean conditions. In such cases, salinity is better replaced by TDS in the limnic environment.

General approximations for the relation between electrical conductivity, temperature, pressure, and density have been developed for fresh water (Chen and Millero 1986). These approximations are restricted to a maximum TDS concentration of 600 mg/kg lake water. Such



Source: Data from Boehrer and Schultze 2008.

FIGURE 5.6 Profiles of temperature, salinity, dissolved oxygen, and density from Rassnitzer See in the former mining area of Merseburg-Ost (Germany) on October 7, 2003. Oxygen concentration numerically corrected for response time of 7.5 s of the sensor.

approaches assume that substances which contribute considerably to density can be detected by electrical conductivity. This is not valid for many organic substances (e.g., humic substances), weak acids (e.g., silicic acid), and suspended matter (e.g., suspended metal oxyhydroxides).

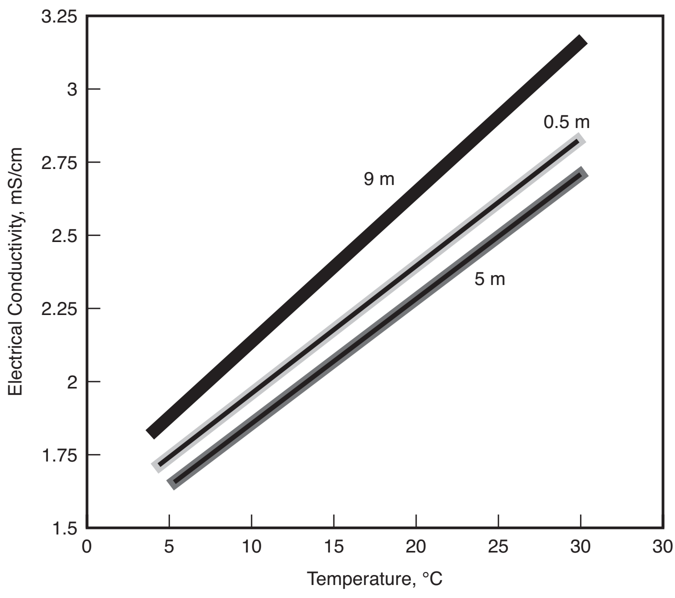
The electrical conductivity of a water parcel is temperature dependent. The term *electrical conductance* $\varkappa_{25}$ is used for the electrical conductivity at a reference temperature of 25°C. Electrical conductance is influenced by chemical composition. Figure 5.7 shows the temperature dependence of electrical conductivity for three different water samples from the same lake.

For detailed considerations on the stratification in pit lakes with highly variable compositions and concentrations, it is strongly recommended to develop site- and time-specific relationships between temperature, electrical conductivity, and density (e.g., Schimmele and Herzsprung 2000). Only rough estimations can be gained from freshwater or ocean water approximations.

The stability of a water column is quantified by the density increase in the vertical dimension according to

$$N \leq = -\frac{g}{\rho}\frac{dp}{dz} \qquad \text{(EQ 5.3)}$$

where g is the acceleration on account of gravity, ρ is density, and z is the vertical coordinate. N is called the stability frequency or Brunt-Väisälä frequency (unit 1/s), which indicates the maximum frequency (ω) for internal waves that can propagate in the respective stratification. The parameter N^2 indicates how much energy is required for the exchange of water parcels in the vertical direction. As vertical excursions in layers of high stability require more energy, it has been found that turbulent transport through such strongly stratified layers is comparably small.



Source: Adapted from Karakas et al. 2003.

FIGURE 5.7 Electrical conductivity of three water samples of Mining Lake 111 versus temperature. Gray symbols represent the measured conductivity; the solid line shows the linear regression.

MEROMIXIS

In some lakes, concentrations of dissolved substances raise the density of the deep waters enough that they do not participate in the total overturn during the annual cycle. Such lakes are termed *meromictic* or permanently or perennially stratified. The chemically different bottom layer is called *monimolimnion*, whereas the water body above is called the *mixolimnion*. In many meromictic lakes, deep circulation erodes the monimolimnion, leaving a sharp gradient at the end of the circulation period. The transition of all water properties between the mixolimnion and the monimolimnion can happen within a few decimeters (see Figure 5.6). This sharp gradient is called *halocline*, *chemocline*, or *pycnocline* to indicate whether the change in properties is due to salinity, chemical gradient, or density gradient, respectively.

Processes Eroding Meromixis

Wallendorfer See in Germany is a salinity-stratified lake in which the vertical position of the halocline was precisely recorded over several years (Figure 5.8). While artificial tracers guaranteed that the groundwater connection only contributed a small part to the vertical shift of the halocline, most of its vertical displacement was attributed to the turbulent erosion during seasonal mixolimnion overturn. On average, the highly saline monimolimnion (80 g/kg) lost 14 cm per year to the less saline (about 5 g/kg) mixolimnion (Figure 5.8, and von Rohden 2002). This small amount is a consequence of the extremely high density difference (about 50 kg/m^3) between monimolimnion and mixolimnion. In lakes with less density difference, faster erosion would be expected.

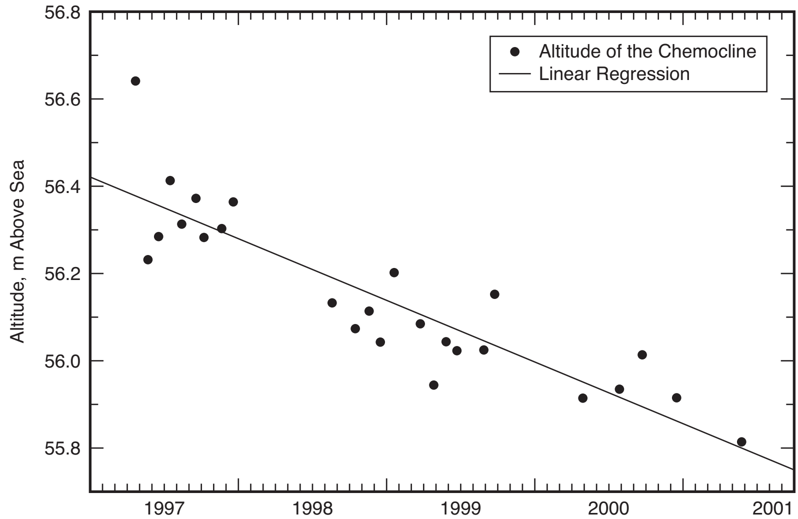
During the stratification period, dissolved substances are transported through the water column by turbulent diffusive processes. Observations of the spreading of an artificial tracer cloud in the monimolimnion of Rassnitzer See (in mining complex Merseburg-Ost, Germany), verified a strong correlation of vertical transport coefficients and stability caused by the density gradient (see Figure 5.9). This indicates that high density gradients (large N^2) limit the vertical transport.

The most prominent source of kinetic energy in a lake is wind, which applies a stress to the water surface. Waves and currents form that cause friction at the lake boundaries and internal current shear. Both effects eventually lead to instabilities and turbulence. Turbulent mixing carries dissolved substances through the water column much faster than molecular diffusion. Consequently, meromixis can preferably be encountered in lakes that are sheltered from the wind, such as lakes with small surface areas, or lakes surrounded by forest or steep side walls of a pit, or lakes that have deep depressions in the lake bed, which are less affected by lakewide currents.

Processes Sustaining Meromixis

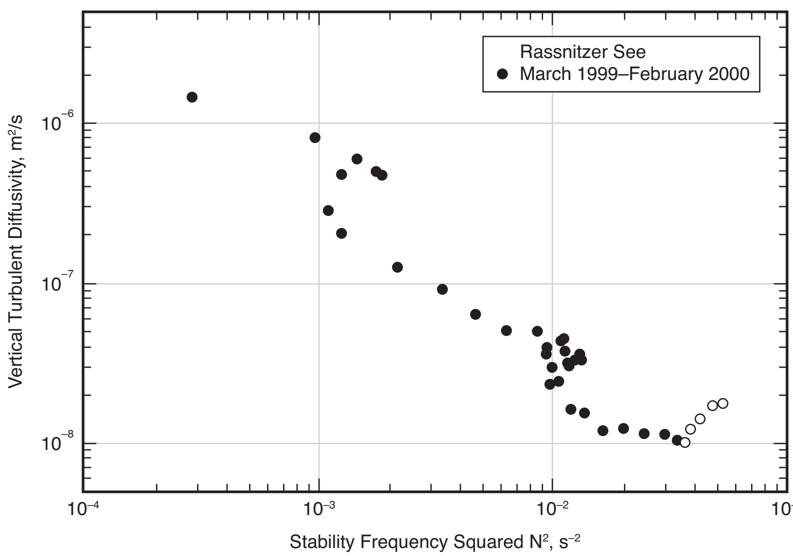
Meromixis will only persist if a process is present that sustains the density gradient. If factors outside of the lake control the gradient, it is called *ectogenic meromixis*; the special case of groundwater is referred to as *crenogenic meromixis*. If the gradient is controlled by factors within the lake, which cause the transport of dissolved substances upgradient, from the mixolimnion to the monimolimnion, then the system is referred to as *endogenic* or *biogenic meromixis*, because biology or microbiology control decisive processes.

If a void is filled from sources of different water quality, density differences may be high enough to form meromixis. After decommissioning, the 330-m-deep Island Copper mine pit (Vancouver Island, British Columbia, Canada) was filled with ocean water and capped with a 7-m-thick freshwater layer. It was designed to be meromictic to dispose of and confine mining influenced water (MIW) to the deep monimolimnion waters (Fisher and Lawrence 2006). Similarly, abovementioned pit lake Rassnitzer See (Figure 5.6) in Germany remains meromictic because of saline



Source: Adapted from von Rohden and Ilmberger 2001.

FIGURE 5.8 Altitude of the chemocline in pit lake Wallendorfer See, Germany, measured at site MA2, over the years 1997 to 2001



Source: von Rohden and Ilmberger 2001, with permission of Birkh�user Verlag.

FIGURE 5.9 Turbulent diffusive transport of an artificial tracer in pit lake Rassnitzer See, Germany, versus density gradient, $N^2 = -g/\rho \; d\rho/dz$. Full circles: measurements inside the monimolimnion; open circles: measurements in the chemocline.

groundwater inflows from a deep aquifer. Inputs of MIW maintain the meromixis in South mine pit in Copper Basin, Tennessee, United States (Wyatt et al. 2006), and in the natural Camp Lake in Manitoba, Canada (Moncur et al. 2006).

Biogenic meromixis originates from organic material decomposing in the deep water of a lake. Organic material, referred to as CH_2O in the following equations, is formed in the epilimnion by photosynthetically active plankton:

$$CO_2 + H_2O \xrightarrow{\text{Light}} CH_2O + O_2 \qquad \text{(EQ 5.4)}$$

or it is carried in by inflowing streams (i.e., allochthonous material). A portion of this material settles. Its decomposition is facilitated by the presence of oxygen or other oxidizing agents (e.g., nitrate, ferric iron, or sulfate) and bacteria:

$$CH_2O + O_2 \rightarrow CO_2 + H_2O \qquad \text{(EQ 5.5)}$$

The end products partly dissolve in the deep layers of the lake and locally raise the water density.

In meromictic mine lakes, decomposition of organic material coupled with iron cycling is often found:

$$CH_2O + 4Fe(OH)_3 + 8H^+ \rightarrow CO_2 + 4Fe^{2+} + 11H_2O \qquad \text{(EQ 5.6)}$$

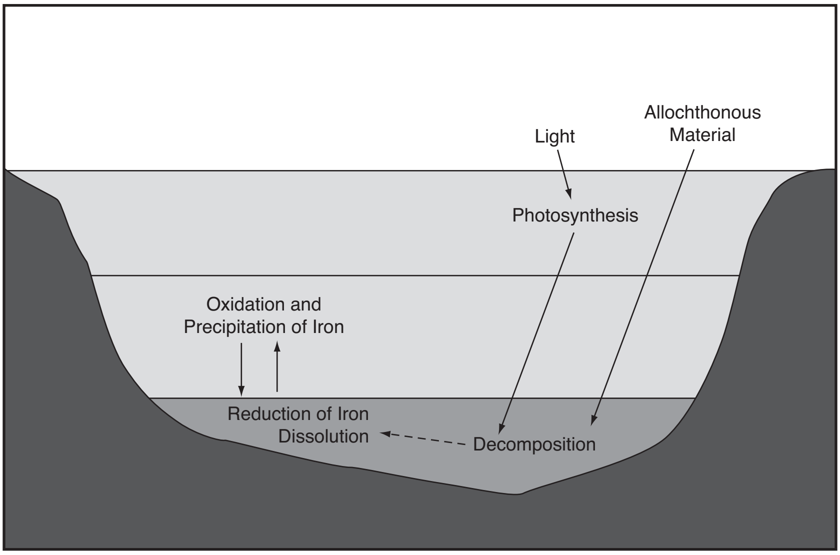
Diffusive and turbulent diffusive transport processes carry ferrous iron and oxygen into the chemocline. Should this ferrous iron get into contact with oxygen, it becomes oxidized and precipitated as ferrihydrite (Figure 5.10):

$$4Fe^{2+} + O_2 + 4H^+ \rightarrow 4Fe^{3+} + 2H_2O \qquad \text{(EQ 5.7)}$$

$$Fe^{3+} + 3H_2O \rightarrow 4Fe(OH)_3 + 3H^+ \qquad \text{(EQ 5.8)}$$

Ferrihydrite settles downward and reenters the monimolimnion. As a consequence, ferric iron is made available for the decomposition of organic material in the monimolimnion again. Hongve (1997) described this cycle in detail for natural meromictic lakes. Both oxidation and reduction are facilitated by the presence of microbial organisms. Reactions 5.7 and 5.8 in sequence release protons, causing the formation of local pH minima in the contact zone between oxic and anoxic layers.

A similar cycle between oxic and anoxic zones can be accomplished by manganese. Also denitrification, sulfate reduction, and methanogenesis are involved in decomposing organic material (Wetzel 2001; Schlesinger 2005). In addition, the calcite cycle has turned natural lakes meromictic (Rodrigo et al. 2001).



Source: Adapted from Boehrer and Schultze 2008.

FIGURE 5.10 Oxidation of ferrous iron to ferric iron and consequent precipitation in oxic layers of the lake. In the monimolimnion, the reduction to ferrous iron is facilitated by using organic material as a reduction agent. Diffusion across the chemocline and mixing of mixolimnetic waters with monimolimnetic waters is very inefficient in terms of transport of iron, as iron is oxidized and precipitated out of the oxic layer.

CONSEQUENCES OF STRATIFICATION FOR WATER QUALITY

Seasonal Stratification

During stratification, the exchange of dissolved substances between hypolimnion and epilimnion is limited. Only turbulent diffusive processes and the gradual thickening of the epilimnion during stratification transport dissolved substances between both layers. On the contrary, sedimentation continuously removes substances from the epilimnion. If substances are redissolved from the sediment or if they are liberated during sedimentation through the hypolimnion as a result of microbial decay of organic matter, they may accumulate in the hypolimnion during the stratification period. The highest concentrations usually occur just above the sediments. The described processes are accompanied by oxygen consumption and may eventually lead to anoxia in the hypolimnion. Anoxia may act as an additional driver for the redissolution of substances from the sediment (e.g., phosphates).

Also, groundwater entering the hypolimnion may contribute to the accumulation of dissolved substances and to oxygen consumption by introducing sulfate and ferrous iron. Chemical transformations during passage of groundwater through the lake sediment must be kept in mind (see Blodau 2004, 2005).

Most commonly, ferrous iron, phosphate, hydrogen sulfide, and reduced species of toxic trace elements (e.g., arsenite, AsO_3^{3-}) and heavy metals can be enriched in deep waters. As hydrogen sulfide and heavy metals precipitate as metal sulfides, they do not coexist at high concentrations at neutral pH. During seasonal circulation, anoxia and accumulated substances are removed by further distribution or oxidation and chemical precipitation.

Meromixis

Meromixis limits the vertical transport through the water column. Dissolved substances can be accumulated over several years or even decades or centuries. In monimolimnia, much higher concentrations are encountered than in hypolimnia. Without considerable import of oxygen, monimolimnia usually show anoxic conditions. Reductive processes are facilitated by parallel oxidation reactions, many involving organic matter. Madison et al. (2003) observed pyrite oxidation in the anoxic monimolimnion of the Berkeley pit lake, Montana (United States). Ferric iron, settling from the oxic mixolimnion and redissolved under the monimolimnetic conditions, oxidized remaining pyrite in the side walls inside the monimolimnion. This process was found to contribute considerably to the acidity budget of the monimolimnion and the whole lake. This process may occur also in other pit lakes. From observations it is known that chemoclines can provide zones of intensive colonization with only few species. Obviously, some plankton species take advantage of such gradients (Tonolla et al. 2004) as observed in Mine Lake 111 and Waldsee near Döbern, Germany (Rücker et al. 1999).

Permanently exposed to the hydrostatic pressure, gases (CO2 [carbon dioxide], H2S [hydrogen sulfide], and others) can accumulate in concentrations far beyond concentrations encountered in mixolimnia. Murphy (1997) made some predictive calculations about whether a limnic eruption (a sudden release of the accumulated gas) can happen in pit lakes. He found that such an event is not very likely but cannot be excluded. None of the theoretical considerations attempting to explain the eruption of Lake Nyos (Africa; Kling et al. 1987) has received wide acceptance. As a consequence, there is no definite answer why some lakes with oversaturated monimolimnia can produce limnic eruptions and others degas quietly. Extreme storms, turbidity currents accompanying flood events, or big landslides at the lakeshore may produce a sudden partial or total overturn of a meromictic pit lake. The consequences for lake water quality may be dramatic and include fast depletion of oxygen or total anoxia, distribution of accumulated toxic substances over the whole water body, or an internal pulse of eutrophication by nutrients formerly accumulated in the monimolimnion.

OPTIONS TO INDUCE OXIC CONDITIONS IN HYPOLIMNIA AND EFFECTS ON WATER QUALITY

Holomixis can be fostered by minimizing wind sheltering (e.g., deforestation of lakeshore), backfilling deep depressions in the lake bed and removing obstacles for lakewide circulation, such as shallow sills. Hypolimnetic water can also be pumped from the lake before dissolved concentrations become a real concern. Artificial destratification/circulation technologies exist that release dissolved gases stored at great depth. The rising bubble plume lifts cool hypolimnetic water to the surface. Such technologies are well established in lake restoration (Cook et al. 2005).

Probably the most undesired consequences of stratification are oxygen depletion and accumulation of nutrients and toxic substances in deep water. Hypolimnetic aeration/oxygenation is a well-established strategy to prevent or remove anoxia. Small bubbles of oxygen are pushed into the deep water, where they dissolve completely. Alternatively, water is withdrawn from the deep water and after oxygen addition returned to the lake. Stratification can be conserved, and, consequently, the undesirable distribution of substances (e.g., nutrients, ferrous iron, and hydrogen sulfide) over the whole lake is avoided (Cook et al. 2005).

Removal of potentially dangerous substances may be performed by chemical precipitation. The required chemicals should be spread only in the part of the water body where dangerous substances are located. Apparatuses primarily developed for hypolimnetic aeration may be used for such purposes (Koschel et al. 2001) as well as submerged spreaders tracked by boats. Ferrous iron can remove hydrogen sulfide, and ferric iron combined with alkaline substances (e.g., lime, soda ash, combustion ashes) may be used to flocculate toxic trace elements or phosphate.

OPEN QUESTIONS

Although stratification and circulation of lakes has been investigated for more than 100 years and models have been applied for about 30 years, the quantitative prediction of chemical stratification remains a challenge (see Tables 5.1 and 5.2). There is no single model currently available that covers the whole complexity of stratification and circulation of pit lakes and the variability and interaction of the relevant factors, such as hydrology of the lake, morphology of the lake basin, exposure to wind, climatic conditions, water quality of inflows, and biogeochemical reactions in the lake water and lake sediment and its coupling back on lake stratification.

Detailed and accurate field studies are needed to further develop and validate models. In particular, the quantitative description of meromixis by models must be improved. Field studies have been conducted at many places, but these data and the results are not easily accessible to the scientific engineering community in all cases. Good documentation of all environmental factors and proper scientific publication would form a valuable contribution for the further promotion of predictive modeling. Beyond this, quantification of chemical transformations and their impact on density stratification still require further research projects to ultimately facilitate predictive modeling for cases of meromixis.

TABLE 5.1 Case studies on stratification and circulation in pit lakes

Pit Lake Name, Location	Reference
Berkeley Lake, Montana, United States	Gammons and Duaime 2006
Blackhawk, Blowout and Duncan, Utah, United States	Castendyk and Jewell 2002
Brenda, British Columbia, Canada	Stevens and Lawrence 1998
East Sullivan Lakes, Quebec, Canada	Tass□ 2003
Goitsche, Germany	Boehrer et al. 2003
Island Copper mine, Vancouver Island, Canada	Fisher and Lawrence 2006
Merseburg-Ost (Rassnitzer See, Wallendorfer See), Germany	B□hrer et al. 1998
Mine Lake 111, Germany	Karakas et al. 2003
Yerington, Nevada, United States	Jewell and Castendyk 2002
Martha mine, New Zealand	Castendyk and Webster-Brown 2007a, 2007b

TABLE 5.2 General papers on stratification and circulation in pit lakes

Title	Reference
Difficulties in predicting the permanent stratification of open cast mining lakes	Boehrer 2000
On the relevance of meromixis in pit lakes	Boehrer and Schultze 2006
Stratification of lakes	Boehrer and Schultze 2008
Pit lakes: their characteristics and the potential for their remediation	Castro and Moore 2000
Physical limnology of existing mine pit lakes	Doyle and Runnels 1997
Mixing mechanisms in lakes	Imboden and W est
The motions of lake water	Imboden 2004
Physical properties of water relevant to limnology and limnetic ecology	Reynolds 2004
The effect of subaqueous disposal of mine tailings in standing waters	Stevens and Lawrence 1997

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CHAPTER 6

Conceptual Models of Pit Lakes

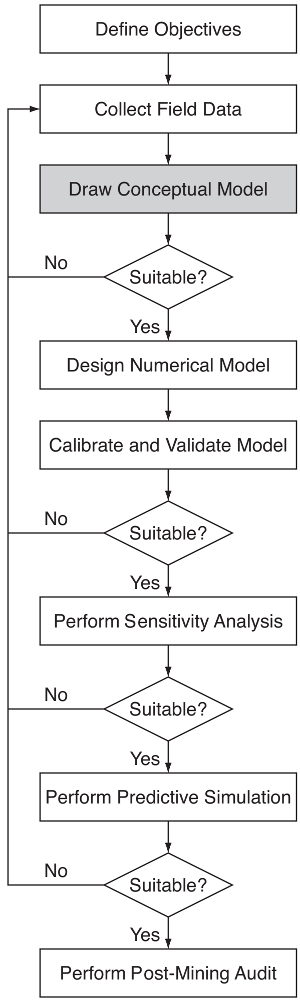
D.N. Castendyk

INTRODUCTION

A conceptual model is a narrative or pictorial representation of a pit lake system that shows the processes which are influencing lake water quality. Typically a pit lake conceptual model consists of a two-dimensional cross section that indicates lake dimensions, wall-rock lithology, and hydrologic, limnologic, geochemical, or biologic processes occurring within the lake. Processes are represented by arrows, text boxes, or equations within the diagram, and the vertical and horizontal position of a given process indicates the spatial location of the process within the pit lake system. Conceptual models may also be used to identify transient processes, meaning those that vary through time.

Conceptual model development is a fundamental step in the development of a pit lake water quality prediction (Figure 6.1). Once the objectives of a prediction have been defined and an initial set of field data has been collected, the modeler will draw a conceptual model that simplifies the field system and organizes the field data to reflect what is known about the system. Simplification is necessary because a complete reconstruction of the field system is not possible; however, oversimplification of the system can generate errors in the water quality prediction. As such, the conceptual model simplifies the field system as much as possible while retaining enough complexity so that it adequately represents pit lake behavior (Anderson and Woessner 1992). The modeler develops a computer-based numerical model from the conceptual model and compares the results of the numerical model to field or laboratory data during model calibration (see Chapters 11 and 15). If a suitable calibration is not achieved, more field data are collected, the conceptual model is refined, and a new numerical model is generated. In this way, the conceptual model becomes a living document that is continually updated as more information is learned about the pit lake system. At the conclusion of the prediction study, the final conceptual model provides an effective tool to visually communicate the logic used to generate the prediction to mine managers, environmental regulators, technical reviewers, and the general public.

Because pit lakes are complex systems affected by multiple processes, it is common practice to use separate conceptual and numerical models to address individual processes occurring within a given pit lake whereby the results generated by one numerical model are used to define the conceptual model for an interdependent process. For example, a hydrologic model is used to determine groundwater inputs to a developing pit lake and these groundwater inputs are specified in a limnologic model that determines the vertical circulation of the lake water column. The resulting circulation data define mixing processes and the vertical distribution of dissolved oxygen in the lake, which are incorporated into the conceptual model for the geochemical prediction. Three papers by Castendyk et al. (2005) and Castendyk and Webster-Brown (2007a, 2007b) demonstrate this sequential approach in the development of a pit lake



Source: Adapted from Anderson and Woessner 1992.

FIGURE 6.1 Flow chart showing the procedure used to model major processes that influence pit lake water quality

water quality prediction for the Martha Au-Ag Mine, New Zealand, beginning with a wall-rock characterization study, followed by a predictive limnologic model, and concluding with a predictive geochemical model. The pit lake water quality pyramid previously shown in Chapter 1 illustrates the hierarchy of processes occurring within pit lakes and provides a logical modeling sequence that ultimately leads to the prediction of pit lake water quality. In some cases, one set of processes interacts with another so that some degree of coupling is required. A common example occurs when limnological processes affect geochemical reactions that in turn influence the water density and limnology. Because coupled models are scarce (see Chapters 11 and 14 for examples), in many cases the limnology and geochemical models are run iteratively.

This chapter reviews generic conceptual models that have been used to define geologic, hydrologic, limnologic, and geochemical aspects of pit lakes. By understanding how these generic conceptual models have been designed, future pit lake modelers will be able to design initial site-specific conceptual models for future open pit mines whereby mine managers and environmental regulators will understand more of the logic used to develop pit lake water quality predictions.

GEOLOGIC CONCEPTUAL MODELS

The minerals present in the walls of an open pit mine influence the pH and composition of groundwater flowing through wall-rock fractures and discharging to the pit lake. Minerals also affect the chemistry of rainwater and snowmelt that flows over the pit walls and reports to the lake surface, called runoff. In addition, submerged minerals can directly react with lake water at the water–rock interface. Two mineral categories exist in the pit walls: primary minerals, which exist prior to open pit mining, such as sulfides and carbonates, and secondary minerals, which are the products of the chemical weathering resulting from exposure of pit walls and waste rock to atmospheric conditions. Secondary minerals include carbonates, oxides, hydroxides, and sulfates of iron, aluminum, manganese, calcium, and other elements, and may occur as a thick, weathered layer or a thin efflorescent crust. Nordstrom and Alpers (1999) provide a detailed discussion of secondary minerals in the mine environment. These definitions are commonly used by mining consultants yet differ from those used by exploration geologists who refer to primary minerals as those formed when the original host rock was emplaced and secondary minerals as those resulting from the hydrothermal alteration of primary minerals.

A geologic conceptual model of a pit lake shows zones of relatively homogeneous wall-rock mineralogy that can be used to infer the characteristics of runoff chemistry produced by different regions of the pit (Figure 6.2). This is typically accomplished by mapping zones of acid-generating rock, acid-neutralizing rock, and acid-neutral rock using results from acid–base accounting (ABA). In ABA, the amount of acid that can be generated by the full oxidation of sulfide minerals (e.g., pyrite, pyrrhotite, marcasite) within a rock sample is compared to the amount of acid that can be neutralized by the complete dissolution of carbonate minerals (e.g., calcite, dolomite) within the same sample (Sobek et al. 1978; White et al. 1999; MEND 2000; AMIRA 2002). From these data, the acid-generating potential of wall-rock zones can be determined and the potential pH of runoff can be included in the geologic conceptual model.

In addition to primary minerals, other factors that affect the pH of runoff include the coating of sulfide or carbonate minerals by secondary minerals, the grain size of the sulfide minerals, the amount of rainfall that occurs at the mine site, and the dissolution of secondary minerals. Fine-grained sulfide minerals found in coal deposits, called framboidal pyrite, may react more readily than coarse-grained, euhedral sulfide minerals found in metal deposits owing to the difference in mineral surface area. However, submicron-sized sulfides are commonly found in hydrothermal gold deposits, and owing to element substitution (especially As) or unstable structure (i.e., marcasite, pyrrhotite), these sulfides may react as quickly as framboidal pyrite. Bowell and Parshley (2005) showed that the dissolution of secondary minerals can significantly influence pit lake chemistry. Unfortunately, methods to identify and quantify secondary minerals in pit wall rocks are not trivial. Electron microprobe or scanning electron microscopy equipment and trained analysts are not widely available, whereas X-ray diffraction methods may be unable to differentiate between the small diffraction peaks generated by both secondary minerals and background scatter (see Castendyk et al. 2005). Rock-forming, primary silicate minerals are generally considered to be inert under near-neutral or mildly acidic conditions, although at very low pH, pyroxenes, amphiboles, plagioclase feldspars, and some clay minerals may contribute to acid neutralization (Langmuir 1997; Paktunc 1999). Once wall-rock mineral zones have been defined, runoff

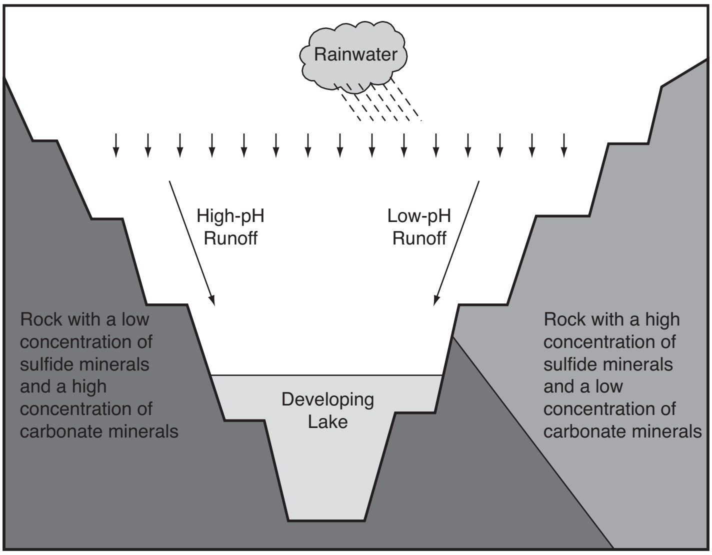


FIGURE 6.2 Geologic conceptual model of a hypothetical open pit mine showing the relationship between wall-rock mineralogy and runoff chemistry

samples should be collected in the field from each zone to verify the relationship between wall-rock mineralogy and runoff chemistry. Humidity cell tests or column tests on rock samples from each zone could also be used for this purpose. See Maest et al. (2005) and Castendyk et al. (2005) for further descriptions of characterization methods used in hard-rock mines.

One common problem in developing a geologic conceptual model is that the definition of homogeneous wall-rock zones can introduce errors by oversimplifying the geology of the deposit. Geologists have long recognized that precious metal deposits are extremely heterogeneous systems as a result of overlapping mineral alteration zones and variable weathering. It is possible that a small area of sulfide-rich wall rock could be responsible for the bulk of acid production within a given mine site. Another problem is that the composition and reactivity of wall-rock minerals will change over the life span of the mine. Fresh minerals are exposed as the size of the pit increases. Sulfide and carbonate minerals are consumed by weathering reactions over time or become coated by secondary minerals, reducing their reactivity. In rare cases, the collapse of a pit wall can expose fresh minerals to water–rock reactions. Consequently, the wall-rock mineralogy observed at the onset of open pit mining will be different from the wall-rock mineralogy at the time of lake filling. Modelers should also be aware that the proportions of rock types exposed above the lake surface will change as the open pit fills with water. For example, 20% of the catchment area of a given open pit may contain acid-generating rocks prior to lake filling, whereas only 5% of the catchment area exposed above the final pit lake surface may contain acid-generating rocks. This configuration is common for ore deposits with an oxidized zone near the surface and a sulfidic zone at depth. Accounting for changing runoff production zones as lake level changes is one of the challenging aspects of modeling pit lakes. It is also difficult to predict what happens to water as it sequentially contacts mineralogically different zones. This often occurs as water runs across a high wall or groundwater flows through backfilled rock.

HYDROLOGIC CONCEPTUAL MODELS

Groundwater models define the exchange of water between the pit lake and the surrounding hydrologic system. Therefore, detailed groundwater models are an integral part of all pit lake limnology and geochemical predictions. Site-specific groundwater models should include the following information: the hydraulic conductivity of the rock layers in contact with the pit lake; the dimensions of both the lake and the groundwater model domain; the boundary conditions of the hydrologic model; whether boundaries are impermeable or constant flux boundaries; and the general direction of the movement of water across each boundary of the model. These points are discussed in further detail in Chapters 4 and 8.

Hydrologic conceptual models show the position of the water table with respect to the walls of the open pit and the lake surface with arrows that indicate the direction of lake inputs and outputs. The water table is an undulating surface above which pore spaces and fractures contain both air and water and below which pore spaces and fractures contain 100% water. Prior to mining, the water table intersects the ore body at some depth below the land surface (Figure 6.3a). In order to excavate the maximum amount of ore available, mining companies use pumping wells to lower the water table below the active mine face, which keeps the open pit dry (Figure 6.3b). Mining companies usually discontinue pumping wells at the conclusion of mining. As a result, the water table rises toward its premining level, and the open pit fills with water to create a pit lake.

The volume of rainfall and surface water that reaches the open pit during lake filling controls the volume of groundwater added to the pit over time. In arid climates where surface water is limited, groundwater is the primary input, whereas direct rainwater and pit wall runoff are secondary inputs and evaporation is the principal output (Figure 6.3c).

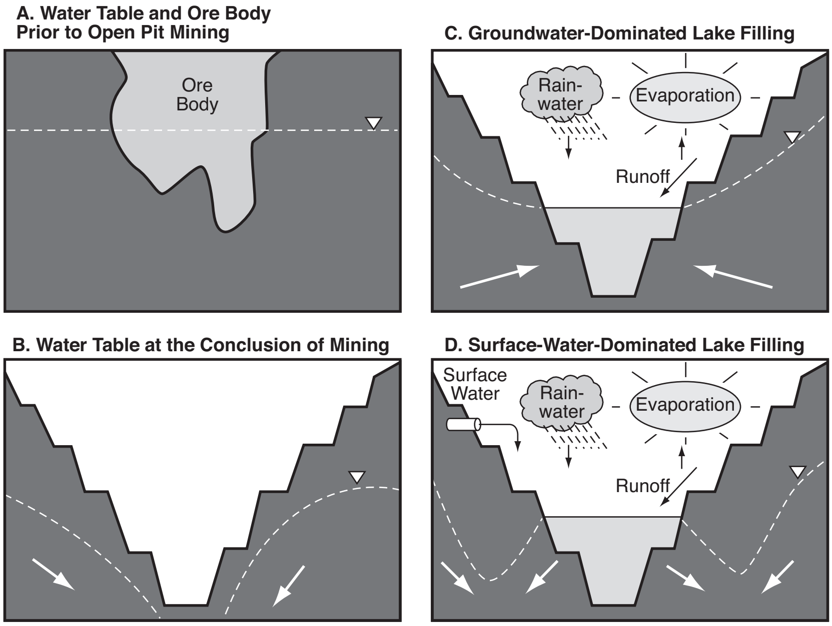
Groundwater moves from regions of high hydraulic head (h_1) to regions of low hydraulic head (h_2) according to Darcy's law:

$$Q = -KA\frac{dh}{dl} \qquad \text{(EQ 6.1)}$$

$$v_x = \frac{-K}{n}\frac{dh}{dl} \qquad \text{(EQ 6.2)}$$

where Q is the flux or discharge of groundwater flow (e.g., m^3/d), K is the hydraulic conductivity of the surrounding rock (m/d), A is the cross-sectional area through which groundwater passes (m^2), dh/dl is the hydraulic gradient ($h_2 - h_1$ divided by dl, the distance between the two observation points), v_x is the average linear velocity of the groundwater (m/d), and n is the porosity (volume of connected voids/volume total).

The hydrology of a pit lake during lake filling conditions depends on the local climate and the closure strategy employed by the mining company. In arid locations, excess surface water is not as likely to be available for lake filling. Consequently, the water table in the wall rocks surrounding the open pit is always higher than the surface elevation of the developing lake, and groundwater continuously flows from the wall rocks to the lake (Figure 6.3d). In humid climates where surface water is not limited, the mining company may choose to divert a portion of the available surface water into the open pit, thereby decreasing the time required to achieve steady-state conditions and potentially improve water quality. Depending on the hydraulic conductivity of the wall rocks and the rate of surface water discharge, more water may be added to the lake by surface discharge than by groundwater discharge. If the elevation of the lake surface rises above the water table in



Source: Adapted from Anderson and Woessner 1992.

FIGURE 6.3 Hydrologic conceptual models showing the location of the water table (dashed line) prior to mining (a), at the conclusion of mining (b), and during various lake-filling conditions (c and d). Arrows indicate groundwater flow directions.

the adjacent wall rocks, the hydraulic gradient will change direction and groundwater will flow away from the lake into the surrounding wall rock (Figure 6.3d). Under these conditions, surface water is the primary lake input and direct rainwater and pit wall runoff are secondary lake inputs, whereas loss of lake water to the groundwater system and evaporation are principle outputs. Some pit lakes may oscillate between conditions shown in Figures 6.3c and 6.3d if the volume of surface water input changes seasonally.

Hydrologic conceptual models of lake-filling conditions often assume that the volume of groundwater input decreases steadily as the lake level rises and the hydraulic gradient between the lake water surface and the surrounding water table decreases. This behavior is analogous to the recovery of hydraulic head in a groundwater well following an aquifer pump test where the rate of recovery is greatest when pumping ceases and decreases exponentially over time (see Fetter 2001). However, unlike the screened interval of a well, as the depth of a pit lake increases, the area that discharges groundwater to the lake increases, owing to the cone-like geometry of the open pit. Consequently, the effect of increasing the discharge area of groundwater flow can be greater than the effect of decreasing the hydraulic gradient, which causes the rate of groundwater discharge to *increase* during the initial stages of lake filling (Chris Gammons, personal communication, 2004). In the later stage of lake filling, after the lake is approximately two-thirds of the steady-state lake elevation, the effect of diminished hydraulic gradient becomes more significant than the effect of discharge area, and groundwater discharge to the lake decreases. These changes in groundwater discharge can be easily demonstrated in a spreadsheet program using Darcy's law (Equation 6.1) and the trigonometric equation for the lateral surface area of a cone (Equation 6.3):

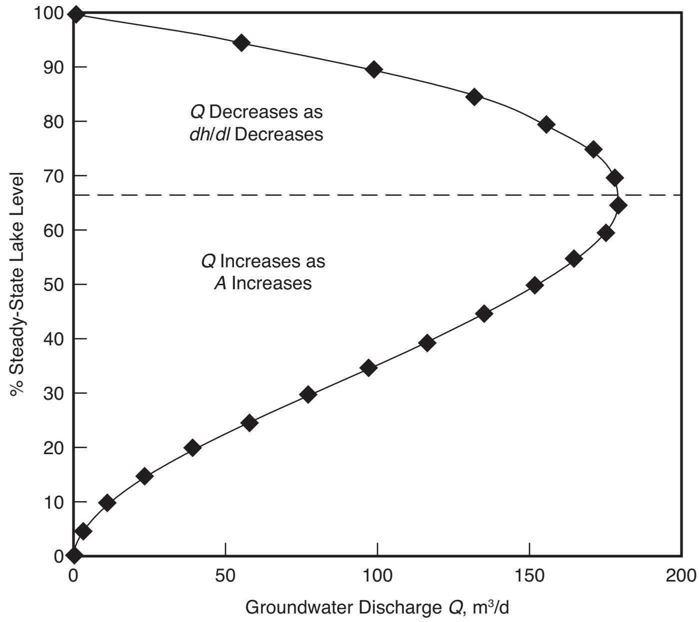


FIGURE 6.4 Hypothetical groundwater discharge (*Q*) to a pit lake during lake-filling conditions as a function of lake elevation. Initially *Q* increases as the discharge area (*A*) increases. After more than 65% of the lake is filled, the effect of the decreased hydraulic gradient (*dh/dl*) becomes more significant than the effect of increased *A*, and *Q* decreases until steady-state conditions are achieved. This plot was generated using Darcy s law (Equation 6.1), the equation for the lateral area of a cone (Equation 6.3), and a hydraulic conductivity, *K*, of 10^{-6} m/s representing fractured igneous rock (Chris Gammons, personal communication, 2004).

$$A = \pi \cdot r \sqrt{r^2 + y^2} \quad \text{(EQ 6.3)}$$

where r equals the radius of the developing lake surface, and y equals the absolute depth of the lake (Figure 6.4).

Groundwater modelers have used two different conceptual models to describe steady-state hydrologic conditions in pit lakes where the elevation of the lake surface and the surrounding water table remain relatively constant over time. In the terminal pit lake model, the elevation of the water table surrounding the lake exceeds the elevation of the entire lake surface, and consequently groundwater flows into the lake but does not flow out (Figures 6.5a and 6.5b). In the flow-through pit lake model, the elevation of the surrounding water table exceeds the elevation of the lake surface for only part of the lake, whereas the elevation of the lake surface exceeds the elevation of the water table for the remaining portion of the lake such that groundwater flows both into and out of the lake (Figures 6.5c and 6.5d).

Three factors that influence the steady-state hydrologic condition are climate, topography, and the hydraulic conductivity of the pit lake wall rocks. Terminal lakes occur in arid regions where the volume of water lost by surface water evaporation is several times greater than the volume of water gained by direct rainfall and pit wall runoff. This occurs where pit lakes are situated in the lowest point of the local or regional topography and/or where the surrounding wall rock has a very low hydraulic conductivity that limits groundwater flow away from the pit. This last factor is probably uncommon because of the abundance of blast-related fractures in pit wall rocks.

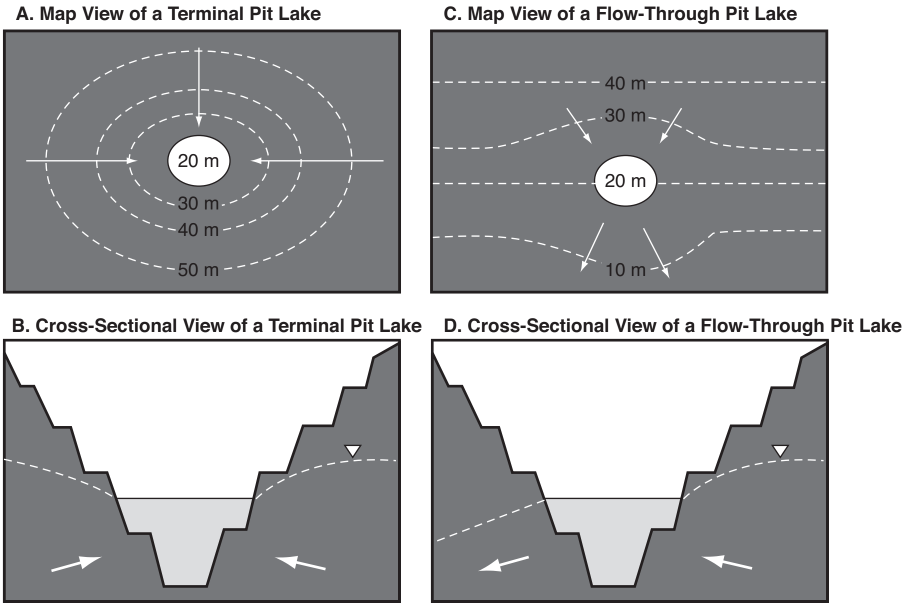


FIGURE 6.5 Conceptual models of terminal and flow-through pit lakes. Map views show water table contours (dashed lines) and flow direction (arrows). Cross-sectional views show the elevation of the water table relative to the lake surface.

Flow-through conditions occur in humid regions where the volume of direct rainfall and pit wall runoff exceed the volume of water lost by evaporation, where a pit lake is located on the top or side of a hill or mountain such that the elevation of lake surface is greater than the elevation of the surrounding topography, and/or where pit wall rocks have a high hydraulic conductivity that can readily transport water away from the lake. As such, terminal pit lakes may be easier to develop numerical hydrologic models for than flow-through pit lakes as one less variable (i.e., groundwater outflow) is required.

These conceptual models become slightly more complicated if the lake has stream inlets or outlets. It is also possible for pit lakes to "leak" groundwater vertically downward, despite having a surface elevation that is lower than the surrounding water table (Fetter 2001). This requires the existence of a high conductivity unit at depth that is connected to a potential discharge point at low elevation. Submerged mine tunnels provide highly conductive groundwater conduits that can pipe groundwater into or away from the lake. Hypothetically, a terminal pit lake may leak groundwater if it is connected to an abandoned underground tunnel that connects to a portal many kilometers away (Chris Gammons, personal communication, 2007). Fetter (2001) provides conceptual models for lakes receiving surface water and discharging surface water as well as leaky lakes.

LIMNOLOGIC CONCEPTUAL MODELS

Limnologic conceptual models are best drawn from the results of numerical limnologic models that predict vertical changes in lake water density over time. Both temperature and chemistry affect the density of water. In temperate climates, lakes become stratified during the summer and winter when lower-density surface water overlies higher-density deep water. These layers mix, or turn over, during the late fall and late spring when the water density is constant with depth. The development of numerical limnologic models is discussed in Chapter 9.

Limnologic conceptual models show whether a lake will completely turn over on an annual basis from the top of the lake to the bottom, or whether a high-density, noncirculating layer, called a monimolimnion, will permanently exist at the bottom of the lake. Lakes that completely turn over on an annual basis are called holomictic (Figure 6.6a), whereas lakes that do not completely turn over are called meromictic (Figure 6.6b). The surface layer of both lakes is called the epilimnion. In the summer, this layer is characterized by warmer temperatures and higher concentrations of dissolved oxygen owing to the dissolution of atmospheric gases across the air–water boundary and photosynthesis by planktonic algae (see Wetzel 2001). Below this layer is the hypolimnion, which exhibits colder summer temperatures than the epilimnion and lower dissolved oxygen concentrations resulting from chemical and biological oxygen demand (COD and BOD, respectively). Examples of COD include oxidation of sulfide minerals exposed on submerged mine walls or oxidation of dissolved Fe^{2+} or Mn^{2+} in the pit lake waters. A common example of BOD includes oxidation of organic carbon by bacteria. Deeper yet, the monimolimnion is typically anoxic and exhibits a higher concentration of total dissolved solids as a consequence of the dissolution of mineral precipitates under low oxidation–reduction (redox) conditions. This layer may also contain sulfate-reducing bacteria that precipitate sulfide minerals and lower concentrations of trace metals. The epilimnion and hypolimnion of both holomictic and meromictic lakes will turn over seasonally.

In addition to showing whether a lake will be holomictic or meromictic, limnologic conceptual models indicate the depth of turnover, the frequency of turnover, and the volumes of water contained in each lake layer. These data are important inputs for numerical geochemical models, which will determine lake chemistry following turnover as a product of mixing reactions between two layers with different volumes and different chemistries. As a result of turnover, dissolved oxygen is supplied to the hypolimnion from the epilimnion, which prevents the hypolimnion from becoming completely anoxic. In these ways, limnologic processes directly influence lake water chemistry.

GEOCHEMICAL CONCEPTUAL MODELS

Geochemical conceptual models show all of the geochemical processes that a modeler will include in a numerical geochemical model of pit lake water chemistry (Figure 6.7). There are many important geochemical processes occurring within pit lakes, and consequently, geochemical conceptual models can be very busy diagrams. The conceptual model identifies processes that occur in specific lake layers to help modelers keep track of vertical differences in lake chemistry.

Basic geochemical processes in pit lakes include

- Chemical mass transport into and out of the lake;
- Dilution/evapoconcentration effects caused by climate conditions;
- Periodic mixing of the epilimnion and the hypolimnion;
- Exchange of carbon dioxide (CO_2) and O_2 gases between the atmosphere and the epilimnion;
- Dissolution and precipitation of minerals from lake water, notably the dissolution of carbonate minerals in wall rock and the precipitation of iron hydroxide;
- Aqueous speciation reactions;
- Redox reactions, notably the oxidation of sulfide minerals in wall rock;

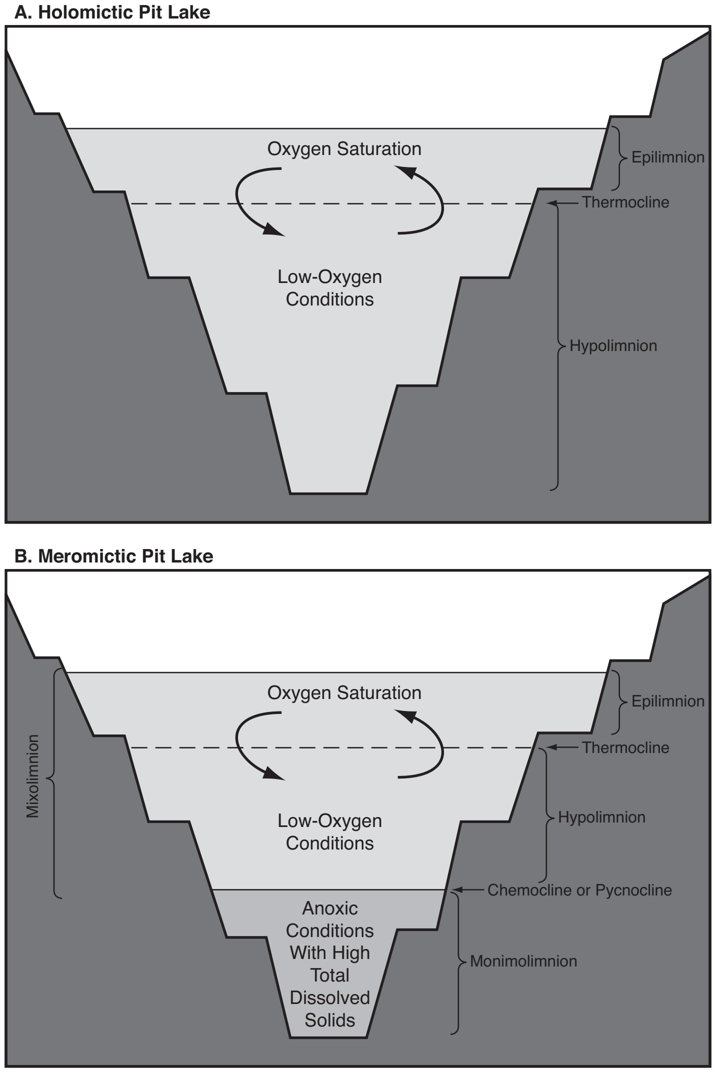


FIGURE 6.6 Limnologic conceptual models of holomictic (a) and meromictic (b) pit lakes. Arrows indicate lake layers that mix annually.

- Microbial processes affecting redox conditions, dissolved concentrations of O_2 and CO_2, the removal of organic carbon, and the precipitation of sulfide minerals at depth; and
- Adsorption–desorption reaction between dissolved solutes and suspended particles, secondary minerals, and/or weathered materials on submerged mine walls.

Several geochemical programs allow modelers to include each of these processes in a geochemical prediction of a pit lake, such as the PHREEQC computer program (Parkhurst and Appelo 1999); see Castendyk and Webster-Brown (2007b) for an example. Table 6.1 describes the causes of various processes, the lake layer where each process is most likely to occur, the effect of each process, and useful notes for modelers. For more information on these processes, see

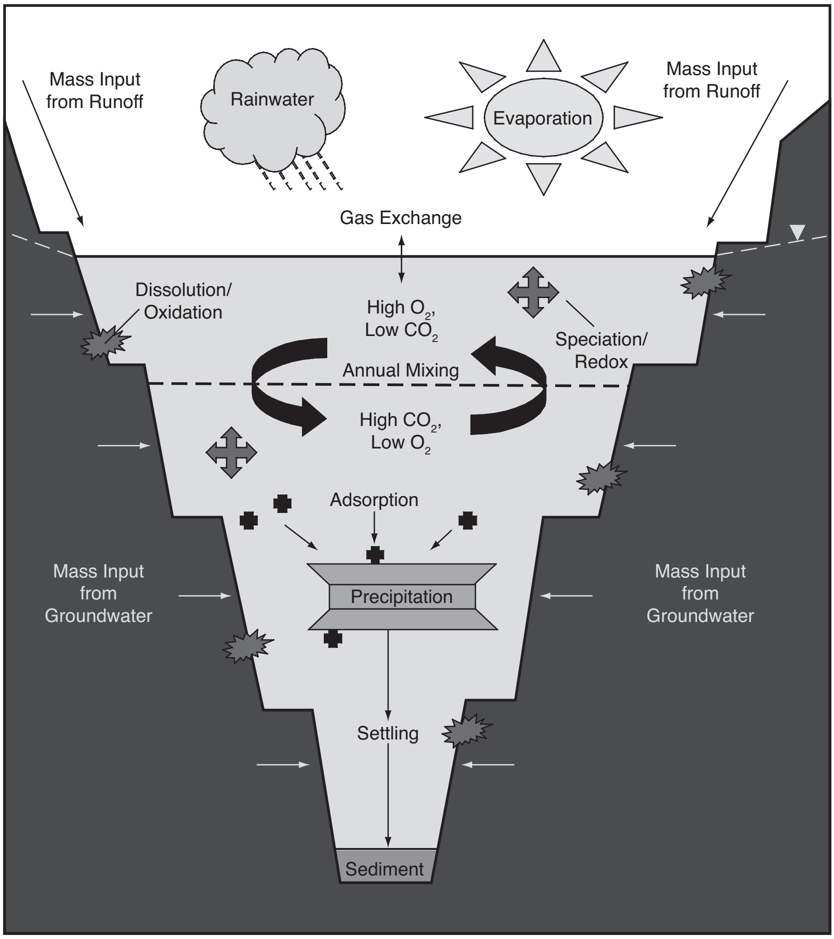


FIGURE 6.7 Geochemical conceptual model of a terminal holomictic pit lake. Reactions included in the diagram are described in Table 6.1.

Drever (1997) or Faure (1998) for a comprehensive overview of geochemistry, and Nordstrom and Alpers (1999) for a discussion of specific reactions occurring at mine sites. Chapters 13, 17, 18, and 19 provide discussions of the effects of microbial processes in pit lake geochemistry.

At the present time, several processes known to affect pit lake chemistry are typically not included in conceptual or numerical models. These include the following:

- Photochemical reactions in the epilimnion, such as the photoreduction of Fe^{3+} to Fe^{2+};
- Biological reactions, such as algal photosynthesis (i.e., CO_2 consumption and O_2 production) and respiration by microbial decomposers (i.e., O_2 consumption and CO_2 production).
- Uptake or cell wall sorption of metals by microorganisms in the water column;
- Trace metal adsorption onto organic surfaces, Al and Mn hydroxide surfaces, and clay surfaces;

TABLE 6.1 Basic geochemical processes to consider in geochemical conceptual models of pit lakes

Process Type	Cause	Affected Layer(s)	Effect	Modeling Notes
Mass transport	Pit wall runoff from rain or snowmelt flows to lake.	Epilimnion	Transports acidity, alkalinity, and dissolved metals to or from the lake. Inputs may also cause dilution of dissolved metals.	Groundwater and surface water discharge may be more complicated to model than lake inputs.
	Groundwater flow to lake.	Whole lake		
	Surface water flow to lake.	Epilimnion		
	Groundwater flow from lake.	Whole lake		
	Surface water flow from lake.	Epilimnion		
Dilution/ concentration	Rain or snow landing on lake.	Epilimnion	Dilution of most dissolved constituents. Possible addition of acid and other solutes (such as nitrate) from acid rain.	The pH (≤5.6) and composition of rainwater are significant, especially near coastal areas (sea spray) or near sites of industrial emissions of acidic constituents.
	Evaporation from lake surface.	Epilimnion	Increase in concentration of all dissolved constituents. Precipitation of secondary minerals that are near saturation (such as gypsum).	It may be appropriate to model only rain input or evaporation based on the difference between the time-weighted average of these parameters.
	Salt exclusion during freezing.	Epilimnion	Increase in salinity immediately below ice layer. May induce mixing of epilimnion and hypolimnion.	May be difficult to model.
Mixing	Annual or biannual turnover of epilimnion and hypolimnion layers.	Epilimnion and hypolimnion	Homogenizes temperature and chemical composition of lake, including dissolved O_2, dissolved metals, and nutrients.	Accurate prediction of the volumes of layers involved in turnover and the frequency of turnover events are required in any limnologic model.
	Wind and waves mix runoff, rainwater, and surface water inputs.	Epilimnion	Epilimnion may be homogeneous throughout the year.	Some high-density surface water inputs may not mix with epilimnion.
	O_2 dissolves into lake surface.	Epilimnion	O_2 available for subaqueous sulfide oxidation.	Production of CO_2 by biologic reactions are also important.
	CO_2 exsolves from lake surface.	Epilimnion	Exsolution of CO_2 increases lake pH by the following reaction: $H^+ + HCO_3^- = H_2CO_3 = CO_2 + H_2O$	Production and consumption of CO_2 by biologic reactions are also important. Field tests to remove CO_2 from acid mine drainage have been unsuccessful.

(Table continued next page)

TABLE 6.1 Basic geochemical processes to consider in geochemical conceptual models of pit lakes (continued)

Process Type	Cause	Affected Layer(s)	Effect	Modeling Notes
Speciation/redox	Dissolved constituents react with one another to form new species at a lower energy state.	Whole lake	Lake chemistry evolves toward thermodynamic equilibrium.	Geochemical programs assume thermodynamic equilibrium is immediately achieved. Not valid for some reactions, notably redox.
Precipitation of minerals	Oversaturated dissolved mineral phases, notably hydrous ferric oxides (HFOs), jarosites, Al and Mn hydroxides, sulfate minerals, and clay minerals.	Whole lake	Precipitation of HFOs and other hydroxides decreases pH, decreases dissolved concentrations of Fe, Al, and Mn, and provides surfaces for metal adsorption.	Research has found discrepancies between observed and predicted concentrations of HFOs and other hydroxides, particularly under low-pH (<4.0) conditions. Models tend to indicate more precipitation than observed. Metal solubility is often controlled by poorly crystalline metastable compounds for which accurate thermodynamic data are lacking.
Dissolution/ oxidation of minerals	Carbonate minerals present in wall rocks.	Whole lake	Carbonate dissolution raises alkalinity concentrations and raises pH. A notable exception is siderite ($FeCO_3$), which is acid-neutral if Fe^{2+} oxidizes. These reactions also affect runoff chemistry.	A large supply of carbonate host rock can perpetually buffer acidity; however, smaller amounts can be consumed over time or isolated from reactions by coatings of secondary minerals.
	Sulfide minerals present in wall rocks with high concentrations of dissolved O_2 or Fe^{3+} in lake water.	Whole lake	Sulfide oxidation lowers lake pH and increases concentrations of sulfate and trace metals. These reactions also affect runoff chemistry.	The surface area of sulfide minerals exposed to lake water controls this reaction. Iron-oxidizing bacteria convert Fe^{2+} to Fe^{3+} below pH 4.0, promoting sulfide oxidation by Fe^{3+}.
	Soluble secondary minerals present on weathered wall rock, such as efflorescent metal-sulfate or acid-sulfate salts.	Whole lake	Dissolution of secondary minerals can release stored acidity and trace metals to lake water. These reactions also affect runoff chemistry.	Quantification of secondary minerals on wall rocks is difficult.
Metal adsorption onto surfaces	Presence of hydroxide precipitates, clay minerals, or organic material with reactive surfaces in solution.	Whole lake	Surface adsorption decreases concentrations of metals and metalloids (e.g., As) in solution. The adsorption of anions is favored at low pH and cations at higher pH.	Most research has been conducted on HFO surfaces, but other surfaces are important, especially organic surfaces. Reactions are strongly pH dependent.

- The activity of iron- and sulfur-oxidizing bacteria, such as *Acidithiobacillus ferrooxidans*, and their influence on pH and solute concentrations;
- Monimolimnion and sediment processes including sulfate-reduction, sulfide precipitation, hydrogen sulfide (H_2S) formation, and the reductive dissolution of Fe and Mn coupled with the release of adsorbed metals;
- Eddy diffusion that transports dissolved metals from the monimolimnion to the hypolimnion;
- Vertical mixing caused by the injection of acid mine drainage, tailings, or lime-treated sludge into the hypolimnion or monimolimnion;
- Partial acid neutralization caused by the dissolution of wall-rock silicate minerals, such as pyroxenes, amphiboles, plagioclase feldspars, chlorite, illite, and volcanic glass under low-pH conditions;
- The decrease in the reactivity of wall rock over time; and
- Variations in groundwater input chemistry and runoff input chemistry over time.

Modelers may exclude these processes to simplify the field situation or because other reactions are assumed to be more significant. Technical limitations also prevent these processes from being modeled at this time, such as insufficient field data to incorporate these processes into a geochemical program or geochemical programs that do not allow for these processes to be modeled. Moreover, given the level of uncertainty associated with many of these processes, the results of a comprehensive geochemical model may have large uncertainty. Future pit lake prediction efforts should endeavor to incorporate these processes into conceptual and numerical models, to validate the accuracy of these predictions, and to rank the sensitivity of pit lake water chemistry to these processes to determine which are most significant.

The implications of the validity of the geochemical prediction are greatest for epilimnion and hypolimnion water quality because these layers are in direct contact with terrestrial ecosystems. Larger uncertainty in the geochemistry of the monimolimnion may be acceptable provided that limnologic models show very little risk of complete lake turnover. However, modelers and mine managers should appreciate the consequences of this assumption because unexpected complete turnover involving the monimolimnion can have serious environmental impacts. Monimolimnion turnover could be caused by a massive pit wall failure or an extreme weather event.

SUMMARY

An accurate conceptual model is an essential precursor to the development of an accurate numerical prediction of pit lake water quality. At the very least, conceptual models should present the following information:

- The catchment area of the open pit, the maximum depth of the lake, the surface area of the lake, and the thickness and volume of individual lake layers.
- Wall-rock zones with relatively homogeneous mineralogy and the acid-producing/acid-neutralizing potential of each zone.
- Annual rainfall compared to annual evaporation. "Arid" locations exhibit annual evaporation that is greater than annual rain- and snowfall, whereas "humid" locations exhibit annual rain- and snowfall that is greater than annual evaporation.
- The proposed lake-filling plan characterized as "groundwater-dominated," "surface-water-dominated," or "variable flooding."

- Structures in the wall rock that provide pathways for groundwater flow such as major faults, blast-related fracture zones, and mine tunnels.
- Characterization of the steady-state hydrology as "terminal" or "flow-through."
- Characterization of the physical limnology as "holomictic" or "meromictic."
- Identification of the basic geochemical processes considered in the geochemical model.

Clear definition of each of these terms will aid the development of an accurate numerical model. Given the number of processes influencing lakes, it is helpful to present the conceptual model as a series of process-specific diagrams rather than one diagram. Whereas conceptual models simplify the field situation to ease the development of numerical models and not every conceivable process influencing lake chemistry will be included, errors in water chemistry prediction are generated by conceptual models that are too simple. For this reason, the accuracy of a water quality prediction for a pit lake may be contingent on the effort spent determining the most significant geochemical processes within a given lake.

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CHAPTER 7

Sampling and Monitoring of Pit Lakes

C. Gammons

INTRODUCTION

This chapter provides advice on the development of water monitoring projects used to explore trends in the water chemistry of developing or existing pit lakes over time. Whereas the collection of rock samples is equally important, particularly for the purposes of site characterization prior to open pit mining, this topic will be fully addressed in the forthcoming *Sampling and Modeling for the Mine Life Cycle* volume of the Acid Drainage Technology Initiative–Metal Mining Sector series. Readers are referred to books by Maest et al. (2005) and Castendyk et al. (2005) for discussions on rock sampling methods pertaining to pit lakes.

BEFORE SAMPLING

The first and most important step in any pit lake monitoring project will be development of a detailed sampling plan. Because each pit lake will have its own peculiar characteristics, it is very difficult to make recommendations that will be universally valid for all field sites. Reading literature on other pit lakes is recommended, as is seeking professional advice and oversight, either from academia, government agencies, or private consultant companies.

A sampling plan should be written in as much detail as possible and reviewed by the project supervisor *before* field samples are collected. A sampling plan should include the following components:

- Summary of the overall objectives of the project
- List of key project personnel and their phone numbers
- Timeline for when and where each set of samples will be collected
- Checklist of equipment and supplies to bring to the field for each sampling event
- List of sample types to be collected at each location in the lake
- List of quality assurance (QA) samples to be collected during each sampling event (such as field duplicate samples, field blanks, and split samples to be sent to different laboratories)
- Standard operating procedures for each field method (e.g., calibration of equipment, bottle rinsing and filtration, acid preservation, decontamination)
- Instructions for labeling sample bottles
- List of chemical parameters to be quantified for each sample type
- List of the laboratory method to be used for each parameter, the identity of the lab that will perform the analyses, and instrument detection limits
- Information on sample preservation and holding time requirements

- Spare copies of chain-of-custody forms
- Site-specific safety plan

The objectives of the project will dictate the level of rigor that is required in terms of quality assurance/quality control (QA/QC). If the work is regulatory in nature, then the level of QA/QC is likely to be substantial, the project sampling plan will need to be reviewed and endorsed by appropriate agency personnel before any field work can be started, and the analytical work will likely require the use of a certified laboratory. On the other hand, field projects for scientific research often do not require regulatory oversight but should nonetheless follow rigorous procedures with respect to planning and implementation of the work.

FIELD SAFETY

A safety plan should be an important part of any project plan and will often exist as a separate document. Sampling of pit lakes will usually take place from a boat, and so there are the usual safety hazards associated with open water and operation of the vessel. There may also be site-specific hazards for a particular pit lake. For example, some pit lakes may have unstable mine walls, increasing the possibility of a landslide while the field personnel are on the water. The resultant *seche* or "tidal" wave could easily capsize a small boat. Also, some pit lakes may release dangerous quantities of dissolved gas in the event of a partial or complete lake turnover. Gases of concern include carbon dioxide (CO_2) and hydrogen sulfide (H_2S). Both of these gases are heavier than ambient air, so release of large quantities of either gas could lead to a lethal layer of CO_2-rich (or H_2S-rich) and O_2-poor air just above the lake surface. If there is a risk of this happening, it may be necessary to carry an O_2 meter at all times and, perhaps, a self-contained breathing apparatus for each field sampler. The potentially tragic consequences related to asphyxiation by CO_2 gas when working around mine waste was brought to the industry's attention by the recent deaths of four individuals at the Sullivan mine, British Columbia, Canada, in May 2006 (BC Government Media Room 2006). Finally, it is recommended that any sampling activity on a pit lake follow a "buddy system," whereby there is a clear line of communication (cell phone, radio phone) between personnel on the boat with personnel on land in the event of an engine failure or other emergency situation. Having a backup boat or "rescue skiff" gassed up and ready to go is a good precaution.

HOW TO SAMPLE

Retrieval of water samples from mining pit lakes is theoretically no different than getting water samples from any lake. Samples from depths of less than 30 or 40 m can be obtained using flexible plastic tubing and a battery-operated peristaltic pump. For deeper water samples, some sort of point sampler is typically used such as a van Dorn or Kemmerer sampler, which is a steel or plastic cylinder with tight-fitting end caps. The caps are spring-loaded in an open position while the sampler is lowered through the lake. When the desired depth is reached, a small weight or "messenger" is sent down the cord to trip the spring, which causes the lids to clamp shut. The contained water sample is then hauled to the surface manually or with a winch. A simpler option is to use a bailer, which is an open-topped cylindrical sample chamber with a check valve in the bottom that is open while the bailer is lowered through the water column but closed while it is raised. Of the two designs, the point sampler is better in terms of minimizing the possibility of mixing of waters at different depths. A large variety of point samplers and bailers are available from different manufacturers. Attention should be paid to the materials used to construct the sampler and its wetted parts, and whether these could be potential sources of contamination. The integrity of the point sampler with respect to leakage can be checked by preloading with a solution that has a specific conductivity (SC) very different from that of the pit lake. After lowering the sealed sampler to the bottom of the lake and hauling it back up, the SC of the contained sample can be checked to see if leakage or mixing has occurred.

Although it is common to measure the temperature and pH of water samples bailed or pumped to the surface, it is better to obtain these measurements in situ using a submersible multiprobe (see "Field Measurements" section later in this chapter). Temperature will change quickly as water samples are raised to the surface, and the drop in pressure may cause exsolution of dissolved gases (such as CO_2) with resultant changes in pH.

WHEN TO SAMPLE

The frequency at which a given pit lake is sampled will depend to large degree on availability of funds and regulatory requirements. At a minimum, a vertical profile of water samples for complete chemical analysis should be collected once a year. More frequent sampling is recommended if seasonal turnover is known or suspected to occur, or if a change occurs in the volume or composition of influent waters. If possible, it is recommended to collect a CTD (conductivity-temperature-depth) profile once a month (see the "Conductivity, Temperature, and Depth" section later in this chapter). This information is relatively inexpensive to collect and is very useful to track seasonal changes in the depth of the thermocline or chemocline, and whether pit lake turnover has occurred or is likely to occur in the future. Another idea is to install a vertical array of sensors to continuously record temperature information over an entire calendar year or ice-free season. Several brands of relatively inexpensive temperature sensors are now commercially available that have built-in data loggers and can be deployed for long periods of time at depths of several hundred meters or more.

Many pit lakes at high latitudes will freeze solid during the winter, raising the possibility of collecting samples through the ice. There are a number of reasons why this may be worthwhile. The presence of ice at the surface of a lake will limit transfer of gas across the air–water interface, which will impact variables such as dissolved oxygen or CO_2 of the epilimnion. As well, exclusion of dissolved salt during crystallization of ice may result in a layer of cold, high-salinity water below the ice layer. The so-formed saline water may then sink through the water column, possibly causing instability of chemical or thermal boundary layers in the lake.

WHERE TO SAMPLE

The question of where to collect pit lake samples refers not only to depth but also location on the lake surface. In terms of depth, it is hard to generalize how many samples are needed to define a vertical profile. If possible, it is recommended to collect a CTD profile within a few days of retrieving any water samples. The results of the CTD profile will show the elevation of any thermal or chemical boundary layers. The number of water samples to collect will then depend on the complexity of the CTD profile, the total depth of the lake, and other factors such as availability of funds and how and by whom the data will be used. For example, five samples may be enough to characterize vertical gradients for monitoring purposes but will likely be insufficient for a scientist conducting detailed research on biogeochemical processes.

In terms of location, it is best to collect samples above the deepest part of the lake. Collection of a depth profile anywhere else may miss a deep layer of high-salinity water. For optimal comparability, it is a good idea to collect samples at the same location from year to year, using geographic coordinates or a buoy as a guide. However, maintaining a precise location in a boat is difficult if there is wind and no possibility of moorage. This may or may not pose a problem, depending on the size of the lake. Providing targets on the walls of the pit can help to keep the boat in position during sampling. The next obvious question might be: "How many depth profiles per sampling event are needed to characterize the chemistry of the entire pit lake?" In a study of the limnology of the Berkeley pit lake (Jonas 2000), nearly identical water chemistry and temperature results were obtained at three widely spaced locations in the lake, despite the facts that the total depth of the water column varied between sites, and sharp gradients in temperature and Fe concentration occurred as a function of depth at each site. These data suggest that the pit lake was well mixed at each elevation in a horizontal direction while retaining sharp chemical and thermal boundaries in cross section. In the absence of any major surface water inputs (such as lime treatment sludge, diverted streams, or tailings), it is probably a safe assumption that horizontal thermal or chemical gradients will be less significant than vertical gradients in most pit lakes. In such cases, a single depth profile may be representative of the entire lake. However, this assumption should be tested by conducting at least one sampling event in which multiple vertical profiles at different surface locations are collected.

It is always a good idea to collect samples of any miscellaneous influent water entering the pit lake before, during, or after filling. Such information is essential if any detailed mass balance of water chemistry is to be attempted. The precise location, approximate flow, and chemical composition of any prominent groundwater springs or collapsed adit discharges into the pit should be quantified. Point sources of discharge can be equipped with flumes and continuous water level recorders to quantify seasonal changes in flow. Diffuse groundwater inputs are much more difficult to gauge precisely but may be approximated by knowledge of groundwater pumping rates prior to the cessation of mining. Permanent or transient (e.g., storm flow) surface water diversions should also be sampled and their individual flows quantified. If possible, try to collect samples of water spilling down the mine walls after a major rain event, especially the first rain after an extended dry period. If the geology of the pit is inhomogeneous, one would ideally collect representative runoff samples from each mappable unit. The paper of Morin and Hutt (2001) gives practical advice on how to collect mine-wall runoff samples, as well as showing how this approach can be used to help predict the ultimate pit lake chemistry.

WHAT TO SAMPLE FOR

The rest of this chapter will center on a list of parameters (Table 7.1) that may be measured in any pit lake monitoring or scientific study. A brief overview is given as to how one might use each type of measurement in a pit lake study, as well some general guidelines on how best to collect the data. Additional details can be found in online documents, such as those by Hem (1985) or U.S. Geological Survey (USGS 2005).

Field Measurements

The date, time, Global Positioning System location, and depth below water surface (or elevation) are all critical parameters that must be recorded in the final project database. Other common field parameters that will be included in most sampling plans are water temperature, specific conductance (SC, also known as C25), pH, dissolved oxygen (DO), oxidation–reduction potential (ORP or Eh), and total suspended solids (TSS) or turbidity. More discussion regarding each of these parameters follows.

Conductivity, temperature, and depth. Electrical conductivity and temperature are two of the most useful parameters to monitor in a pit lake. Combined, they can be used to estimate the salinity (often reported in practical salinity units, or psu) and density (kilograms per cubic meter, kg/m^3)

TABLE 7.1 List of parameters that may be quantified in pit lake studies

Field Measurements	Class	Less Common Procedures	Class
Date and Time		Stable isotopes ($\delta^{18}O$, $\delta^{2}H$) of water	Can be archived
Location		Nutrients (nitrate, phosphate, ammonia)	Time sensitive
Depth or Elevation		Dissolved sulfide	Immediate
Water Temperature	Immediate	Dissolved Fe(II)/Fe(III)	Time sensitive
Conductivity (SC, also known as C25)	Immediate	Dissolved As(III)/As(V)	Time sensitive
pH	Immediate	Dissolved inorgannic and organic carbon	Time sensitive
Dissolved Oxygen	Immediate	Chlorophyll-a	Time sensitive
Oxidation–Reduction Potential or Eh	Immediate	Rare earth elements	Can be archived
Total Suspended Solids or Turbidity	Time sensitive	Radionuclides	Time sensitive
Color/Appearance of Water	Immediate	$\delta^{18}O$, $\delta^{34}S$ of dissolved sulfate	Can be archived
Light Penetration	Immediate	$\delta^{13}C$ of dissolved inorganic carbon, dissolved organic carbon	Time sensitive
Chemical Analysis		**Pit Lake Sediment**	
Total and dissolved metals	Can be archived	Solid phase	Can be archived
Major anions	Time sensitive		Time sensitive
Alkalinity and/or acidity	Time sensitive	Pore water	Time sensitive

Notes: Immediate = collect data in situ or immediately after bringing to surface; time-sensitive = analyze as soon as practical; can be archived = shelf life ≥6 months.

of the water, and how these parameters change with depth. These data are critical for determining whether a given lake is likely to turn over annually (holomictic) or will be permanently stratified (meromictic). Because the conductivity of water increases rapidly with temperature, the normal convention is to report conductivity values corrected to a reference temperature of 25°C. Temperature-corrected conductance measurements are referred to as *specific* conductivity. Most modern conductivity meters will automatically make the temperature correction. It is important to know whether the meter is displaying raw conductance or SC when recording data.

The most obvious way to determine depth below surface is to use a calibrated rope or cable. However, many ropes stretch over time, and therefore the cable should be periodically recalibrated. Also, drift of the sampling vessel across the lake surface from wind can induce an angle in the submerged rope that can lead to a significant overestimation of the true depth at a given sampling point. To avoid these problems, a pressure sensor can be used to determine depth. If the lake water is highly saline, then a density correction may need to be applied.

There are several brands of submersible data sondes that can measure and continuously record temperature and SC readings, as well as other parameters such as pH, DO, Eh, turbidity, and water depth. The costs vary widely but will typically run more than US$5,000 per unit. It is not uncommon to have to replace one or more of the probes on an instrument of this type each year, which can be an unexpected cost to the project. An increasingly popular device for pit lake sampling is the CTD probe. This is similar to the data sondes listed above but has internal circulation of water for more precise temperature and SC measurement, as well as a more sophisticated depth (pressure) sensor. All of these units have the ability to store data in a built-in data-logger.

Because depth is recorded automatically along with SC and temperature, there is no need for the sampler to keep track of depth from the lake surface. To be sure that there is no bias in the data owing to slow equilibration rates, it is recommended to collect complete depth profiles while the multiprobe is both lowered and raised through the water column. It is also recommended to check the calibration of the depth reading of the sonde or CTD according to the manufacturer's instructions.

Finally, to ensure the consistency of all depth measurements, it is important to record the water level of the pit lake at the time of sampling. This can be done using a permanently mounted, finely graduated staff gauge fixed near the shoreline. These measurements allow the altitude of the lake surface to be calculated along with the elevation of the chemocline and other important boundaries.

pH. Measurement of pH is relatively straightforward and needs no elaboration here. If possible, pH should be determined in situ at the same depth that water samples are collected. The pH of a deep water sample may change in response to temperature or pressure changes as it is pumped or bailed to the surface. This is particularly true for lakes with high concentrations of dissolved CO_2. Such waters will lose CO_2 to air by diffusion or effervescence (evolution of CO_2 gas bubbles) in response to pressure drops. Also, the pH of acidic waters containing large amounts of dissolved Fe and sulfate can be quite temperature-dependent on account of the abundance of chemical species such as hydrogen sulfate (HSO_4^-) that are more stable in warm versus cold water.

If in situ pH measurement is not possible, another method is to use a peristaltic pump with lots of inexpensive 0.25-inch-diameter tubing and a flow-through cap on the multiprobe in the boat. If the flow is sufficient and no DO is introduced to the sample chamber, once a stable temperature reading is reached, other parameters will approximate their in situ value. Otherwise, try to collect pH as soon as possible after samples are brought to the surface. In the author's experience with mine waters from Butte, Montana (United States), it is not always a good idea to store nonfiltered and nonpreserved samples for later laboratory pH measurement. For example, a small amount of Fe(II) oxidation during storage can result in a drastic drop in pH.

Dissolved oxygen and ORP/Eh. The oxidation–reduction (redox) state of water is typically characterized by a combination of DO and oxidation–reduction potential (ORP or Eh) measurements. In very broad terms, pit lake waters can be classified into three categories with respect to their redox state:

- *Oxic* (containing DO well above the practical quantification limit, i.e., >1 mg/L);
- *Suboxic* or *transitional* (DO near or below detection, but no H_2S); and
- *Anoxic* or *sulfidic* (H_2S present at detectable levels, i.e., >0.1 mg/L).

The fate and transport of trace metals and other contaminants changes dramatically depending on the redox (and pH) conditions of the water column. Depletion of DO with depth is common in pit lakes and may be an indication of subaqueous O_2-consuming reactions such as oxidation of sulfide minerals on submerged pit walls, oxidation of dissolved metal (such as Fe^{2+} or Mn^{2+}), or decay of organic matter.

Using conventional equipment, it is straightforward to quantify DO in the range of 1.0 to >10 mg/L. Below about 1 mg/L, DO measurements are problematic, and caution should be used to interpret data near the detection limit. In many cases, the trace amounts of DO that are recorded by the equipment will be a false artifact of chemical interferences or pressure effects. Although galvanic DO electrodes are still the most widely used, luminescent DO (or LDO) electrodes are becoming increasingly popular. LDO electrodes are reportedly more sensitive at low DO levels, and are also less prone to calibration drift. For suboxic or anoxic water, an Eh or

ORP electrode is used to quantify redox state. Conversion of field ORP measurements to true Eh must involve either an instrument calibration using a solution of known Eh (usually ZoBell's solution) or manual adjustment of the field ORP reading knowing the temperature-dependent voltage difference between the Ag-AgCl (or $Hg-HgCl_2$) reference electrode used in the field and the standard hydrogen electrode. A good overview of calibration procedures for Eh measurement is given by Nordstrom and Wilde (2005). Properly calibrated, Eh readings should be accurate to within ±10 mV. When budgeting time for field measurements, be aware that in situ ORP readings may take up to 15 minutes to stabilize.

Total suspended solids and turbidity. TSS and/or turbidity measurements are needed to understand how particulate matter is distributed throughout a given pit lake. TSS often increases just above a thermal or chemical boundary layer (such as a thermocline) that impedes gravitational settling of solid particles. Turbidity has a strong influence on light penetration and Secchi depth (see the following "Light Penetration and Secchi Depth" section), which in turn influences biological productivity. Waters with high TSS content may have a density that is significantly higher than the value predicted from SC and temperature alone. This is particularly true of processing slurries, such as mill tailings or lime treatment sludge that are comprised of predominantly fine particles.

TSS is usually obtained by filtering a known volume of water and weighing the residual dried solid. Detailed turbidity gradients with depth can be obtained in situ using a data sonde with a turbidity sensor. It is worthwhile to collect both TSS and turbidity on a subset of samples so that a correlation can be established between the two parameters. Used filter papers from the TSS measurements can be stored for later mineralogical or chemical characterization of the suspended particles. The particles will transform as they dry and oxidize, so forethought should be given as to how any such data will be used.

Light penetration and Secchi depth. In turbid or dark-colored pit lakes, light penetration may be a limiting factor for growth of algae. The simplest way to measure light penetration is to use a Secchi disk. This is a circular plate divided into alternating black and white quadrants that is suspended in the water column at the end of a calibrated rope. The depth below water surface at which point the sampler can no longer see the disk is known as the Secchi disk depth, or simply Secchi depth. The Secchi depth is usually taken as the average of the depth of disappearance while slowly lowering the disk and the depth of reappearance while slowly raising it. A rule of thumb is that sunlight will penetrate to a depth of 1.7 times the Secchi depth, and the *light attenuation coefficient (k)* is simply equal to 1.7 divided by the Secchi depth (in meters). A high k value may reflect high turbidity (suspended particles) or the presence of light-absorbing compounds in the water such as dissolved organic carbon (DOC) or ferric iron. Secchi measurements should be taken within a few hours of noon.

Color and general appearance of the water. Color is rarely used in data reporting of water quality, and yet it can be a useful indicator of redox state, especially for Fe-rich, acidic lakes. For example, the color of water in the Berkeley pit lake has been variously described over the years as "amber," "brown," "red," and "blue-green." There must be something important going on to turn the color of water from amber to blue-green. Unfortunately, this information has not been consistently recorded in the Berkeley pit lake database, and so it is difficult to correlate the color changes to other chemical parameters. Although color is always somewhat subjective, this problem can be minimized by using Munsell color charts. Other observations, such as "the sample was fizzing an odorless gas when pumped to the surface," or "the color rapidly changed from green to red after sampling" are useful and should be recorded.

Chemical Analyses

The list of possible chemical analytes that can be quantified in a given mine water sample is very long, and potentially daunting to someone trying to set up a monitoring plan. However, the biggest costs related to a large-scale monitoring project are often in salaries, subcontracts, and technical oversight, and not in analytical costs. After someone has gone to the trouble of collecting a set of water samples from a given pit lake, it makes sense to try to analyze for as many constituents as possible. Many of these parameters will not be critical in terms of regulatory reporting but nonetheless may be needed if any detailed geochemical modeling studies are done in the future. When in doubt, or if money is tight, archive a set of replicate samples for later analysis. Many of the chemical parameters listed below are not time-sensitive, and therefore the samples can be stored indefinitely. The Nevada (United States) Division of Environmental Protection Profile I and Profile II lists of analytes provide good starting points for developing analytical plans.

Metal concentrations. Coming up with a list of major and trace metals to be monitored will be one of the first objectives of the sampling plan for any pit lake project. This list will often include, at a minimum, all of the metals obtained from a routine ICP-AES (inductively coupled plasma atomic emission spectroscopy) laboratory analysis. Determinations of P and S should be considered as part of the ICP scan if a vacuum or purged path spectrometer is utilized. However, ICP-AES may not have a detection limit low enough for some contaminants of concern, such as As, Cd, Pb, or U. In this case, split samples can be analyzed by a more sensitive method, such as GF-AAS (graphite furnace atomic absorption spectroscopy) or ICP-MS (ICP mass spectroscopy). To prevent precipitation and adsorption reactions within the sample bottle, aqueous samples should be acidified with a few drops of concentrated nitric acid (HNO3) to lower the pH below 2.0. If dissolved metal concentrations are to be determined, then samples should be filtered prior to acidification. Samples should be stored at approximately 4°C until analysis.

The question of whether to collect filtered (usually 0.45 µm) versus nonfiltered samples for metals analysis is an important one and will be heavily influenced by the nature of the applicable regulatory standards (e.g., total-recoverable versus dissolved). It is a good idea to analyze a subset of the total number of samples for both filtered and nonfiltered metal concentrations. Whereas site regulatory standards may be based on total recoverable concentrations, dissolved concentrations will be more useful in thermodynamic modeling of pit lake water chemistry. In addition, a comparison of total versus dissolved concentrations will give insight into how a given metal is partitioned between the aqueous and solid phases.

Anions. Analyses of major anions are usually done on filtered (0.45 µm) samples by ion chromatography (IC). A routine IC analysis will quantify bromide (Br^-), chloride (Cl^-), nitrate (NO_3^-), and sulfate (SO_4^{2-}). Phosphate (PO_4^{3-}) and nitrite (NO_2^-) can also be determined if concentrations are above the detection limits of the IC. If accurate analyses of low-level nutrient concentrations are critical, then colorimetric or gravimetric analyses may be more appropriate (see "Nutrients" section later in this chapter). Use of gravimetric sulfate analyses may be required to determine high concentrations of sulfate, particularly in highly acidic samples. However, many colorimetric tests suffer from matrix interferences that must be carefully evaluated with site-specific QC tests. Bicarbonate (HCO_3^-) and carbonate (CO_3^{2-}) cannot be identified by IC; these species are usually quantified indirectly from alkalinity and pH.

Alkalinity and acidity. Alkalinity is defined as the sum of the titratable bases in a water and is usually determined by the amount of acid that is needed to lower the pH of 100 mL of sample to an endpoint of ~4.5. Alkalinity is a critical measurement because it is needed along with pH to determine the concentrations of dissolved CO_2 (also referred to as carbonic acid, or H_2CO_3), HCO_3^-, and CO_3^{2-} in the water. Alkalinity titrations are usually performed on unfiltered samples. Because unfiltered water samples can undergo chemical reactions between the time they are collected and when the samples are analyzed in the lab, it is a good idea to perform alkalinity titrations in the field, or as soon as practical after sample collection. To minimize sample degradation, there should be zero head space in the bottle after collection of an alkalinity sample, and the bottles should be stored at 4°C until analysis.

An acidity titration is essentially the opposite of an alkalinity measurement: a known volume of initially acidic sample is titrated with dilute base (usually sodium hydroxide or NaOH) to a pH end point (usually 8.3). The addition of hydrogen peroxide is also recommended to convert metals to their fully oxidized forms before filtration and titration of the filtrate for samples with high dissolved metal concentrations (ASTM D 1067, 2006). However, acidity titrations are more complicated to perform than alkalinity measurements, and results are very method-sensitive (see Kirby and Cravotta 2005a and 2005b for a good discussion). Because of these problems, some researchers prefer to skip direct acidity measurements and instead will calculate this parameter knowing the pH and metal concentrations from ICP-AES, and also knowing the speciation of dissolved Fe between Fe(II) and Fe(III) (see the "Redox Speciation of Iron, Arsenic, and Other Metals" section later in this chapter).

Less Common Procedures

Aside from the common chemical and physical parameters enumerated above, there are many additional types of information that one could collect to help understand pit lake biogeochemistry. Because some of these approaches may be unfamiliar to the reader, Table 7.2 gives references for published case studies for each method.

Stable isotopes of water. Stable isotope analyses ($\delta^{18}O$ and $\delta^{2}H$, also known as δD) of water samples can provide valuable hydrological information about a pit lake that is impossible to obtain from conventional chemical analyses alone in many cases (Seal 2003). For example, stable isotopes can be used to quantify different sources of water into a pit lake (e.g., relative contributions of direct precipitation, groundwater inflow, and surface water inputs) if the isotopic compositions of the different sources are distinct from one another. Stable isotopes can also be used to estimate the approximate amount of evapoconcentration that has occurred within the pit lake (Gammons et al. 2006). Although inclusion of stable isotope analyses adds to the total project budget, the

TABLE 7.2 List of parameters that may be quantified in pit lake studies

Topic	References
Stable isotopes in pit lakes	Eccarius 1998; Asmussen and Strauch 1998; Seal 2003; Seal et al. 2004; Kn□ller et al. 2004; Pellicori et al. 2005; Gammons et al. 2006a
Fe redox speciation	Jonas 2000; Madison et al. 2003; Pellicori et al. 2005; Gammons and Duaime 2006
As redox speciation	No references found
Nutrients and chlorophyll-a	Cobelas et al. 1992; Axler et al. 1998; Jonas 2000; Ramstedt et al. 2003; Dessouki et al. 2005; Fisher and Lawrence 2006
Dissolved sulfide	Whittle et al. 2003; Asmussen and Strauch 1998; Kn□ller et al. 2004
Dissolved organic carbon	Cobelas et al. 1992; Ramstedt et al. 2003; Cameron et al. 2006
Rare earth elements	Wolkersdorfer 2002; Gammons et al. 2003
Radionuclides	Bain et al. 2001; Gammons et al. 2003; Lottermoser et al. 2005
Age dating of water in pit lakes	No references found
Pit wall runoff characterization	Morin and Hutt 2001
Pit lake sediment characterization	Friese 2004; Twidwell et al. 2006

costs are relatively small (usually less than US$80 per sample for a combined $\delta^{18}O$ and δD analysis). To collect a sample, simply fill a small bottle (10 mL is plenty) with filtered water and cap tightly with no head space. Isotope samples are easily compromised by interacting with the atmosphere and should be collected in gas-tight, glass bottles. "Polyseal-lined" caps on glass bottles are recommended. Even if money is not available now, it makes sense to collect isotope samples during each pit lake sampling event and archive them for later use. One important instruction is to never add acid or any other preservative to a bottle that will be used for water isotope analysis. The reagent will have its own water isotope signature that may be very different from that of the sample.

Nutrients. To completely understand the biology of a given pit lake, information is needed on the distribution of nutrients (nitrate, nitrite, ammonia, phosphate) in space and time. Nitrification of mining pit lakes is a common topic of study in the recent literature, and there are numerous examples of the use of limnocorrals to test different nutrient addition schemes (e.g., Whittle et al. 2003). In addition to IC, there are many field colorimetric tests for analysis of nutrients. Some of these tests have detection limits that are much lower than conventional IC analysis. These tests require a portable spectrophotometer, which will typically run in the US$1,000 to US$4,000 price range. However, colorimetric tests are notorious for matrix interferences that can be serious. In many cases, these problems can be overcome by use of appropriate QC procedures (matrix-matched standards, spikes, duplicates), but in other cases, high concentrations of an interferent may make a particular colorimetric test impossible.

Dissolved sulfide. A pit lake may develop an anoxic bottom layer where bacterial sulfate reduction occurs. If so, then dissolved sulfide (hydrogen sulfide, H_2S, and bisulfide, HS^-) may accumulate to measurable concentrations in the overlying water column. Dissolved sulfide is an extremely reactive and also highly toxic compound that should be high on the priority list for quantification if there is a possibility that it is present. The sampler will know whether H_2S is around because it has a very strong, unpleasant smell of rotten eggs. The simplest way to analyze for H_2S is the methylene blue colorimetric test. This requires a portable spectrophotometer (see the "Nutrients" section previously in this chapter). The reagents are inexpensive, and the test is both simple and highly sensitive. Because H_2S is volatile and degrades quickly in the presence of oxygen, sulfide analyses should be performed in the field immediately after sampling.

Redox speciation of iron, arsenic, and other metals. Determination of the redox speciation of dissolved Fe is highly recommended, especially for acidic pit lakes. If the pH of the water column is above about 4 and iron concentrations are above detection limits, nearly all of the dissolved Fe will likely be ferrous (Fe^{2+}). However, at lower pH the solubility of ferric iron (Fe^{3+}) increases and both redox states can coexist. Highly acidic pit lakes such as the Berkeley pit contain hundreds or even thousands of milligrams per liter of Fe, and the Fe(II)/Fe(III) profile with depth reveals much about the redox structure of the water column (Jonas 2000; Madison et al. 2003; Pellicori et al. 2005; Gammons and Duaime 2006). Information on the concentrations of both Fe oxidation states is essential if any geochemical modeling is going to be performed.

There are several different colorimetric tests to speciate dissolved Fe, all of which will require a field or laboratory spectrophotometer. One of the more popular tests is the Ferrozine method (Stookey 1970). A conventional Ferrozine speciation involves the independent analysis of dissolved Fe(II) and total dissolved Fe. Dissolved Fe(III) is then determined by difference. Although it is probably a good idea to do the Ferrozine tests on-site if time allows, it is also possible to preserve samples in the field for later laboratory analysis. Sample preservation entails collection of a filtered sample (0.45 µm or smaller pore size) into an opaque bottle (to prevent photoreduction of Fe(III)) with addition of hydrochloric acid (HCl) (*not* nitric acid, HNO_3) to 1% v/v. The bottle should be sealed with no head space, wrapped with parafilm, and stored on ice or in a refrigerator until analysis. Oxidation of dissolved Fe(II) is extremely slow in cold, acidic water in the absence of bacteria, even if low levels of DO are present.

Depending on the project, it may be useful to speciate other redox-sensitive elements, such as As and Se. Both of these metalloids are major contaminants of concern in many pit lakes, especially those in central and northern Nevada (Tempel et al. 2000).

Dissolved inorganic carbon. The total dissolved inorganic carbon (DIC) concentration in a natural water sample is given by the following equation:

$$DIC = [H_2CO_{3,aq}] + [HCO_3^-] + [CO_3^{2-}] \qquad \text{(EQ 7.1)}$$

$H_2CO_{3(aq)}$ is the dominant form of DIC at pH <6.3, CO_3^{2-} is dominant at pH >10.3, and HCO_3^- is dominant between these pH values. The concentration and speciation of DIC may be important for geochemical, limnological, or biological modeling of a pit lake. For example, the solubilities of common carbonate minerals such as calcite, dolomite, or siderite are strongly dependent on the partial pressure of CO_2 (i.e., the pCO_2). As shown by Eary (1999), it is common for water in pit lakes to have a pCO_2 that is very different from that of the atmosphere (currently $\sim 10^{-3.4}$ atmospheres but rising slowly on account of global combustion of fossil fuels). As well, elevated concentrations of dissolved CO_2(g) can increase the total dissolved solids (TDS) of a water without increasing the SC, and this has implications to density calculations. An extreme example is the hypolimnion of Lake Nyos, a volcanic crater lake in Cameroon. As shown by Schmid et al. (2004), a large proportion of the TDS of the deep water in this lake is CO_2(aq). Rapid exsolution of this dissolved gas during a lake turnover event in 1986 led to catastrophic loss of life of humans and livestock.

DIC can be directly measured in the laboratory using a carbon analyzer (coulometric titration). This method converts all forms of DIC to CO_2 and then measures the quantity of the CO_2(g) produced. For waters that have a measurable amount of alkalinity, it is sometimes more convenient (and more inexpensive) to simply calculate the concentration of each inorganic C species based on the field pH, alkalinity, and temperature using a geochemical modeling program such as PHREEQC or Visual MINTEQ. However, if DIC is an important parameter, it is probably a good idea to run some samples on a carbon analyzer to confirm the geochemical modeling results. For acidic waters with pH <4.5, nearly all of the DIC will be present as H_2CO_3, and a C analyzer must be used to approximate DIC.

Dissolved organic carbon. There are a number of ways that DOC may be important in pit lake studies (Cameron et al. 2006). Soluble DOC compounds containing carboxylic acid functional groups can increase the mobility of otherwise insoluble metals (such as Al or Fe at high pH). Low-molecular-weight DOC compounds are the "food" that sustains many heterotrophic bacteria, including sulfate-reducing bacteria. DOC also attenuates light, and plays an important role in many photochemical reactions, such as Fe(III) photoreduction. The detection limit for DOC in water varies from laboratory to laboratory, but it is typically around 1 mg/L as C. High sulfate concentrations pose a challenge on account of evolution of copious sulfur dioxide (SO_2) that can interfere with the C analysis (Cameron et al. 2006).

Chlorophyll-a. Measuring the concentration of chlorophyll-a as a function of depth is a useful way to assess the primary productivity of a pit lake. Chlorophyll-a can be estimated quickly on raw samples using a portable fluorometer. A more accurate analysis involves filtration of a known volume of water (usually 1 L) followed by digestion of the algae collected on the filter in acetone. The concentration of chlorophyll-a can then be determined by fluorometry or by using a spectrophotometer.

Rare earth elements and radionuclides. Rare earth elements (REE, i.e., lanthanides) are usually not a concern from a water quality point of view but are sometimes used by geochemists to gain insight into processes such as weathering, water–rock interaction, mineral precipitation, or adsorption. Although all of the REEs have similar ionic size and mass, the aforementioned geochemical processes can sometimes induce subtle fractionation trends across the lanthanide series. Examples of studies where REE geochemistry has been applied to mining pit lakes include works by Gammons et al. (2003) and Wolkersdorfer (2002).

Radionuclides include long-lived radioactive isotopes of uranium of ^{234}U and ^{238}U, as well as more dangerous isotopes with shorter half-lives, such as the radon isotopes ^{226}Ra and ^{222}Rn. Being a gas, radon has its own peculiar characteristics with respect to how it migrates in the subsurface, as well as presenting some difficulties with respect to sample collection. Collection of samples for uranium or radium analysis is more straightforward, although speciation at an isotopic level can be expensive. Unless the mineral deposit being mined was known to be enriched in uranium, it is doubtful that radionuclides will be on a priority list for quantification. Nonetheless, surprisingly high concentrations of elements such as uranium may be present. For example, about 1 mg/L of dissolved uranium is present in the Berkeley pit lake (Gammons et al. 2003), a deposit mined chiefly for copper.

Less common isotopes. The author's research group in Butte has found that stable S and O isotopes of dissolved sulfate are particularly helpful to discriminate between different sources of sulfate (e.g., dissolution of anhydrite vs. pyrite oxidation), as well as providing some insight into the mechanisms of sulfide mineral oxidation (e.g., aerobic vs. anaerobic). The interested reader should read works by Pellicori et al. (2005) and Seal (2003) for recent discussions. Other systems that may be useful include Sr isotopes and $^{13}C/^{12}C$ analysis of DIC (e.g., is DIC derived from mineral dissolution or oxidation of organic matter?). There is currently much discussion in the geochemical community about the putative fractionation of stable isotopes of heavy metals, such as Cu, Fe, or Zn, during abiotic or microbially catalyzed reactions. This author is not aware of any papers that have specifically applied this approach to pit lakes. No doubt someone will do this soon.

Pit Lake Sediment and Pore Water

Finally, although somewhat outside the scope of this chapter, it may be of interest to collect core samples of pit lake sediment and pore water for chemical, mineralogical, or microbial analysis. This type of information may not be critical for regulatory work, but it is a very useful way to get information on microbial activity in the benthic sediment and minerals precipitating from lake water. For example, a zone of bacterial sulfate or ferric iron reduction may be present at shallow depth in the lake sediment, whereas no reduced S or Fe is present in the overlying water column. Because pit lake sediment is often fine-grained and unconsolidated, it is usually possible to extract an intact core sample simply by dropping a weighted sampler by gravity to the bottom of the lake. The core sample can be sectioned for analysis of pore water chemistry (including pH, Eh, DOC, and dissolved metal concentrations), sediment mineralogy (e.g., by X-ray diffraction or scanning electron microscopy), and solid-phase chemistry. Special care should be taken if redox-sensitive measurements are to be collected, such as Eh or Fe(II)/Fe(III) analysis (see Twidwell et al. 2006 for a recent example).

Pore waters in pit lake sediment can be extracted by filtration or centrifugation of fresh core samples delivered to the laboratory. Another option is to collect in situ samples of sediment pore water using diffusion samplers, or "peepers." A peeper relies on diffusion of solutes across a semi-permeable membrane to collect a vertical array of representative pore water samples. The peeper is typically installed in shallow sediment at the bottom of a lake and allowed to sit undisturbed for a period of one week or more. After hauling to the lake surface, the peeper samples are extracted for pore water chemistry using a portable glove box if redox-sensitive measurements are taken. The interested reader is referred to works by Martin and Pedersen (2002) for an excellent case study. Unfortunately, installation of peepers will be extremely difficult for deep pit lakes, and lack of deep sediment may pose insurmountable problems for their practical use in many pit lakes.

CONCLUSIONS

The purpose of this chapter has been to give some ideas and guidelines as to how to formulate a water sampling plan for a mining pit lake. Considerations such as where to sample, how often to sample, and how many samples to collect per event are important, and are difficult to generalize to all pit lakes. An even more daunting task is to decide on a list of analytes to be quantified. However, given that it is very often the case that analytical costs are a relatively minor portion of the total project budget for large-scale monitoring projects, it is strongly recommended that as many constituents be quantified on as many samples as is practical. Having a complete chemistry database of the lake and any influent groundwater or surface waters will be invaluable in any attempt to construct a detailed hydrogeochemical model of the system.

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CHAPTER 8

Predicting Groundwater Inputs to Pit Lakes

W.L. Niccoli

INTRODUCTION

Groundwater is typically the most important source of water to a pit lake (see Figure 4.7 in Chapter 4). Groundwater inflow provides water as well as chemical mass additions to the pit lake. Thus, the purpose of this chapter is to discuss predictions of the quantity and quality of groundwater inflow to pit lakes. Specifically, this chapter will attempt to address the following:

- How is the volume of groundwater input predicted over 10-year, 50-year, 100-year, or 500-year time spans?
- How is the chemistry of groundwater input predicted over 10-year, 50-year, 100-year, or 500-year time spans?
- How can climate change be factored into prediction models?
- What models are commonly used, how do these models work (in general), what data are required, and how are these data collected?
- What is the accuracy of these predictions, and how have models been validated?
- What are the current data gaps, major assumptions, and limitations of these predictions?

In this chapter, groundwater inflow is defined as the groundwater that is upgradient of the zone of mining influence (Figure 8.1). The chapter discusses:

1. Building a robust conceptual model in time and space that describes the known processes of inflowing groundwater quantity and quality
2. Using the conceptual model to chose an appropriate tool to quantify groundwater inflow quantity and quality predictions
3. The uncertainty of predictions
4. The data gaps associated with groundwater inflow predictions.

PREDICTING GROUNDWATER INFLOW

The steps of predicting groundwater flow have been discussed in many university classrooms as well as by several prominent books on the subject (e.g., Spitz and Moreno 1996; Anderson and Woessner 2002). The popular groundwater modeling software, Visual MODFLOW by Waterloo Hydrogeologic, now has a practice problem that involves predicting groundwater flow to a pit lake. Typical papers on pit lakes (e.g., Blair and Parizak 1991; Niccoli et al. 2004) also describe

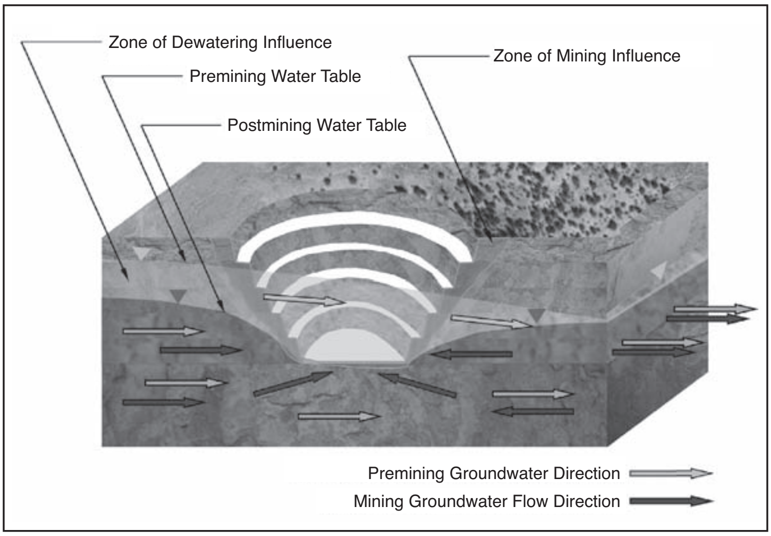


FIGURE 8.1 Zones of groundwater influence

approaches for predicting groundwater flow to pit lakes. Thus, the intent of this chapter is to summarize what is well known in the hydrologic community.

Predicting groundwater inflow should take place in three steps: (1) define the questions that are ultimately being asked (i.e., define the problem), (2) conceptualize the system, and (3) quantify the system appropriately to answer the questions being asked. While the focus in recent time has been on using numerical models for the quantification step, the procedures remain the same regardless of what method is used for quantification.

The first step in any modeling (predictive or forensic) effort must start with definition of the question being asked. Related to groundwater inflows associated with pit lakes, the questions may include

1. Compared to other inflows (e.g., rainfall), what is the significance of groundwater inflow?
2. Will groundwater inflow balance outflow at some point in the future?
3. Will groundwater inflow rates change over time?
4. What are the expected magnitudes of groundwater inflow?
5. Where will groundwater inflow occur?
6. Will groundwater inflow cause stability problems in the pit?
7. What will the groundwater inflow quality be over time?
8. At what time will groundwater inflow to the pit reach a steady state?

The second and equally important step to any modeling or calculation effort is the development of a robust conceptual model that is an accurate representation of the system. A conceptual model may be defined as a semiquantitative description of the system under analysis. Although Chapter 4 describes the steps and procedures associated with conceptualizing the entire pit lake, a subset of the overall pit lake model must address the hydrologic system that controls groundwater flows into the pit. The level of detail to which the conceptual model is built should be commensurate with the question at hand. For example, if the question is to define the quantity of groundwater inflow, the conceptual model should focus on the regional processes (e.g., streams, precipitation recharge) that supply groundwater in the area of the pit. If the question is where groundwater will enter the pit, then focus should also be placed on defining the features (e.g., fracture patterns, faulting) that might control the local groundwater flow fields. The conceptualization for groundwater inflow must include

1. **A framework**—a description of the geology (size, extent, and type) in which groundwater inflow takes place and a quantification of the hydraulic properties of the geologic materials;
2. **Boundary conditions**—descriptions and quantifications of sources of groundwater, restrictions, and constraints to groundwater flow; and
3. **Stresses on the system**—such as mining an open pit, pumping from a well field, or infiltration from a water disposal site.

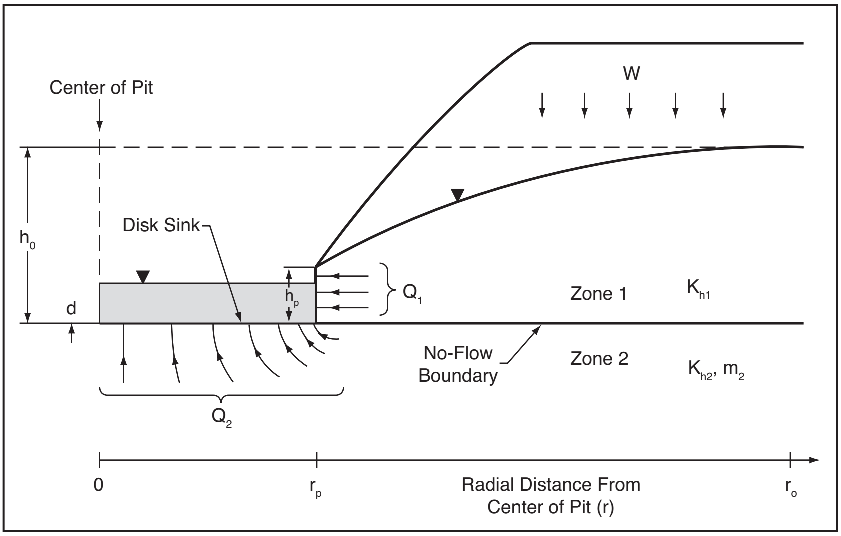
The conceptual model should consider the full range of potential conditions, such as the extreme conditions placed on a hydrologic system during the period considered for prediction, and provide realistic ranges for parameters necessary to quantify the analysis for average and extreme conditions.

Once the important factors controlling groundwater inflow are identified and a conceptual model is built, the problem can be quantified (i.e., modeled). The method of quantification depends on the data available, the problem or question being asked, and the conceptual model. In mine pits that have undergone dewatering and where records are available (e.g., pumping records from dewatering, flow rate changes with pit elevation, groundwater monitoring well data, monitoring well chemistry data, high wall seep and spring flow rates, geology and geologic structures within the pit, surface runoff flow data, and climatologic data), the approach for estimating groundwater inflow may simply consist of a thorough record review coupled with basic understanding of hydrology. Straightforward approaches such as those described by Marinelli and Niccoli (2000) and analytical element models (Fitts 1997) will likely be able to address most of the questions associated with general groundwater inflow (e.g., what is the magnitude of groundwater inflow?) associated with a pit lake. To put into perspective the types of data that are utilized in a straightforward analysis, the equations and parameter definitions from Marinelli and Niccoli (2000) are provided herein (Figure 8.2):

$$Q_{total} = Q_1 + Q_2 \qquad \text{(EQ 7.1)}$$

where

Q_{total} = the total inflow to the pit
$Q_1 = W \times \pi \times (r_0 - r_p)$
W = the aerial recharge derived from regional information
r_0 = the radius of influence
r_p = the radius of the pit
$Q_2 = 4 * r_p * (K_{h2}/m_2) * (h_0 - d)$
$m_2 = (K_{h2}/K_{v2})^{1/2}$
K_{h2} = the horizontal hydraulic conductivity in zone 2
K_{v2} = the vertical hydraulic conductivity in zone 2
h_0 = the hydraulic head before dewatering
d = the depth of water in the pit



Source: Adapted from Marinelli and Niccoli 2000.

FIGURE 8.2 Groundwater inflow parameters

This equation is provided as an example, and before it should be used for predicting pit inflows, the reader is encouraged to examine the conditions for which this equation applies.

In the last 10 years, there have been large strides made in the technology required to make numerical groundwater models more practical. Contemporary software packages include Visual MODFLOW (produced by Waterloo Hydrogeologic), Groundwater Vistas (produced by Scientific Software, Inc.), Groundwater Modeling Systems (produced by Environmental Modeling Systems, Inc.). Improvements in graphical user interfaces and computing power have increased the complexity and number of parameters that can be considered in the analysis of groundwater inflow. However, just because a model can simulate very complex systems with multiple variables does not automatically mean that the final result is more correct or appropriate than a general analysis developed from observations of site hydrologic behavior. It is common that one does not know the magnitude of most of the parameters that are necessary to operate a complicated model. Thus, a more complex model may not yield a more robust solution. Figure 8.3 illustrates diminishing returns with added complexity to analysis in terms of the understanding gained. Kelson et al. (2002) provide a thorough analysis of a case where a complex model was used to address a straightforward system. The additional complexity introduced by the ability of the numerical model to consider more parameters did not justify the uncertainty added to the model results.

However, there are cases where a more complex model is necessary to address the questions at hand (e.g., a pit that has complex geometry and intersects multiple aquifers where the head distribution near the pit wall is critical to wall rock stability). More complexity in a modeling approach is typically desired when the scale of the problem or question at hand gets smaller. For example, sometimes it is desirable to know from which zones of the pit groundwater is generated in what quantities because this could have a dramatic influence on the pit lake water quality.

Before developing a more complex modeling approach, it is important for the investigator to consider the additional costs that more complex approaches add to the modeling effort and whether or not the costs are justified by the gain in model certainty. The point is that the investigator should choose the tools wisely so that appropriate accounting is given to the question being

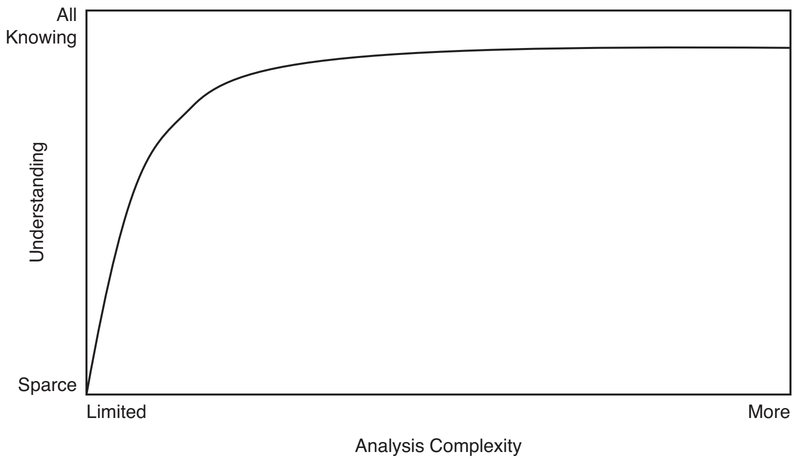


FIGURE 8.3 Analysis complexity vs. understanding

posed and the answer required. The purpose herein is not to suggest that one approach is better than another but to encourage a search for choosing the right tool to address a given situation or problem of quantifying groundwater inflow.

PREDICTING GROUNDWATER INFLOW QUALITY

Predicting groundwater inflow quality follows the same general approach as predicting groundwater inflow quantity. The first step is to define the questions to be addressed by the prediction (see Chapter 3). The remaining steps are conceptualization of the geochemical system associated with the groundwater inflow followed by quantification.

This chapter defines groundwater inflow (and the method that it is defined in most pit lake predictive exercises) as the groundwater flow that occurs before it reaches the area surrounding the pit that has been influenced by mining activities (e.g., dewatering, blasting). Figure 8.1 displays a conceptualization of groundwater inflow, mining influenced area (i.e., areas of geologic strata physically modified by mining activities), and dewatering influenced area associated with a pit lake. Groundwater inflow quality is typically modified by the near-pit perturbations, such as increased fracture density and oxygen availability to sulfide mineralization in wall rock. Prediction of this chemical mass addition caused by the near-pit perturbations is described in Chapter 10. There are typically few perturbations made to groundwater inflow chemistry prior to its entering the zone of mining influence, and in many predictive cases, the groundwater inflow chemistry is defined as a constant. This constant chemistry is typically based on historic, upgradient groundwater monitoring.

However, the investigator should examine the processes that dictate upgradient groundwater flow and evaluate the potential for these processes to change within the term of the prediction. Potential impacts to groundwater quality upgradient of the pit could be due to anthropogenic effects (e.g., mine development, metallurgical process and mine waste disposal facility impacts to groundwater). Miller (2002) discussed the potential impact to groundwater quality related to the initial lowering and subsequent rise of the groundwater table on account of mine dewatering and pit lake filling. Care should thus be taken to ensure that the zone of dewatering influence (Figure 8.1) is considered in any long-term prediction of inflowing groundwater quality. However in most cases, the mass produced from the zone of mining influence is several orders of magnitude greater than that produced in the zone of dewatering influence, mainly because most of the natural mineralization occurs near the open pit location, which is also why the mine is located where it is, and increased surface area of mineralization because of mining induced fractures. In the few cases where mineralization occurs in the zone of dewatering influence, the conceptual quality model should consider influences of this zone on

- Potential oxidation of sulfide minerals and accumulation of oxidation by-products,
- Precipitation recharge as it makes its way to the groundwater system, and
- Groundwater flushing through this zone after groundwater rebounds.

Natural changes may also influence the quality of groundwater inflow and should be considered in the conceptual quality model. For example, a fluctuating groundwater table due to drought conditions may expose previously unoxidized sulfides resulting in releases of metals and sulfate that could change the inflowing groundwater chemistry. Wet conditions could increase precipitation recharge to the groundwater system and dilute the system.

CLIMATE CHANGE AND GROUNDWATER INFLOW QUANTITY AND QUALITY PREDICTIONS

As with any change to a system, the impacts of climate change on groundwater quantity and quality will depend on how the climate affects the entire groundwater system. Unlike surface flow, groundwater tends to respond in an attenuated manner such that immediate changes observed at the surface may not be seen in the underground system for a longer period of time, especially in deep groundwater systems. Those groundwater systems that have a larger attenuation capacity (e.g., systems associated with deep hard-rock mines) will not be influenced by climate change as readily as more shallow systems (e.g., systems associated with alluvial gravel pits). Ultimately all groundwater is derived from precipitation, and thus climate change will ultimately impact groundwater inflow quantity and quality to a pit lake. Therefore, potential climate change should be one of the components considered in any conceptual model that is addressing issues for a period of time that is longer than the data record on which the analysis is based. In general, predictions much beyond 50 years should include an estimate of climate change. Predicting how climate change influences precipitation and net groundwater recharge is discussed in Chapter 3.

GROUNDWATER INFLOW PREDICTIVE UNCERTAINTY

As with any predictive analysis, there is inherent uncertainty because not all controlling parameters can be quantified in their future state. Thus, whenever predictive modeling is undertaken, consideration must be given to the amount of uncertainty associated with the prediction in order to interpret the prediction results in light of the problem being addressed. Various approaches are available for dealing with uncertainty, from evaluating the extremes of parameter inputs (i.e., a worst- and best-case analysis) to a quantitative uncertainty analysis such as that described by Niccoli and Finley (1999) and Niccoli et al. (1998).

The level of uncertainty associated with any analysis is dependent on the amount and quality of data used and the complexity of the problem at hand. Figure 8.4 illustrates the relationship between the quantity and quality of data, the problem complexity, and the resulting uncertainty. Shown on Figure 8.4 are zones that describe typical types of mines under analysis.

There is high uncertainty if limited or poor data are used in the analysis and the problem at hand is complex. For example, high uncertainty might exist in a case where the geology has not been well defined and there are only three observation wells to define the hydrogeologic system.

In this case, it is possible that aquifer pumping tests may yield hydraulic conductivities that range over five orders of magnitude. This large range in hydraulic conductivity leads to substantial uncertainties for the mine dewatering team in knowing where groundwater will inflow into the pit and at what quantity for pit wall stability reasons. This situation may be the case for new mines. The problem with analyses that have a high uncertainty is that there typically are not enough data to allow the uncertainty to be quantified. In these types of cases, professional judgment and planning play large roles in interpretation of the available hydrologic data. Professional judgment often can be used to bind the parameters involved in the analysis, while planning allows more information to be collected and the uncertainty quantified over time as a project progresses.

Predictions with low uncertainty also exist when they are based on relatively good and abundant data and the problem under scrutiny is straightforward. For example, uncertainty in predicting groundwater inflow is low in a case where a mine pit has been dewatered for 30 years, the other components of the water balance have been measured, multiple monitoring wells exist around the mine pit, and the problem is deciding if the pit lake will form and be terminal or flow-through. In this example case, the abundant data greatly reduces uncertainty in estimating the water balance associated with the pit lake system and, thus, predictions of the final status of the pit lake are robust.

One additional type of uncertainty that occurs from time to time during groundwater hydrologic studies of pit lakes is what can be described as "fictional uncertainty." Fictional uncertainty typically arises from questions related to hydrologic components for which there are no evidence of the feature's existence, which means that the feature is not part of the conceptual model. A typical example arises when a pit lake is predicted to be terminal. The question often raised is in regard to the existence of a high flow capacity fracture extending from the bottom of the pit that could move water away from the pit. To rebut the justification for such a process when there is no evidence of the fracture's existence is the impossible task of proving a negative or attempting to validate a process that physical science does not support (e.g., moving water upgradient). To move forward with such an analysis is simply to invoke one variation of the worst-case scenario.

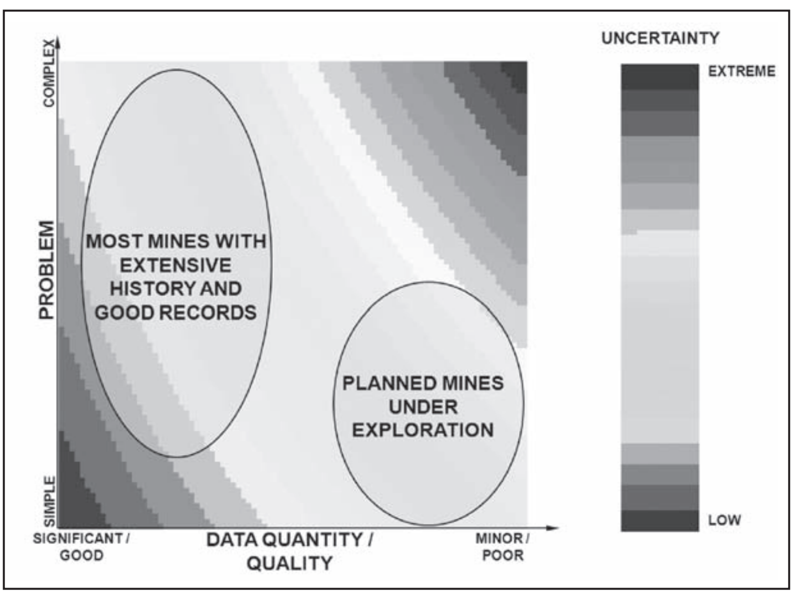


FIGURE 8.4 Matrix of uncertainty

In either event, this type of uncertainty analysis is neither helpful nor productive. Rather, greater emphasis should be placed on defining and quantifying the processes that make the pit lake terminal (e.g., groundwater heads in the flow field surrounding the pit lake) and increasing the overall understanding of the processes associated with groundwater flow.

HYDROLOGIC CHARACTERIZATION DATA GAPS

As described in the previous section, the groundwater inflow component of a pit lake water balance can be fairly well defined depending on the stage of data collection and mining. The methods of groundwater inflow data collection, conceptualization, and quantification are well established. Yet, uncertainty will always remain.

One major data gap that this author feels exists in relation to predictive groundwater analysis is a need for an expanded understanding among mining companies, groundwater practitioners, regulatory agencies, nongovernmental organizations, and the public in regard to uncertainty analyses. Our system of regulation and litigation in the United States is heavily dependent on quantifying, defending, permitting, and regulating on "single number" results, when in fact, where groundwater is concerned, predictions are always uncertain and the more appropriate result is a range of values. Quantifying the range of prediction values would allow better judgment in the decisions made based on predictive analyses. For example, if an uncertainty analysis shows that a prediction results in a 99.99% chance that the outcome would not impact the regulated environment, then the postanalysis monitoring program can be somewhat minimized. In contrast, if the uncertainty analysis shows a significant uncertainty that the regulated environment could be impacted, then mitigation measures can be planned in advance and a rigorous monitoring program developed. This data gap can be filled by our instructional institutions with more emphasis being placed on developing and training practitioners in stochastic hydrology. Renard (2007) discusses the history, usage, and future use of stochastics in hydrogeology and emphasizes this point.

One additional data gap in regard to predictive analyses of groundwater flow is that of data validation. Typically, groundwater hydrologic analyses are calibrated to data collected at one point in time. Although this step is critical in establishing a robust analysis, a comparison of analysis results to data collected in the future and a reexamination of the analyses would enhance the overall understanding of uncertainty and our predictive capabilities.

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CHAPTER 9

Predictive Modeling of the Physical Limnology of Future Pit Lakes

D. Castendyk

INTRODUCTION

By mixing two solutions with different chemistry and different volumes, a new solution with a unique chemistry will be produced. In pit lakes, surface waters exchange oxygen and carbon dioxide with the atmosphere, host photosynthesizing microorganisms, and receive both rainwater and pit wall runoff, whereas deep lake waters are oxygen limited and receive only groundwater inputs. Consequently, vertical mixing events, called turnover, affect pit lake water chemistry. For this reason, the frequency of turnover events, the maximum depth of circulation during turnover, and the volumes of mixing layers should be determined prior to the development of geochemical water quality predictions. Lakes that undergo complete turnover on an annual basis are called holomictic, whereas lakes that do not fully circulate exhibit a permanently isolated bottom layer and are called meromictic. Boehrer and Schultze (2006) provide a comprehensive review of existing meromictic pit lakes. Predictive modeling of physical limnology can indicate whether a future pit lake should become holomictic or meromictic.

This chapter reviews the factors that influence turnover in pit lakes, numerical models that have been applied to pit lakes, and data required to predict the physical limnology of future pit lakes using the one-dimensional (1-D) numerical model DYRESM (Dynamic Reservoir Simulation Model) (Imberger and Patterson 1981; Antenucci and Imerito 2001). Examples of DYRESM results are provided and the limitations of predictions are discussed.

FACTORS INFLUENCING PIT LAKE TURNOVER

Seasonal Changes in Water Temperature

For both natural and human-made lakes, the frequency of turnover is largely controlled by seasonal variations in the amount of solar radiation received by the surface water layer and the resulting buoyancy of this layer. At the start of spring, the water column has a uniform temperature and density and offers minimal resistance to vertical mixing. Wind energy applied to the lake surface generates a downward mixing force, and, under spring turnover conditions, this force homogenizes water chemistry with depth. As spring progresses, the number of hours of daylight increases and the sun rises higher in the sky such that the surface of the lake receives more solar radiation, which raises the temperature and lowers the density of the surface layer relative to underlying water. Because low-density water is more buoyant than high-density water, solar radiation creates an upward buoyancy force that resists the downward mixing force caused by wind. Initially, the downward mixing force is able to overcome the upward buoyancy force, which causes warm surface water to mix with the entire water column and raises lake water temperatures at depth.

At some point following a period of quiescent wind activity coupled with clear skies and high solar input, the upward buoyancy force of the surface water will be sufficiently large to impede the downward mixing force and a period of summer stratification will begin. Under stratified conditions, the lower-density surface layer is called the epilimnion, whereas the higher-density deep layer is called the hypolimnion. The boundary between these layers is called the thermocline. Stratification persists until late fall when reduced solar input combined with heat lost from the epilimnion to the atmosphere decreases the buoyancy of the epilimnion. Eventually, the downward mixing force becomes greater than upward buoyancy force and the lake enters a period of fall turnover. Pit lakes situated in climates where the winter air temperature does not drop below 4°C, such as the North Island of New Zealand, will continuously circulate throughout the winter and restratify during the following spring. Lakes in colder climates will undergo a period of winter stratification once the temperature of the water column drops below 4°C, at which the maximum density of water occurs, and lower-density surface water overlies higher-density deep water. Winter stratification is supported by ice cover, which prevents wind energy from impacting on the lake surface. Turnover resumes in the spring following the breakup of ice cover, the exposure of the surface layer to wind energy, and the warming of surface water above 4°C. For further discussion on seasonal turnover, see works by Imberger and Patterson (1990) and Wetzel (2001).

Lake Morphology

The morphology of pit lakes differs from the morphology of natural lakes in two significant ways. First, pit lakes tend to be much deeper than natural lakes with equivalent surface area as shown in Figure 9.1. For this reason, the downward mixing force required to completely turn over pit lakes can be larger than the downward mixing force required to turn over natural lakes with similar surface areas. Second, the surfaces of pit lakes are surrounded by high pit walls that shelter the lake surface from wind (Figure 9.1). This reduces the wind velocity over the lake surface and the downward mixing force within the lake. As a result of these morphologic characteristics, pit lakes are generally thought to be more likely to develop permanent stratification than natural lakes (Lyons et al. 1994; Doyle and Runnells 1997; Castro and Moore 2000).

The relationship between the surface area and maximum depth of lakes was quantified by Hutchinson (1957) with the relative depth equation (z_r), which expresses the ratio of the maximum depth (z_m) to the mean diameter of the lake surface (d) as a percentage:

$$z_r = \frac{z_m}{d} \cdot 100\% \qquad \text{(EQ 9.1)}$$

Assuming that the lake surface has a circular geometry and the mean radius of the lake is equal to half the mean diameter, d is calculated from the surface area of the lake (A_0) using the equation for the area of a circle:

$$A_0 = \pi \left(\frac{d}{2}\right)^2 \qquad \text{(EQ 9.2)}$$

Rearrangement and substitution of Equation 9.2 into Equation 9.1 yields the standard equation for z_r (%):

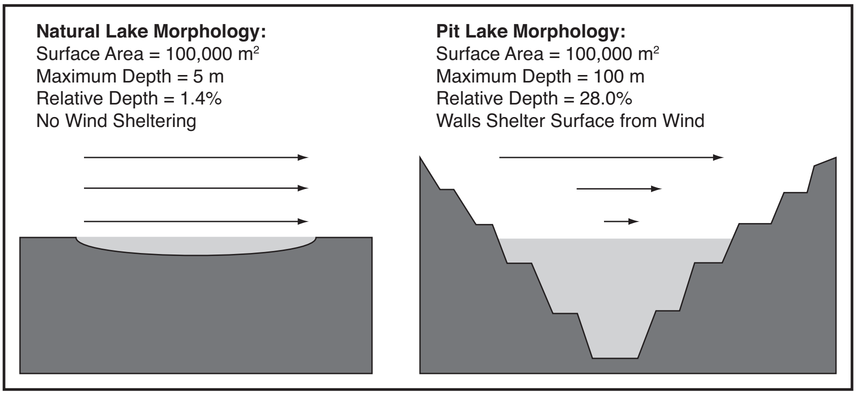

FIGURE 9.1 Comparison between the morphology of a natural lake and the morphology of a pit lake with equal surface area (not drawn to scale). Arrows represent the magnitude of wind velocity above the lake surface.

$$z_r = \frac{50 \cdot z_m \cdot \sqrt{\pi}}{\sqrt{A_0}} \quad \text{(EQ 9.3)}$$

Most natural lakes have relative depths of 2% or less (Wetzel 2001), whereas some existing pit lakes have relative depths ranging from 1% to 45% (Table 9.1).

Although the morphology of pit lakes influences physical limnology, direct comparisons between the relative depth and physical limnology of existing pit lakes do not identify a consistent relationship between these two properties, as shown in Table 9.1. For example, some pit lakes with low relative depth, such as Lake Goitsche in Central Germany (z_r = 1%), exhibit permanent stratification (Boehrer et al. 2003), whereas other pit lakes with high relative depth, such as the Blowout lake in Utah, United States (z_r = 34%), undergo complete annual turnover (Castendyk and Jewell 2002). Although currently meromictic, the Berkeley lake in Butte, Montana, United States (z_r =32%–40%), has exhibited episodes of complete lake turnover in the past (Jonas 2000; Gammons and Duaime 2006). With such variability, the value of relative depth as a tool for predicting future limnology is limited.

Pit Wall Stability

Pit wall stability may be a contributing factor to turnover. Because pit lakes are surrounded by steep, highly fractured walls, mass wasting is a common process witnessed in existing pit lakes. In some cases, turbidity currents generated by slope failure have been strong enough to disrupt stratification and induce turnover. In December 1997, a pit wall collapse in the Summer Camp lake, Nevada, United States, affected redox conditions and concentrations of total dissolved solids at the chemocline near the bottom of the lake (Parshley and Bowell 2003). Similarly, a landslide event in October 1998 initiated lake mixing in the Berkeley pit lake, Montana (Madison et al. 2004). Pit wall stability is discussed in detail in Chapter 23.

Chemistry of Lake Inputs

The chemistry of lake inputs can also influence turnover in a pit lake. The density and buoyancy of water are more significantly affected by concentrations of total dissolved solids (TDS) or salinity than by temperature. This concentration effect is commonly observed where rivers discharge

TABLE 9.1 Comparison between the relative depth and physical limnology of 22 existing pit lakes

Pit Lake	Location	Maximum Depth, m	Surface Area, ha	Relative Depth, %	Physical Limnology	Conductivity, mS/cm	Total Dissolved Solids, mg/L	Reference
Goitsche	Central Germany	48	1,330	1	Meromictic	0.5–5.5		Boehrer et al. (2003)
Merseburg-Ost	Central Germany	20	200	1	Meromictic		6,500–80,000	B□hrer et al. (1998)
Lake 111	Lusatian, Germany	10.2	10.7	3	Meromictic	1.6–2.4		Karakas et al. (2003)
B-Zone	Saskatchewan, Canada	55	29	9	Holomictic			Doyle and Runnells (1997)
Sleeper	Nevada, United States	118	100	11	Holomictic	1.5–3.3		Atkin and Schrand (2000)
Boss	Nevada, United States	7	0.25	12	Holomictic		12,000	Atkins et al. (1997)
Waterline	British Columbia, Canada	40	5.9	15	Meromictic			Martin et al. (2003)
Yerington	Nevada, United States	109	31	17	Holomictic		631	Miller et al. 1996; Jewell and Castendyk (2002)
Blackhawk	Utah, United States	26	1.6	18	Holomictic	0.95	622	Castendyk and Jewell (2002)
Aurora	Nevada, United States	20	1.0	18	Holomictic		491	Price et al. (1995); Atkins et al. (1997)
D Pit	Saskatchewan, Canada	26	1.6	18	Holomictic			Doyle and Runnells (1997)
Udden	Northern Sweden	50	4.6	18	Meromictic	1.0–2.0		Ramstedt et al. (2003)
Copper Basin	Tennessee, United States	60	8.1	19	Meromictic	1.0–6.0	200–4,600	Colarusso et al. (2003)
Brenda	British Columbia, Canada	140	3	20	Meromictic	0.8–1.2		Stevens and Lawrence (1998)
Enterprise	Northern Territory, Australia	110	821	21	Holomictic	0.26–0.34		Boland and Padovan (2002)
Main Zone	British Columbia, Canada	120	25	21	Holomictic			Martin et al. (2003)
Island Copper	British Columbia, Canada	380	190	24	Meromictic			Fisher and Lawrence (2000)
Spenceville	California, United States	17	0.20	34	Meromictic	1.25–13.1		Levy et al. (2997)
Blowout	Utah, United States	71	3.4	35	Holomictic	1.0	799	Castendyk and Jewell (2002)
Gunnar	Saskatchewan, Canada	110	7.0	37	Meromictic	0.3–2.0	175–1,050	Tones (1982)
Berkeley	Montana, United States	242	29	40	Meromictic/Holomictic	4.1–7.1	2,000–5,000	Davis and Ashenberg (1989); Gammons and Duaime (2006)
East Sullivan	Quebec, Canada	106	4.3	45	Meromictic			Tass□ (2003)

Source: Adapted from Castendyk and Webster-Brown 2007.

into the ocean and produce a plume of fresh water that floats over seawater for some distance offshore. Differences between the salinity of input waters to a pit lake can similarly generate a chemical density gradient, called either a halocline or a chemocline. A chemocline may result from low-salinity surface water, such as rainwater, added above high-salinity lake water, or high-salinity groundwater discharging beneath low-salinity surface water. Given the mineralized nature of surface and groundwater at mine sites, pit lakes have a greater potential to receive waters with elevated salinity than natural lakes. A chemocline may also result from the downward settling of mineral precipitates (such as Fe and Mn oxides) through the water column and the dissolution of these species under reducing conditions near the bottom of the lake. Unlike thermoclines, chemoclines are not seasonally dependent. Therefore, a strong density gradient owing to salinity may persist throughout the year and inhibit the vertical mixing of bottom waters. Using these principles, the Island Copper lake in British Columbia, Canada, was engineered to be meromictic by flooding most of the empty pit with seawater and filling the topmost meters of the water column with fresh water (Fisher 2002). See Chapter 17 for further description of the Island Copper pit lake.

REVIEW OF PHYSICAL LIMNOLOGY MODELS APPLIED TO PIT LAKES

Numerical models of physical limnology, also called hydrodynamic models, integrate multiple processes influencing lake stratification and turnover to simulate or predict the physical limnology of a lake, and several studies have successfully simulated the physical limnology of existing pit lakes. The DYRESM by Imberger and Patterson (1981) has been used to model the Brenda lake in British Columbia (Hamblin et al. 1999), the Island Copper lake in British Columbia (Fisher 2002), the Dexter lake in Nevada (Balistrieri et al. 2006), and a coal mine lake in Western Australia (Ivey et al. 2006). Atkins et al. (1997) used CE-QUAL-W2 (Cole and Buchak 1995) to model the Yerington, Aurora, and Boss lakes in Nevada. Colarusso et al. (2003) also used CE-QUAL-W2 to model an acidic, stratified pit lake in southeast Tennessee (United States) that currently discharges low TDS water from the top of the water column to a downstream watershed. The objective of the modeling was to determine whether the lake was susceptible to turnover with the resultant potential to cause water degradation in downstream watersheds. Colarusso et al. (2003) determined that its current meromictic structure is stable against destabilizing effects of wind for simulated storms and should remain stable even if TDS levels were substantially increased.

Three studies have explored future limnologic conditions in existing pit lakes. Jewell and Castendyk (2002) discussed a 50-year prediction of circulation in Yerington lake using the 1-D Princeton Ocean Model. In an impressive, interdisciplinary study of two pit lakes in the abandoned Merseburg-Ost coal mine in central Germany, Böhrer et al. (1998) coupled a 1-D hydrodynamic model with a three-dimensional (3-D) groundwater model to create a 100-year prediction of pit lake water quality. Stevens and Lawrence (1997) predicted the effects of 40 years of subaqueous disposal of mine tailings in a pit lake based on variations in the coefficient of eddy diffusion.

Despite this body of research, the author is not aware of any published studies that predict the future limnology of a pit lake that currently does not exist apart from his own work on the Martha Au-Ag Mine, New Zealand, using DYRESM (Castendyk and Webster-Brown 2007). This probably indicates the level of uncertainty inherent in physical limnology predictions.

PREDICTING FUTURE LIMNOLOGY WITH DYRESM

Given the number of studies that have successfully modeled existing pit lakes using DYRESM and the free distribution of this software online (www.cwr.uwa.edu.au), the remainder of this paper describes how to use DYRESM to predict the physical limnology of a future pit lake. DYRESM assumes that vertical variations in temperature, salinity, and density have a significantly greater influence over lake circulation than horizontal variations. This allows DYRESM to model a lake as a series of vertically stacked layers. DYRESM uses the heat, mass, and momentum fluxes to individual layers to model the vertical temperature, salinity, and density structure of a lake over time (Antenucci and Imerito 2001).

The model requires six input data files: meteorological, inflow, outflow, morphology, initial profile, and parameters. The meteorological data file specifies daily values for short-wave radiation, long-wave radiation, air temperature, vapor pressure, precipitation, and wind speed. These conditions are applied to the surface layer only, from which heat, mass, and momentum are distributed to the lower lake layers. Historic meteorological data must be obtained from the closest meteorological station to the open pit mine. Ideally the mining company will have a meteorological station on-site with several years of data available. Because DYRESM uses daily time steps, the meteorological data file will be very large, especially for a model of 10 years or more. If input values for long-wave radiation are not available, these data can be estimated from global long-wave radiation output. Average meteorological conditions can be used in the meteorological data file to produce a model of average limnologic conditions; however, it is also valuable to understand how extreme weather events influence turnover, especially for pit lakes that are meromictic under average conditions. If a limited meteorological data set is available, the available data set can be repeated to produce a model of the desired duration, recognizing that this will reduce the natural variability reflected in the results.

The *inflow* data file specifies the volume, temperature, and salinity of each lake input, with the exception of rainwater landing on the lake surface, which DYRESM automatically calculates from the meteorological data file. The volume of daily pit wall runoff (i.e., rainwater landing inside the lake catchment that flows over the pit walls to the lake surface) can be estimated using the rational equation:

$$Q = CIA \qquad \text{(EQ 9.4)}$$

where Q is the daily runoff inflow volume; C is the runoff coefficient, which accounts for the amount of rainwater that infiltrates into the ground; I is the daily precipitation rate; and A is the open pit mine catchment area excluding the lake surface (see Fetter 2001). The modeler specifies that daily runoff is added to the surface of the lake. If additional surface water will be added to the lake from a river or stream, the daily volume should be estimated from the lake water balance and added to the lake surface.

The modeler estimates the volume of daily groundwater input from the hydrologic model for the pit lake. From these data, the modeler specifies the depths where groundwater is added to the lake, the volume flux of groundwater inputs, the salinity and temperature of groundwater inputs, and the time when inputs become active during lake filling. Groundwater may be added from a single point at the bottom of the lake to reflect the contribution of a deep, vertical mine shaft that intersects the bottom of the pit (Castendyk and Webster-Brown 2007) or may be distributed evenly along the walls of the pit to represent diffuse groundwater input (Ivey et al. 2006).

The temperature of runoff and surface water inputs can be estimated from the daily air temperature; however, groundwater temperature should be measured in the field, particularly for open pits excavated above historic underground mines. These pit lakes may receive warm groundwater discharging from deep, vertical mine shafts that conduct deep groundwater heated by the local geothermal gradient to the lake. For example, Gammons et al. (2006) discuss the elevated temperature of flooded underground mine workings in Butte, Montana, and Gammons and Duaime (2006) relate the long-term increase in the temperature of the monimolimnion in Berkeley lake to these warm groundwater inputs. It is likely that groundwater will be warmer than hypolimnion and monimolimnion waters in most pit lakes. Warm, low-salinity groundwater inputs may contribute to vertical circulation, whereas high-salinity groundwater inputs may contribute to stratification regardless of elevated temperatures. Castendyk and Webster-Brown (2007) demonstrate the sensitivity of whole-lake turnover to small changes in the salinity and temperature of groundwater inputs.

DYRESM requires a salinity value for each lake input expressed in practical salinity units (psu). Whereas geochemists define salinity by the total quantity of dissolved salts in water measured by weight in parts per thousand, physical limnologists define salinity by the ratio of the electrical conductivity of a water sample divided by the electrical conductivity of a seawater standard that has a salinity of 35 psu at a temperature of 15°C under atmospheric pressure (UNESCO 1976). For limnology predictions, the salinity of each lake input is calculated from field measurements of electrical conductivity and temperature using the UNESCO International Equation of State for Seawater described by Fofonoff (1985). Online calculators are available that simplify this conversion (http://ioc.unesco.org/oceanteacher/resourcekit/M3/Converters/).

The *outflow* data file specifies the depth of lake outlets and the outflow rate over time. The potential outflows from a pit lake include groundwater outflow, surface outflow, and evaporation, which DYRESM automatically calculates from the meteorological data file. During lake filling, groundwater outflow will typically be zero, owing to the direction of the hydrologic gradient surrounding the open pit (see Chapter 1). For pits that intersect historic mine workings, surface water and runoff added to the pit may initially drain into dry mine tunnels below the pit, in which case long-term limnologic modeling should begin after input volumes exceed output volumes and a permanent water body begins to develop in the pit. At any time during lake filling, groundwater outflow can be activated in the outflow data file using rates from the hydrologic model. Specifying the height of an outflow point, such as a specific mine tunnel, is relatively straightforward; however, determining when the outflow point becomes active as well as determining the outflow rate over time requires close integration of the hydrologic and limnologic models. The net daily groundwater outflow volume can also be distributed between multiple points evenly spaced with depth to represent diffuse groundwater outflow, as demonstrated by Ivey et al. (2006). Owing to pit lake morphology, shallow lake layers are in contact with more wall rock area than deep lake layers. For this reason, groundwater outflow volumes should be proportional to lake depth, whereby shallow lake layers output more groundwater than deep layers. Finally, the modeler specifies a lake overflow elevation in the outflow file, which DYRESM uses to calculate the daily surface water outflow volume once the lake reaches its steady-state elevation.

The *morphology* data file defines the geometry of the lake as a stack of layers, each layer having a specified area and thickness. Layer dimensions can be measured from mine plans that show the expected topography of the open pit upon mine closure and the surface elevation of the future lake. Pit wall benches provide useful intervals for defining layer thicknesses. The area of each layer can be measured by hand or by using mapping software.

Because DYRESM was designed to model existing lakes, it is incapable of modeling conditions at the moment of mine closure when the open pit is dry. The temperature and salinity data of an initial pool at least 15 m deep must be specified in the *initial profile* data file. Sensitivity analyses performed by the author have shown that the temperature and salinity of the initial profile did not affect the limnology predicted for a 194-m-deep pit lake; however, this may not apply to shallower pit lakes.

The *parameters* data file specifies the values of constants used in the model and the default values are recommended (Antenucci and Imerito 2001). However, the light extinction coefficient should be adjusted to reflect the anticipated turbidity of the future pit lake. This parameter controls the adsorption of solar radiation with depth and the heating of lake water, and will be different for clear lake water than for cloudy lake water. In general, light will penetrate deeper in nutrient-poor, acidic pit lakes than in nutrient-rich, neutral-alkaline pit lakes because of less biological activity and greater water clarity.

Examples of DYRESM Results

Figure 9.2 shows the temperature profile for the Blowout pit lake in southwest Utah from 1997 to 1998, modeled with DYRESM version 2.5.0, above observed temperatures for the same period reported by Castendyk and Jewell (2002). The pit lake had existed for roughly 30 years prior to data collection such that the elevation of the lake surface was mostly constant. Both data sets show summer stratification beginning in April and continuing through December with a maximum epilimnion depth between 10 and 20 m. Summer surface temperatures exceeded 20°C. Both modeled and observed data show uniform temperatures with depth corresponding to complete lake turnover between late December and early April, which characterizes this lake as holomictic. For existing pit lakes, such comparisons between modeled and observed data help to calibrate and validate models, and to predict conditions occurring between observations.

Figure 9.3 shows a 6-year prediction of temperature versus surface height for a future pit lake at the active Martha mine, New Zealand, modeled with DYRESM version 2.5.0. Owing to the diversion of surface water into the empty pit at the conclusion of mining, the surface height rises rapidly from 15 m, the height of the initial profile, to 194 m, the elevation of the surface overflow weir, in 4.5 years. After this elevation is achieved, surface water is no longer diverted into the lake. Thermal stratification develops each summer with an epilimnion extending to approximately 20 m. Partial turnover of the upper two-thirds of the water occurs each winter. Owing to the density contrast between low-salinity surface water inputs and warm, high-salinity groundwater inputs, DYRESM predicted that a warm, high-density layer will develop in the bottom third of the water column that does not mix with overlying layers during annual turnover events. This is an example of a meromictic prediction for a future pit lake.

Uncertainties in DYRESM Predictions

Like every numerical simulation of natural conditions, many assumptions are incorporated into DYRESM that must be considered when interpreting the accuracy of model results. The following points identify some significant assumptions used by DYRESM, the uncertainty or limitations associated with each assumption, and strategies that can be used to improve the accuracy of predictions. Table 9.2 provides additional assumptions.

- **Assumption:** Meteorological conditions inside the open pit are equal to conditions recorded at the meteorological station. **Uncertainty:** Meteorological stations are typically situated outside the open pit mining area so that equipment does not interfere with mining. As a result, recorded meteorological conditions will differ from actual conditions inside the pit, particularly reductions in wind velocity and solar radiation caused by the sheltering and shading effects of the pit walls. The presence of a lake may also influence local meteorological conditions in the future. **Comment:** Analytical equations are

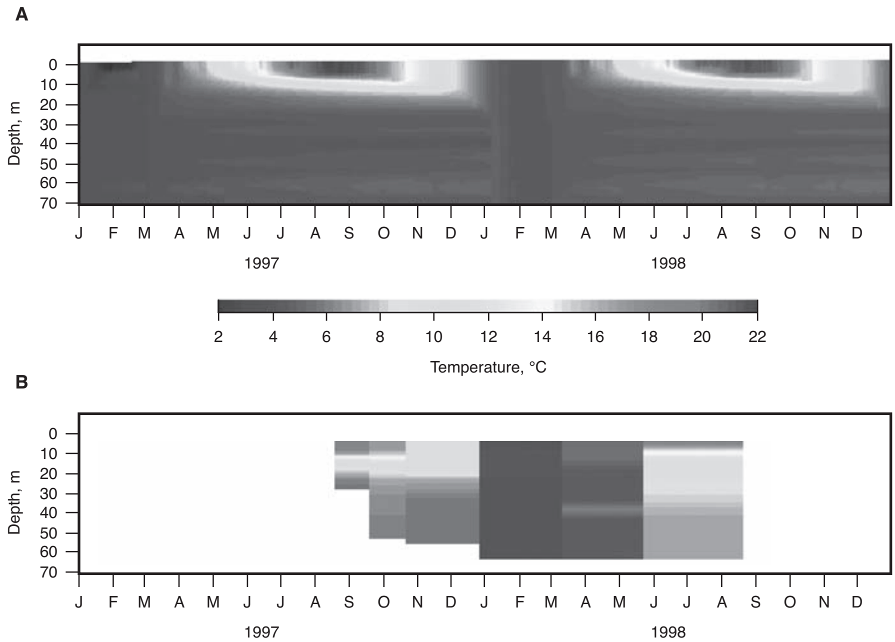

FIGURE 9.2 Modeled (a) and observed (b) temperature profiles for the holomictic Blowout pit lake, Utah, between 1997 and 1998, generated with DYRESM

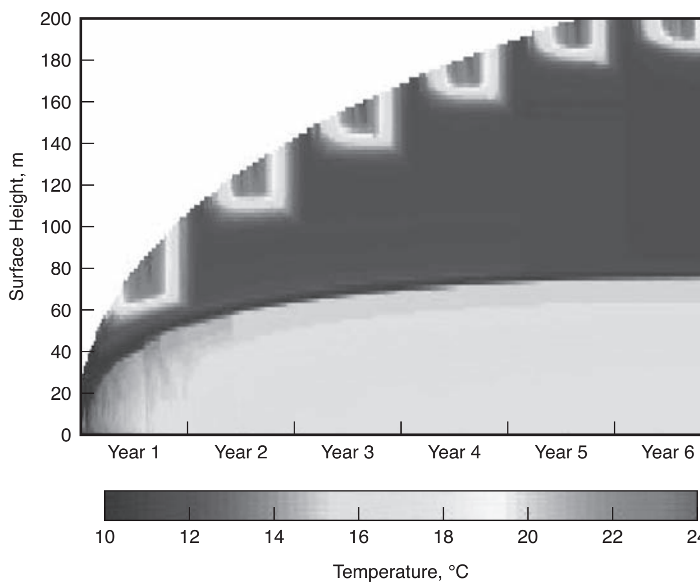

FIGURE 9.3 DYRESM prediction of the temperature profile of a future meromictic pit lake at the Martha mine, New Zealand

TABLE 9.2 Assumptions and uncertainties associated with DYRESM predictions of future pit lakes

DYRESM Assumption	Uncertainty	Comment
Meteorological conditions inside the open pit are equal to conditions recorded at the meteorological station.	Meteorological stations are typically situated outside the pit in open areas and recorded conditions will differ from actual conditions in the pit, notably wind velocity and solar radiation, owing to sheltering and shading from the pit walls.	Analytical equations are available to adjust wind velocity and short-wave radiation data prior to modeling.
Ice never covers the lake.	Ice may cover lake during winter months if air temperatures drop below 0°C.	DYRESM version 2.5.0 discussed herein does not allow for ice cover. The Centre for Water Research plans to release a new version of DYRESM in December 2008 that models ice cover (Jason Antenucci, personal communication, 2008).
The historic meteorological record accurately reflects future conditions.	Surface temperatures in many regions are predicted to increase over the next 40 years, which will increase evaporation rates (Hoerling and Eischeid 2007).	The meteorological data file may need to be based on predicted meteorological conditions. Short-term (10-year) predictions may be more accurate than century-long predictions.
The density of each lake layer is constant across the full spatial extent of the layer. This allows a 1-D assumption to apply.	Spatial changes in density can occur across layers, notably along the shoreline where high-density pit wall runoff or landslides may generate high-density currents that travel downward along the walls of the pit.	A 2-D or 3-D model may be needed. However, DYRESM has successfully modeled the movement of vertical plumes in a pit lake water column (see text).
An accurate light extinction coefficient is known prior to lake development.	Several physical, chemical, and biochemical processes affect turbidity such as pit wall erosion, precipitation of colloidal iron hydroxide minerals, and microbial productivity. The extent of these processes and their effect on water turbidity may be unknown prior to lake development.	Geochemical models can indicate the quantity of iron oxide likely to precipitate from lake water. Sensitivity analyses can be performed on the light extinction coefficient to explore the effects of variable turbidity on the predicted limnology.
The morphology of the final open pit is known.	Mine plans can change. Mining companies may expand the original pit design or terminate operations sooner than expected depending on the market value of the ore.	Changes that modify the expected depth, volume, or surface area of the pit lake warrant remodeling of the limnologic prediction.
The water balance of the future pit lake is known.	The water balance relies on accurate knowledge of the open pit morphology, an accurate hydrologic prediction, plus knowledge of the closure plans that dictate controls on future lake surface elevation and may involve flooding the lake with surface water. Each of these factors may change if mine plans change.	All assumptions that are incorporated into the hydrologic model apply to the water balance and the limnologic prediction. Input data must be accurate to achieve the most accurate prediction.
The chemistry, temperature, elevation, and timing of all lake inputs are known in advance of lake filling.	The temperature and chemistry of lake inputs may be unknown or may change during lake filling. The elevation and inflow volume of groundwater will change during lake filling as the water table rebounds. These factors can determine whether a lake becomes holomictic to meromictic, owing to the formation of dense water layers.	Integration of hydrologic and geochemical predictions with limnologic predictions can improve the accuracy of limnologic predictions.
The 15-m-deep initial profile does not affect final limnology.	For shallow pit lakes (<30 m) the initial profile will constitute half the depth of the pit lake, and the initial profile may influence physical limnology.	Shallow pit lakes may require a limnologic model better designed to accommodate lake filling than DYRESM.
Vertical mixing only occurs when two adjacent water layers attain equal density.	Stevens and Lawrence (1997) demonstrated that eddy diffusion across the chemocline can result in significant mass transfer from monimolimnion to hypolimnion over a 50-year time scale.	Stevens et al. (2005) provide equations to evaluate turbulence occurring below the chemocline of strongly stratified pit lakes. This turbulence results from the unique morphology of pit lakes.

available to adjust wind velocity data in the meteorological data file, as described by Ivey et al. (2006). The number of daylight hours received by the lake surface can be corrected by adjusting the specified latitude of the pit to compensate for light shading from pit walls (David Hamilton, personal communication, 2003); however, the effect of shading will decrease as the lake fills over time.

- **Assumption:** The density of each lake layer is constant across the full spatial extent of the layer, which allows a 1-D assumption to apply. **Uncertainty:** Spatial changes in density can occur across layers in pit lakes, notably along the shoreline where high-salinity pit wall runoff may be added to the lake during rainstorms and where large volumes of rock may be instantaneously added to the lake surface in the event of a slope failure. Both can create high-density currents that travel downward along the walls of the pit, which disrupt 1-D conditions and possibly invalidate the model. **Comment:** Other limnologic programs model two-dimensional or 2-D (i.e., CE-QUAL-W2) and 3-D (i.e., CWR-ELCOM) conditions and may be more appropriate than DYRESM to simulate these processes (Cole and Wells 2008; Laval and Hodges 2000). However, DYRESM has successfully modeled the vertical movement of low-density plumes through the water column of the Island Copper pit lake in British Columbia (Fisher and Lawrence 2000; Fisher 2002) and may still be appropriate.
- **Assumption:** The chemistry, temperature, elevation, and timing of all lake inputs are known in advance of lake filling. **Uncertainty:** Small changes to the temperature and/or salinity of high-volume lake inputs like groundwater can change a limnologic prediction from holomictic to meromictic (Castendyk and Webster-Brown 2007). The limnology of pit lakes that are flooded with a combination of surface water and groundwater are particularly sensitive to differences in the density of input water as observed at the Island Copper pit lake in British Columbia (Fisher and Lawrence 2000; Fisher 2002) and Lake Goitsche in Germany (Boehrer et al. 2003). Because the chemistry of mine-impacted water can vary over time as wall rock weathers, the salinity of lake inputs will also vary over time. Additional variability comes from the physical characteristics of the groundwater inputs. Whereas surface water, pit wall runoff, and rainwater are added to the lake surface, the elevation and inflow volume of groundwater will change over time as the water table rebounds. **Comment:** The accuracy of a limnologic prediction may be improved by integrating limnologic model inputs with hydrologic and geochemical model outputs, as demonstrated by Böhrer et al. (1998), who provided one of the most comprehensive predictions of pit lake limnology available.

CONCLUSIONS

Predicting whether a pit lake will be holomictic or meromictic is not a simple task. Predictions based only on comparisons between the lake surface area and maximum depth (i.e., relative depth) oversimplify the physical limnology, and data from existing pit lakes do not show a consistent relationship between morphology and observed limnologic behavior. Numerical models have accurately simulated the limnology of existing pit lakes and show promise for predicting the limnology of future pit lakes. These models are data rich, and the accuracy of limnologic predictions depends on the validity of input data to conditions inside the open pit at the time of mine closure. Perhaps because of the uncertainties involved, few limnologic predictions of future pit lakes have been published. The absence of predictions from the literature represents the largest data gap in the field of pit lake limnology. Preclosure predictions that are compared to field observations collected after hydrologic equilibrium has been achieved will improve our understanding of the strengths and weaknesses of these models as well as our ability to predict the physical limnology of pit lakes in advance of mine closure.

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CHAPTER 10

Approaches for Developing Predictive Models of Pit Lake Geochemistry and Water Quality

W.M. Schafer and L.E. Eary

INTRODUCTION

Water quality in pit lakes is often a pivotal issue affecting mine permit approval and closure planning. Owing to the complexity of water–rock interactions, limnological processes, and climatic and hydrological factors that influence pit lakes, predicting their water quality evolution is a complex undertaking. Pit lakes containing poor-quality water such as the Berkeley pit in Butte, Montana, United States, are often viewed as the most tangible adverse effect of mining, which heightens the public awareness of pit lake water quality. Since mining ended in 1982, the Berkeley pit has filled to a depth of 275 m with 140,000 ML of acidic water with a pH of less than 3 and 180 mg/L copper (Gammons and Duaime 2006).

If adverse pit water quality poses a risk to adjacent water bodies or ecological receptors, water treatment or other costly mitigation measures may be required. As a consequence, various approaches have been developed to predict water quality in pit lakes. It is important to recognize that predictive models of hydrochemistry are most effectively used in the hydrologic sciences when they can be validated against measured hydrologic and chemical responses. Model validation provides continuous feedback that can be used to refine a model and improve its predictive accuracy. Because pit lakes may require many tens or hundreds of years after mine closure to fill with water, there are limited opportunities to validate pit lake models. Model results, especially for trace chemical constituents, are inherently subject to large uncertainties (Maest et al. 2005). The uncertainty of pit lake models does not invalidate their use but needs to be considered when making policy decisions on the basis of model results.

The goal of most modelers is to "build a better model" that improves predictive accuracy and reduces or at least better defines model uncertainty. As a result, pit lake modeling is constantly evolving. Consequently, there is no standardized approach that is suitable for every pit lake model. Different modeling steps will be mandated for greenfield mines, mine expansion projects, mines at the verge of closure, and existing pit lakes. Some of the challenges faced by pit lake modelers that dictate the modeling approach include

- The extent and availability of hydrologic and geochemical data that can be used to define model inputs and formulations of water and chemical balances;
- Determining how to simulate solid-phase chemical precipitation, sorption, and other processes that may affect solubility;
- Simulating water–rock interactions;

- Understanding the difference between open and closed hydrologic systems and how they affect water quality evolution;
- Assessing how the physical limnology of the pit lake will affect seasonal variation in temperature, density, and convective mixing, which in turn influence gas partial pressures;
- Predicting future climatic conditions and accounting for the influence of dry or wet conditions on filling rate, and chemistry;
- Accounting for uncertainty in model parameter estimation and its effect on water quality predictions;
- Creating models with the flexibility to enable simulation of various mitigation strategies, including selective mining, backfill placement, water management (e.g., pumping water from other mine and processing facilities into the pit lake for rapid filling, pumping water out of the pit lake for a beneficial or consumptive use); and
- Understanding the dynamic interactions between the inorganic geochemical system and the biological system (e.g., metals forms and ecotoxicity, sequestration of metals by algae, organic carbon production and decomposition and its effect on gas partial pressures).

APPROACH

There is no universally accepted pit lake water quality model. The techniques used to develop a predictive tool depend on the nature of the system being simulated, the stage of mine development, and the nature of the decisions to be made on the basis of the model results. The first step in developing a pit lake model is to assess how the model will be used to support decision making. If a pit lake model is developed for a mine project, there is a decision to be made that may be influenced by the expected pit lake chemistry. The decision framework (Figure 10.1), of which the pit lake model is a part, should contain some feedback mechanism so that the potential effects of changes in the mine operating plan in application of mitigation actions can be quantified in terms of pit lake chemical response.

The next step in developing a water quality predictive tool is formulation of a conceptual model of the pit lake model (Figure 10.2). Regional climate is usually considered the primary external factor that "drives" pit lake hydrology. Rainfall ultimately determines the rate of inflow from direct precipitation, groundwater, and surface water pathways. Additionally, temperature

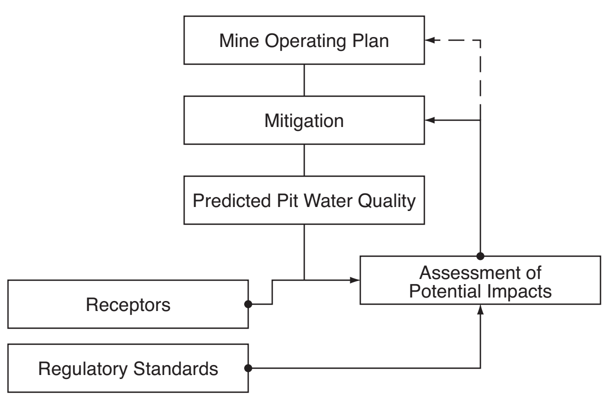
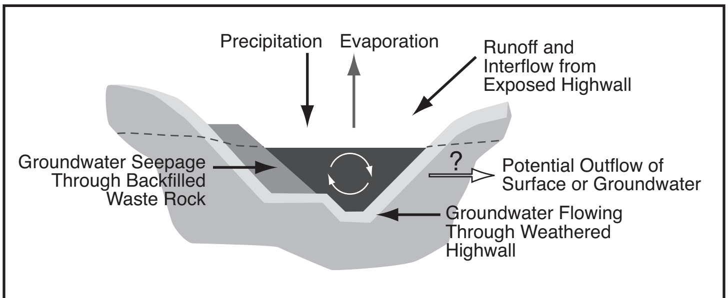


FIGURE 10.1 Use of pit lake model results to support mine project design and decision making



Constraints	Description
Climate	Determines precipitation, runoff, and evaporation
Pit area, depth, and volume	Governs fill rate
Rock geochemistry	Affects chemistry of input waters
Limnology of pit lake	Influences dissolved gas levels, mixing/stratification, and redox conditions

FIGURE 10.2 Conceptual model of factors affecting pit lake hydrology and geochemistry

affects the density of the water column, and along with wind, largely determines the tendency for a pit lake to mix. Temperature along with wind speed and relative humidity affect the evaporation rate.

Water quality in a pit lake is determined by the proportional amount and chemistry of each type of water that flows into the pit lake in combination with in-lake hydrochemical processes that may add, subtract, or redistribute chemical species in the water column. Additionally, the amount of water lost from the pit lake system by either evaporation or outflow into surface or groundwater also affects water quality. Finally, chemical conditions at various depths in the pit lake that are controlled by limnological processes will ultimately dictate the levels of dissolved oxygen, carbon dioxide, and the amount of carbon available for decomposition. These factors also have an important influence on pit lakes. Finally, nutrient levels along with water quality play an important role in determining the pit lake's biological productivity, which also affects carbon available for decomposition.

Contributions to the water balance of a pit lake include precipitation, surface runoff and interflow, and groundwater flowing through weathered rock in the highwall or in waste rock placed as backfill in the pit. The proportional amount and chemistry of the influent waters are critical elements that affect water quality. Evaporation will tend to increase the concentrations of constituents in the pit lake. If an outflow occurs from the pit lake, (e.g., it is part of an externally drained hydrologic basin), a steady-state geochemical equilibrium will develop in the pit lake at some point after filling is complete. If no outflow occurs from the pit lake (e.g., it is an endorheic basin from which no outflow occurs), the lake will continuously increase in salinity through time similar to many lakes in desert regions (e.g., Pyramid Lake in Nevada and Great Salt Lake in Utah, both in the United States).

Figures 10.3 and 10.4 illustrate the distinction between open and closed basin pit lakes and the importance of outflow quantity on equilibrium chemistry. Additionally, these figures illustrate the value of simple mass load type pit lake models (e.g., spreadsheet models) in evaluating long-term pit lake chemical evolution. A simple spreadsheet was used for this illustration to estimate total dissolved solids (TDS) in a pit lake for cases in which the amount of

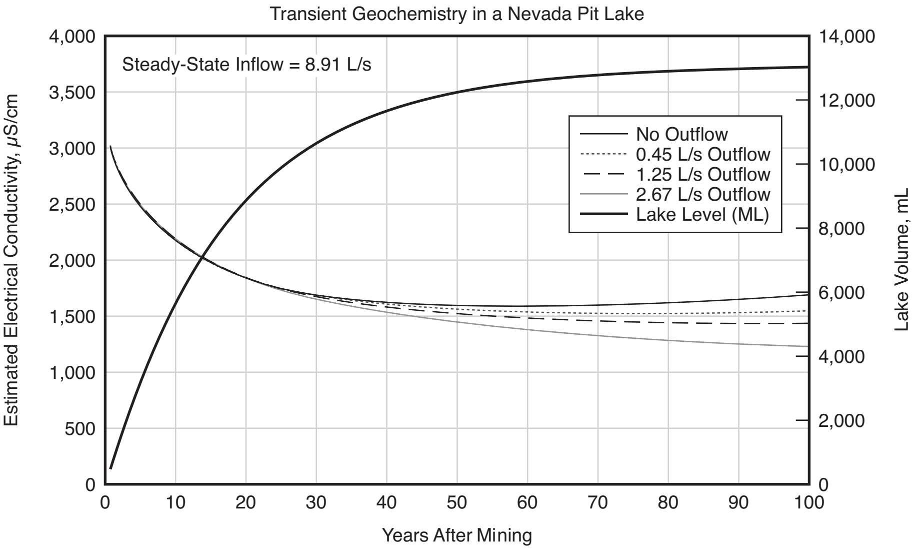


FIGURE 10.3 Pit lake filling rate and water quality assuming four cases for mine outflow (0%, 5%, 12%, and 30% of inflow rate)

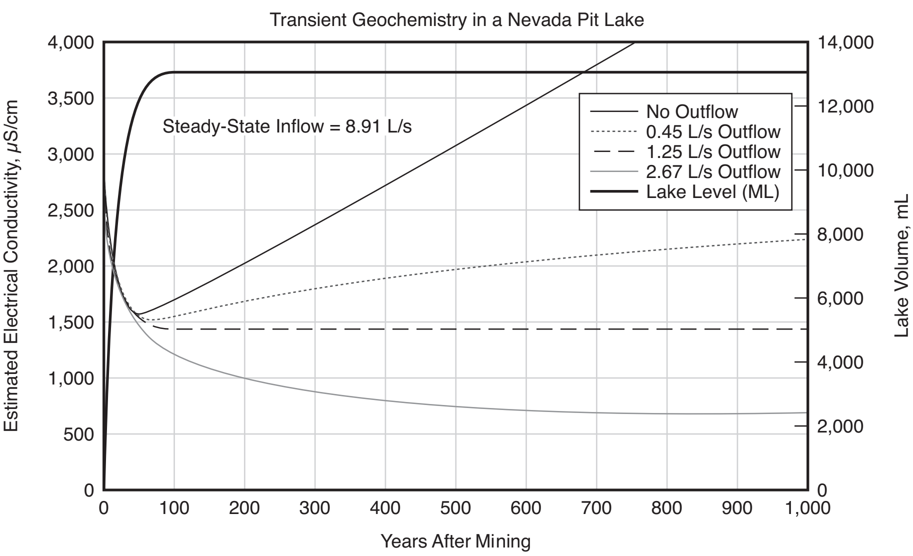


FIGURE 10.4 Pit lake filling rate and water quality assuming four cases for mine outflow (0%, 5%, 12%, and 30% of inflow rate)—results shown through 1,000 years

groundwater outflow varied from 0 to 2.67 L/s. The steady-state pit lake inflow was 8.91 L/s (27.5 L/s inflow during early stages of filling). The model assumed that combined inflows had an initial TDS of 3,000 mg/L, which decreased to 200 mg/L over about 30 years. Based on pit geometry and evaporation rate used in the model, the pit lake reached hydrologic steady

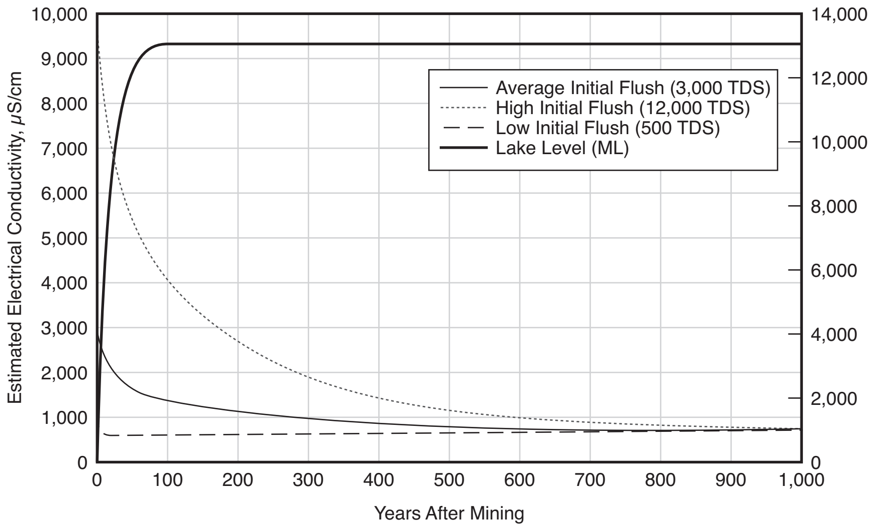


FIGURE 10.5 Pit lake filling rate and water quality assuming three differing cases for initial mass load contributions from weathered rock

state in about 30 years. At this stage, water quality was similar for all cases. The TDS ranged from about 1,250 to 1,650 mg/L TDS, depending on the amount of outflow (Figure 10.3).

When the water quality results are extrapolated for 1,000 years (Figure 10.4), the water quality for the three cases are strongly divergent, varying from about 700 mg/L TDS for the highest amount of outflow to nearly 2,200 mg/L for the low outflow case. The high and moderate outflow cases were approaching geochemical steady state after 1,000 years, but the TDS in the low outflow case was still gradually increasing. There is no steady-state geochemistry for the closed basin example (no outflow), as the calculated TDS (at 5,000 mg/L after 1,000 years) continuously increases through time.

Simple mass load models such as these can provide reliable estimates of TDS, chloride, sodium, and magnesium concentrations for TDS levels less than about 10,000 mg/L, and are reliable for TDS and sulfate when TDS is less than about 2,000 to 3,000 mg/L.

These models address another common misconception about pit lake chemistry. Often, pit lake chemistry is assumed to reach equilibrium at the same time that so-called hydrologic equilibrium occurs (e.g., when the pit water level stabilizes). As Figures 10.3 and 10.4 show, the time scale for geochemical equilibrium may differ greatly from conditions at hydrologic steady state and may require more than 10 times as long to occur, if geochemical equilibrium occurs at all.

The same mass load model also illustrates the importance that initial chemical mass loads may play in transient pit lake water quality. In many pit lake systems, water inputs to the pit lake during early stages of filling are predicted to have much poorer water quality than long-term conditions. This is because of the rapid rinse-off of soluble salts that occurs as weathered rock (and especially backfilled rock) is inundated. The three cases shown in Figure 10.5 differ only in regard to the "first flush" inflow chemistry, which persists about 20 years. The case with higher mass loads during the first flush remain significantly more saline for about 500 years, although the equilibrium geochemistry is the same for all cases since the long-term water quality of inflow is the same in all cases.

DATA NEEDS FOR EACH CHEMICAL MASS INPUT AND LOSS

The data needed for simulating pit lake water quality depends on the type of simulation to be performed. There are two fundamental model formulations, deterministic and probabilistic, that affect how input data are selected. In a deterministic model, a single value is input for each input parameter and there is a single model result. A range of possible values are used to describe some or all input variables in a probabilistic model. As a result, model results consist of a range of values as well, with an associated probability of occurrence.

There are also two potential ways of assessing the variation in water quality through time. In a steady-state model, only the long-term final water quality is predicted. However, owing to the very long time frames associated with reaching chemical steady state, this approach is seldom if ever used for pit lake models. More often, transient models, which assess changes in water quality over time, are used.

Development of a pit lake model consists of two distinct components, a hydrologic or water balance component and a water chemistry component. A variety of methods have been used to estimate the hydrologic components of a pit lake, although MODFLOW or other groundwater models are most frequently used for this purpose (see Chapter 8).

The first step in implementing a hydrologic model for a pit lake is developing a relationship between pit elevation, area, and volume. These relationships can be modeled using look-up tables or polynomial equations are usually derived from the mine operating plan.

Climate

Development of a climatic data set is a particularly important step in pit lake modeling. Often, an annual time series of rainfall (and snowfall) and evaporation data are compiled based on historical climatic records. The "average" climate data set is then used to simulate the pit lake hydrology, including groundwater fluxes, surface water inflows, precipitation onto the lake surface, and evaporation. The problem with using averages to depict future climate is that most hydrologic processes do not have a linear relationship to rainfall. For example, in arid areas, a disproportionate amount of groundwater recharge and runoff occur in wetter-than-average years. As a result, a water balance that is based on average rainfall will often underestimate groundwater or surface water fluxes. Use of synthetic climate data that retains typical year-to-year variations in rainfall provides more accurate simulation of hydrologic behavior. Use of several model iterations using independent synthetic climate data sets removes the artifacts that abnormally wet or dry years may impose on the model results in a particular year.

Evaporation

Evaporation is an important process affecting pit lakes and has a unique effect on water quality because evaporation removes water without removing chemical mass. As a result, evaporation often induces chemical precipitates to form in a pit lake, increases chemical concentrations, or both. Sometimes evaporation can cause a reduction in ion concentration. For example, if gypsum ($CaSO_4{\cdot}2H_2O$) precipitates, an equal amount of calcium and sulfate ions are removed from solution. If the initial sulfate concentration is higher than that of calcium (in molarity) and gypsum forms, then evaporation will tend to increase sulfate concentration while decreasing calcium. This is because as gypsum forms, the product of calcium and sulfate activity remains constant. In addition, depending on the ratio of calcium to alkalinity in the pit lake, evaporation will result in long-term evolution to either a near-neutral pH, Ca-Na-($\pm SO_4$)-($\pm$Cl), or an alkaline (pH > 9) Na-HCO_3-CO_3-($\pm SO_4$)-($\pm$Cl)-dominated solution (Eary 1998).

Daily evaporation can be estimated from climatic data using a variety of techniques (Allen et al. 1998). Modelers must be careful to distinguish between pan evaporation, potential evaporation, or potential evapotranspiration. Evaporation from a class A pan is often used as a reference evaporation rate in North America. Pan evaporation differs from lake surface evaporation owing to temperature, boundary layer, and turbulence effects. A commonly used conversion is

$$\text{lake evaporation} = (0.7) \cdot (\text{class A pan evaporation}) \qquad \text{(EQ 10.1)}$$

In reality, the value of the coefficient (0.7 in Equation 10.1) varies by climate and time of year. Some hydrologic models compute potential transpiration from climatic data, and this reference evaporation rate is generally computed for a grass-covered field with ample water, which is similar to the lake evaporation rate.

Water–Rock Interaction

Water–rock interactions affect the quality of water that flows into a pit lake by groundwater or surface water pathways. Simulating the interaction of runoff water or groundwater with mineralized rock is the most challenging aspect of pit lake modeling, especially because it may depend on time, mineralogy, climate, and hydrology. Typically, the release of common ions and metals through this process accounts for the majority of the chemical mass (or at least of metals) in mine pit lake systems, yet it is among the most poorly understood and most difficult-to-quantify processes.

Accounting for water–rock interactions usually consists of two steps. The first is to characterize the degree of mineralization of exposed rock, and the second is to quantify the rate or degree of chemical reaction that is occurring in the rock. Mineralization is usually measured using static test methods, wherein the abundance of potentially acid-generating sulfides is compared to the acid-neutralizing minerals, principally carbonates (Sobek et al. 1978). In theory, rock with a negative net neutralization potential will become acidic assuming that all sulfides oxidize according to the reaction pathway on which the tests are predicated. Similarly, rock with a positive net neutralization potential should remain neutral to alkaline. The behavior of rock samples in the field is far more complex owing to a number of complications such as incomplete reaction, formation of coatings, dissimilar kinetics, noncarbonate buffering, and non-acid-generating sulfides (e.g., galena). Despite these qualifications, when used in the context of site-specific conditions and mineralogy, static test data provide a valuable means of assessing the acidification potential of rock.

The data needed to simulate pit lake chemistry are far more involved than merely determining acidification potential, however. While the acid-generating rocks exposed in a highwall, when present, are likely to contribute the majority of metals and sulfate, the concentrations of metals and common ions in pore water through time are needed to simulate pit lake chemistry. The processes affecting geochemical reactivity of mineralized rocks can be difficult to represent in a predictive model because of feedbacks between reaction rates, which are dependent on solution composition, and solution compositions, which are affected by reaction rates. For this reason, a combination of empirical and theoretical formulations is often used to develop a predictive model.

A variety of geochemical tests yield the concentrations of metals and common ions in water in contact with rock. However, few of these methods can be used to directly simulate the water–rock interaction occurring in pit lakes. For example, humidity cell tests, meteoric water mobility procedures, synthetic precipitation leaching procedures (SPLPs), and shake flask tests all yield ion concentrations. However, no test adequately simulates water–rock interaction in mine pits if it is conducted on an unweathered sample. Ion concentrations are sensitive to differences in the water–rock ratio, especially for moderately to highly soluble ions (e.g., nitrate and metal ions in low-pH solutions). The water-to-rock ratio in an SPLP test is 20:1, while the water-to-rock ratio in backfilled waste rock may be two orders of magnitude lower (0.2:1). Humidity cells are often performed on fresh rock samples, so they cannot provide a direct indication of ion concentrations. Kinetic tests such as humidity cells and some column tests provide necessary information about reaction rates, however, that can be used to calculate the abundance of leachable ions.

Column tests conducted on weathered rock provide the most meaningful data for use in pit lake models. Results from properly designed and conducted column tests provide direct indication of pore water chemistry and also show how concentrations may vary as a function of cumulative leaching by groundwater. Column leach water quality data are usually compared to the volume of solution eluted from them. Typically, groundwater from the site is used to leach the columns. When mineralized rock is leached with groundwater, the resultant rinsing curves (which are similar to breakthrough curves) typically have three components: the initial flush, a declining limb, and a steady-state chemistry representative of long-term conditions (which will often approximate the chemistry of the added solution). Although many empirical curves can be used to fit column test data, one useful algorithm is shown in Equation 10.2:

$$\begin{aligned} C_{PV} &= C_0, \text{ for } PV \le 1 \\ C_{PV} &= C_{GW} + (C_0 - C_{GW}) * PV^{-n}, \text{ for } PV > 1 \end{aligned} \qquad \text{(EQ 10.2)}$$

where C_{PV} is the concentration at any point on the leaching curve, C_0 is the concentration in the initially eluted solution that typically has the highest concentration, PV is the number of cumulative pore volumes rinsed, C_{GW} is the chemistry of the rinsate solution, and n is an experimentally derived exponent varying from less than 1 for slowly soluble constituents to as high as 10 for more soluble ions.

When scaling test results from lab to field scale, the final required adjustment is to assess the amount of rock that has been subject to weathering. In highwall systems, oxygen diffuses into the rock mass in fractures, so the fracture volume and density affects the proportional volume of weathered rock. Additionally, the thickness of the oxidized shell must be determined. Many techniques have been used to predict the changes in the depth of oxidation through time, but the most commonly used method is the approximate analytical solution to the Davis–Ritchie shrinking core model of pyrite oxidation (Davis et al. 1986). The thickness of rock oxidized during a typical mine life can vary from a fraction of a meter to many tens of meters, depending on fracture abundance, sulfide content (high-sulfide rocks have thinner oxidized zones), and sulfide reaction rate.

A final complexity in pit lakes is the need to account for spatial variability in the exposed highwall. Fortunately, most mines have developed a tool, the mine block model, for tracking mineralized ore, which can also be used to keep track of the abundance of mineralized rock in the highwall.

MODELING TOOLS

A number of modeling approaches have been developed for combining the various effects of chemical and physical processes that occur in pit lakes into a predictive model of water quality. A detailed discussion of methods for integrating multidisciplinary numerical models is provided in Chapter 14. Briefly, these approaches include

- Spreadsheets or dynamic systems models for hydrologic and simple chemical mass balance;
- Spreadsheets or dynamic systems models for hydrologic and chemical mass balance plus geochemical models for chemical processes;
- Spreadsheets or dynamic systems models for hydrologic and chemical mass balance, plus geochemical models for chemical processes, plus limnologic models for water column circulation behavior; and
- Experimental simulation and interpretation with geochemical models.

Spreadsheets (e.g., Microsoft Excel) may be the most common approach to modeling a pit lake. Spreadsheets can be used to represent a pit lake as a simple batch reactor where the water quality is represented as a mixing process that is a function of the difference between the rates of chemical inflow from various hydrologic inputs (e.g., groundwater, precipitation, surface runoff, wall-rock flushing) and the rates of chemical outflow of various hydrologic sources (e.g., evaporation, loss to the groundwater system, spillover). Data for these various rates, when combined with water elevation–volume relationships for the pit, can be used to represent the chemical evolution of the pit lake with time through simple mixing calculations. Dynamic systems models, such as GoldSim (www.goldsim.com) and Stella (www.iseesystems.com) may also be used for constructing mixing models based on hydrologic and simple chemical mass balance.

The addition of a geochemical model to the mixing model of hydrologic and chemical mass balances provides the capabilities to represent the effects of chemical processes, such as solution and gas-phase equilibrium, reaction kinetics, redox (oxidation–reduction), and surface adsorption, for affecting water quality. The most commonly used geochemical models for pit lakes are PHREEQC (Parkhurst and Appelo 1999), MINTEQA2 (Allison et al. 1991), and EQ3/6 (Wolery and Daveler 1992). Under some circumstances, it is desirable to integrate the geochemical models directly with the mixing model, such as the direct coupling of PHREEQC with GoldSim (Eary 2006).

A third level of complexity that is increasingly common is the use of a limnologic model to predict the extent of circulation of the water column in the pit lake. Normally, the limnologic model is used separately to predict the frequency of water column overturn and mixing (see Chapter 9), and this information is used to guide the specification of redox-related parameters, such as dissolved oxygen, temperature, and nutrients in lake layers as a function of depth. Limnologic models most commonly used at the current time are DYRESM and DYRESM-CAEDYM (CWR 2006) and CE-QUAL-W2 (WQRG 2006).

DATA GAP IN MODELING APPROACHES

A major data gap for developing and applying numerical models of pit lake chemistry is the lack of data to compare to model predictions. Without such data to conduct validation studies of the model, it is impossible to determine the strengths and weaknesses of the models and levels uncertainty in predicted results (see Chapter 15). This major data gap could be filled by two types of studies. The first would be the development and distribution of a database of pit lake water quality and associated data on pit geology, hydrology, climate, biochemistry, and geochemistry. This database could be used to develop general trends in water quality that could be compared to model predictions so that when the prediction deviates from the observed trends, then it becomes apparent that either the model is incorrect or there is some specific hydrogeochemical reason for the deviance that needs to be explained. The second type of study is the assemblage of water quality data over time from monitored pit lakes to provide guides on how their chemical compositions change as the pits fill and then reach hydrologic equilibrium. This information would be particularly important for assessing the effects of wall-rock leaching during the initial stages of pit inundation, which is a process that can be very difficult to represent in pit lake predictive models. A comparison of the monitoring data on water quality and rate of filling to model predictions made either initially for mine permitting or for mine closure would also provide a measure of the reliability of the predictive models to represent the real systems.

LIST OF EXAMPLE CASE STUDIES

Considering the number of pit lake predictions that have been done for new mines over the last 15 or so years, there are few papers in the scientific literature on pit lake models or modeling methodologies. Table 10.1 provides a list of papers where models have been described and applied to predicting water quality in mine pit lakes.

TABLE 10.1 Papers in the published literature that have focused on numerical models designed to predict water quality in mine pit lakes

Reference	Modeling Approach
Bird and Mahoney (1994)	Numerical approach for mixing calculations of water chemistry using a combination of geochemical models (MINEDW and PYREACT for mixing and leaching kinetics and MINTEQA2 for chemical equilibrium)
Bird et al. (1994)	Assessment of inverse and forward modeling approaches for predicting pit lake water quality
Vandersluis et al. (1995)	Mixing model approach based on water mass balance and chemical equilibration with PHREEQE
Kirk et al. (1996)	Mixing model approach based on water mass balance and chemical equilibration with PHREEQE, adsorption with MINTEQA2, and limnology with CE-QUAL-W2
Pillard et al. (1996)	Mixing model approach based on leaching kinetics, water mass balance, and equilibrium with PHREEQE and MINTEQA2
Havis and Worthington (1997)	PITQUAL model: a simple water and chemical mass balance model without chemical equilibrium
Kempton et al. (2000)	Probabilistic model of pit lake chemistry based on leaching kinetics and chemical equilibrium with MINTEQA2
Tempel et al. (2000)	Mixing model approach based on water balance with leaching kinetics and chemical equilibrium with EQ3/6
Balistrieri et al. (2006)	Limnological and chemical prediction of water quality with DYRESM and comparison to observed water quality
Schafer et al. (2006)	Experimental approach based on mixing solutions representative of different chemical sources in proportion to their hydrologic mass balances with comparison of experimental results to PHREEQC predictions
Werner et al. (2006)	Linked model approach with MODGLUE for lake internal chemical processes, CE-QUAL-W2 for limnology, and PHREEQC for redox and precipitation

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CHAPTER 11

Modeling Pit Lake Water Quality: Coupling of Lake Stratification Dynamics, Lake Ecology, Aqueous Geochemistry, and Sediment Diagenesis

C.E. Oldham, S.U. Salmon, M.R. Hipsey, and G.N. Ivey

INTRODUCTION

Evolution of water quality in pit lakes, including master state variables such as redox (oxidation–reduction) potential and pH, is initially determined by the quality of surface and subsurface inflows and how the inflow waters interact with pit mineralogy. However, as the volume of inflows relative to the lake volume decreases (as would be expected as the pit fills), the physical, chemical, and biological processes in the lake itself begin to impact, or even dominate, lake water quality. For example, physical processes such as energy transport in the water column affect stratification and mixing, which in turn control the transport of many species to different regions of the lake; biological and microbially mediated lake processes include primary production and the metabolism of organic matter; geochemical processes include the buffering of pH, in circumneutral waters by dissolved inorganic carbon, or under acidic or alkaline conditions by precipitation and dissolution reactions of mineral phases.

There are many feedbacks between these different processes. For example, diagenesis in sediments involves microbially mediated redox reactions driven by the availability of organic matter; the resulting release of nutrients to the water column drives further primary production (organic matter production), and the release of alkalinity may neutralize water column acidity. Another feedback example is the way in which sorption of phosphate to surfaces of amorphous Al and Fe minerals (the solubility of which is pH dependent) may limit phosphate concentrations and hence curtail autochthonous organic matter production. External factors, such as climate, combined with the interactions between the internal processes and lake bathymetry influence the overall lake water quality, including the generation, distribution, and fate of contaminants. The impacts on water quality of all of these lake processes must be balanced against the impact of inflowing waters. For example, the geochemical characteristics of inflows may counteract alkalinity generation within the lake. Alternatively, it can be considered that lake processes may ameliorate poor water quality of inflowing water.

An essential method to test these authors' understanding of what drives the overall water quality in pit lakes and to quantify the relative contribution of inflows versus lake processes is the use of numerical models that adequately describe and quantify the conceptual models and, most importantly, allow the multitude of processes to progress at the appropriate time and length scales. Such models may also be used to investigate the response of pit lake water quality to remediation measures, such as nutrient or organic carbon addition to stimulate alkalinity generation from sediments (e.g., Wendt-Pothoff et al. 2002) or to remove metals from the water column (e.g., Crusius et al. 2003). A comparison of numerical simulation results with carefully conducted field and laboratory observations allows the conceptual models to be rigorously tested under controlled scenarios.

Subsurface water quality is typically assumed to be dominated by geochemical processes, and models such as PHREEQC (Parkhurst and Appelo 1999) have been used successfully to predict inorganic chemical concentrations in the subsurface environment (Prommer et al. 2000). In contrast, density stratification, dissolved oxygen (DO) dynamics, nutrient concentrations, and primary productivity in drinking water reservoirs and lakes have been successfully modeled over the last few decades using a variety of stratification and/or nutrient-phytoplankton-zooplankton (NPZ) models. However, these models pay scant attention to geochemical cycling occurring in lakes (Ramsay et al. 2006). When modeling the long-term water quality of pit lakes, the significance of this geochemical cycling simply cannot be ignored. As will be shown in this chapter, stratification and biological cycling cannot be ignored in attempts to predict remediation outcomes. Up to now, there has been no available model that combines geochemical process descriptions of suitable complexity to resolve the issues of concern within mining influenced waters with the limnological process descriptions of the classic lake stratification and NPZ models.

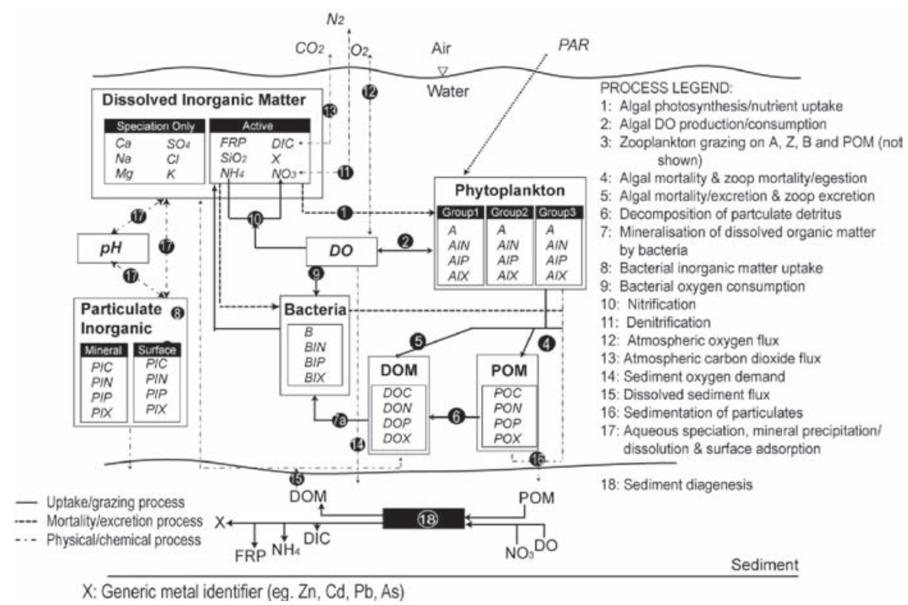
METHODS

Model Description

In 2003–2006, an established NPZ model for lake aquatic ecology, the Computational Aquatic Ecological Dynamic Model or CAEDYM.v2 (e.g., Romero et al. 2004), which can be coupled to one-dimensional (1-D) or three-dimensional (3-D) lake, or two-dimensional (2-D) river, hydrodynamic models, was significantly revised to include aqueous speciation and solubility equilibria, as well as kinetically controlled reactions, in both the water and sediment. The newly developed geochemistry module is applied to determine the dynamics of pH, major ions, metals, and other critical water quality parameters in mine lakes. The kinetic descriptions within the sediment were adapted from CANDI. CANDI is an early-diagenesis model that describes the breakdown of deposited organic matter (Boudreau 1996). CANDI has typically been applied to marine systems (e.g., Haeckel et al. 2001; König et al. 2001; Luff and Moll 2004). An important feature of the new model (CAEDYM.v3) was the ability to investigate feedback between geochemical, ecological, and diagenetic processes. The conceptual description of these processes, including feedback between components, is shown for the water column (Figure 11.1) and the sediment (Figure 11.2). The model was coupled to the 1-D (laterally averaged) hydrodynamic model DYRESM for application to Lake Kepwari in Western Australia.

Aqueous speciation and solubility equilibrium control were accounted for by solving the mass-action expressions for the simulated components, which included Al, Ca, Mg, Na, K, Fe(II), Fe(III), Mn(II), Mn(IV), SiO_2 (silica), Cl, DIC (dissolved inorganic carbon), SO_4 (sulfate), PO_4 (phosphate), NO_3 (nitrate), CH_4 (methane), and H_2S (hydrogen sulfide). The mass-action expressions were solved according to the numerical method of Barrodale and Roberts (1980), as discussed in Parkhurst and Appelo (1999) and in the CAEDYM documentation (Hipsey et al. 2007). Mineral phases for the validation work were limited to those that were significant in the mine lake geochemistry and which were expected to interact with diagenetic processes. These mineral phases include gibbsite, iron hydroxide, and iron sulfide. The mass-action constants from the WATEQ4F database (Nordstrom et al. 1990) were used for speciation. In addition, all dissolved phase geochemical variables can be set to be subject to diffusion in the sediment as in Boudreau (1996).

Unlike other diagenetic models that have incorporated the CANDI approach, the implemented code accounted for both labile and refractory dissolved organic carbon (DOC_L and DOC_R respectively) as well as labile and refractory particulate organic carbon (POC_L and POC_R respectively). Depending on the nature of the investigation, the model may be configured to use either a static or dynamic model for organic matter diagenesis. The organic matter breakdown pathway of the static model is conceptually summarized in Figure 11.1 and uses semiempirical parameterizations for sediment-water flux rates of dissolved components. The dynamic diagenesis model discretizes the sediment profile in layers, and in each layer simulates the hydrolysis of the complex organic matter pools (POC_{VR}, POC_R, DOC_R, and POC_L) and terminal metabolism of low-molecular-weight DOC_L by oxidants (O_2, MnO_2 [manganese dioxide], Fe(III), and SO_4^{2-} [sulfate]), the release and transformation of nutrients (NH_4^+ [ammonium], PO_4^{3-} (phosphate), NO3– [nitrate]), and reduced by-products (Mn2+, Fe(II), NH_4^+, H_2S, CH_4, FeS [iron sulfide]). A complete list of reactions is available in work by Boudreau (1996). The reactions were implemented identically to CANDI, but the generic organic matter term was replaced by DOC_L in the breakdown equations, and the POC_{VR}, POC_R, POC_L, and DOC_R breakdown steps were included using the same reaction rates for all cases except nitrification. For nitrification, a rate of 0.05 day^{-1} was used and no denitrification was allowed to occur below pH 5 because acidity has been found to limit denitrification (Devlin et al. 2000). Note that for brevity, the results shown here are of the more simple static sediment module to allow detailed examination of water column processes and interactions.



NOTE: The various parameters outlined for the geochemical nutrient and biological cycles are defined in detail in Hipsey et al. 2007.

FIGURE 11.1 Schematic of the revised CAEDYM.v3, indicating interactions between the geochemical, nutrient, and biological cycles. Note that the sediment module shown is simplistic; details of the sediment module are given in Figure 11.2.

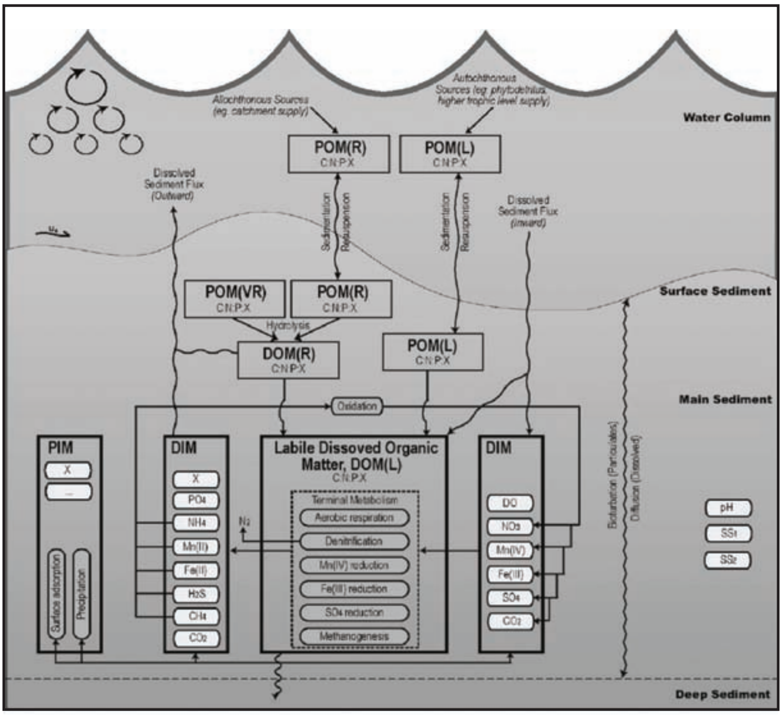


FIGURE 11.2 Schematic of the revised CAEDYM.v3, indicating sediment process description. Note that connectivity between sediment processes and lake ecology or inflow inputs is predominantly via supply of organic matter of varying lability to the sediments.

Model Validation Lake Kepwari

Parallel to model development, lake monitoring was conducted to provide the detailed validation data required to increase the authors' confidence in the developed model. Lake Kepwari, formerly known as Mine Lake WO5B, is a coal pit lake located 160 km south-southeast of Perth, Western Australia (Figure 11.3). Since the cessation of mining in the pit in 1997, the slopes and overburden piles surrounding the pit have been landscaped and revegetated, and coal seams exposed during mining were covered and/or submerged.

The region has a Mediterranean climate with hot, dry summers (12° to 29°C) and cool, wet winters (4° to 15°C). The majority of the rainfall occurs between May and September, and the average annual potential evaporation is estimated to be between 1,450 and 1,650 mm.

After dewatering of the WO5B pit ceased in 1997, the void started to fill with groundwater and precipitation. Between 1999 and 2005, the lake was rapidly filled by winter diversion of the adjacent ephemeral Collie River South Branch (Figure 11.4). Prior to the first river diversion, the lake volume was 10% of the final lake capacity of approximately 24 GL. Since 2005, the lake has been at capacity volume. Ongoing annual river diversion is planned to replace loss to evaporation. An earlier water balance with forward prediction (Varma 2002) estimated a groundwater inflow of up to 0.6 GL per year if the lake volume was less than about 20 GL, no groundwater recharge if the volume was above 20 GL, and annual surface runoff to the lake of 2 to 50 ML. The current volume of annual groundwater discharge or recharge to the void is unknown, as is surface water inflow.

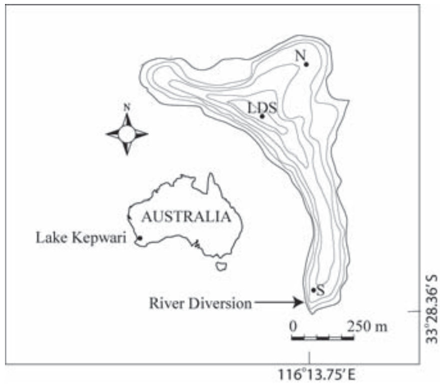


FIGURE 11.3 Lake Kepwari, showing bathymetry using 10-m depth contours. The Lake Diagnostic System (LDS) was located at the center of the lake; water samples were typically collected close to the LDS and from station S.

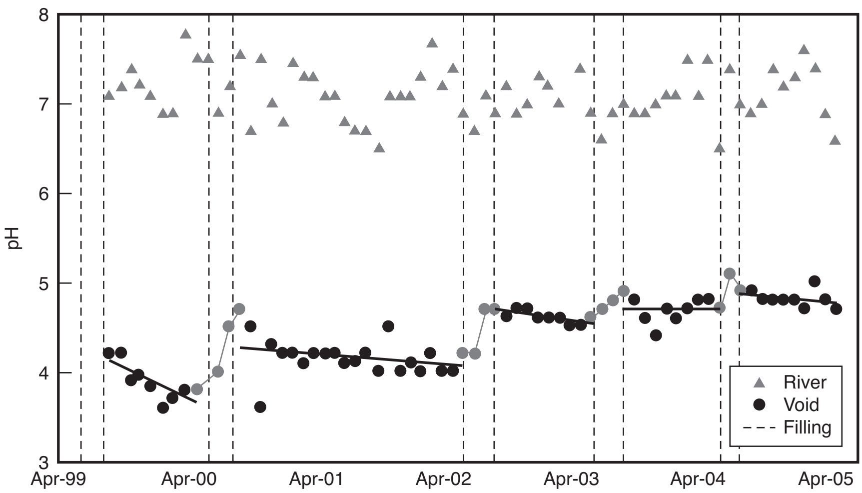


FIGURE 11.4 The Lake Kepwari void was filled with a diversion of the Collie River South Branch. The riverine diversion was pH ~7, and the lake pH increased during diversion periods. However, after diversion was completed each year, the lake pH gradually dropped until the following winter diversion.

Field Sampling

A Lake Diagnostic System (LDS; Precision Measurement Engineering, Carlsbad, California) was installed at the deepest point of the lake (Figure 11.3) from March 2004 to March 2006. The LDS measured wind speed, wind direction, air temperature, relative humidity, and short-wave and net radiation, at a sample rate of 15 seconds. The LDS also sampled water column temperature via 20 thermistors over a depth of 60 m. Water column profiles of temperature, conductivity, DO, oxidation–reduction potential, pH, and photosynthetically active radiation, with depth resolutions ranging from 2 cm to 2 m, were measured approximately every 2 to 6 months from

May 2004 to May 2005. Geochemical, nutrient, and organic carbon sampling was also performed on seven occasions between March 2004 and July 2005.

RESULTS

Largely without calibration, the numerical model prediction of Lake Kepwari water quality over an annual cycle closely reproduced the patterns of stratification and overturn observed in the lake (Figure 11.5), as well as evapoconcentration as indicated by the conservative tracer chloride (Figure 11.6). The model also reproduced major temporal and spatial patterns for nonconservative species, with only NH_4 and DOC (Figure 11.6) and pH and Al (Figure 11.7) shown here for brevity. A sensitivity analysis indicated that even though the lake is now at full capacity, surface and groundwater inflows may still be important factors in the long-term evolution of the lake water quality. The model also allowed the importance of geochemical processes for lake water quality to be tested, for example, solubility equilibrium control of pH by Al hydroxide phases (Figure 11.7). The modeling study of Lake Kepwari thus demonstrated the capability of the model to reproduce the main features of the current water quality in the lake and highlighted the need for testing against data sets, which include well-constrained water and mass balances for the lake.

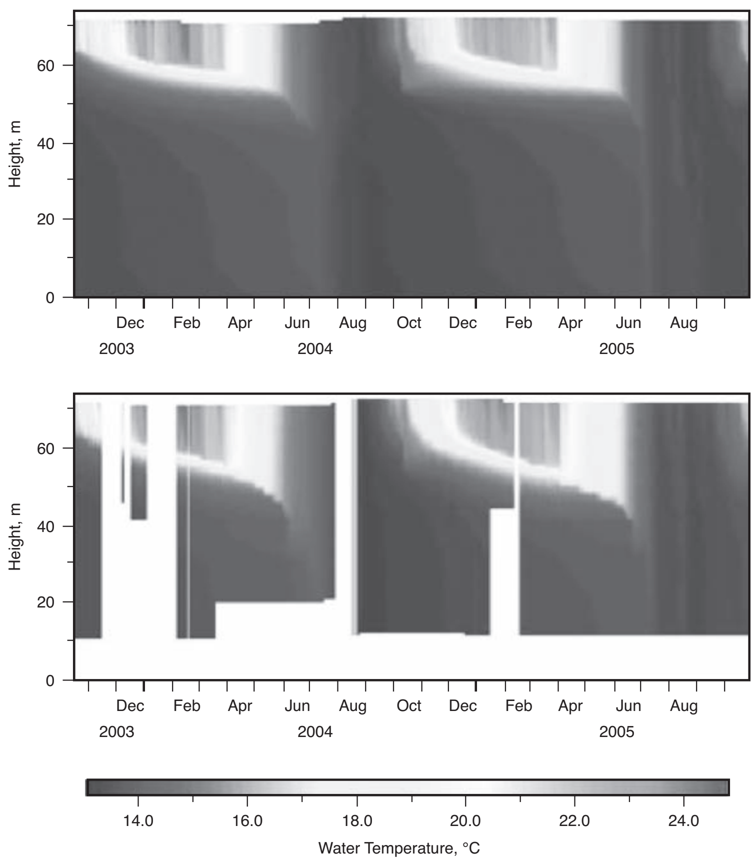


FIGURE 11.5 Temperature stratification cycles in Lake Kepwari from October 2003–October 2005. The top panel shows DYRESM output, and the bottom panel shows LDS field data. Temperature stratification occurs from October to May each year. Note that the riverine diversion occurs in June-July when the lake is isothermal, and therefore its signature is rarely detectable.

DISCUSSION

The strength of the developed model is the tight coupling between the inorganic geochemical cycles and the biological nutrient cycles, as highlighted in the scenario simulations shown in Figure 11.8. When gibbsite dissolution was not included in the model setup (Figure 11.8, panel A), the pH in the lake increased with time. At low pH there is an inhibition of nitrification and therefore an accumulation of NH_4 in the bottom waters. As the pH increases, under the "no gibbsite" scenario, the NH_4 concentrations in the bottoms waters decrease. This could have a significant impact on primary productivity in the lake. When gibbsite solubility is included in the model setup, the lake is maintained at around pH 5, Al concentrations decrease, and the inhibition of nitrification at lower pH causes the accumulation of NH_4 (Figure 11.8, panel B). Finally, in Figure 11.8, panel C, the addition of alkalinity (whether internal or external) is included, which counteracts the acidity source of the gibbsite. As a result, the pH increases, the Al completely drops out of solution, and the NH_4 concentrations decrease. These results illustrate the usefulness of the model, particularly under possible remediation scenarios. For example, one of the most common remediation methodologies is the addition of alkalinity, either in inorganic form or

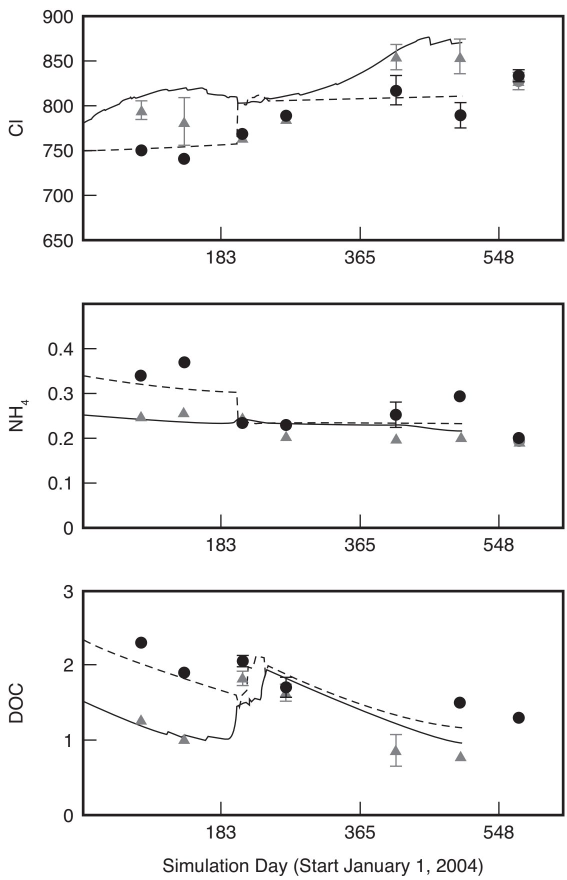


FIGURE 11.6 Model output (lines) compared to field data (symbols) for Cl, NH_4, and DOC. Solid lines indicate depth and volume averaged surface water concentrations, and dashed lines indicate depth and volume averaged bottom water concentrations. Triangles show measured bottom water concentrations, and circles show measured surface water concentrations.

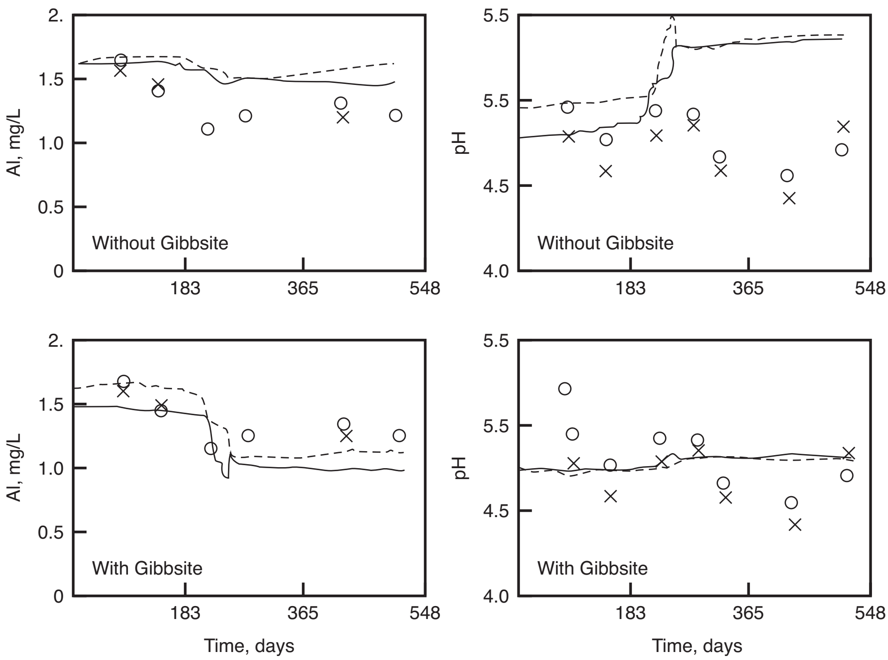


FIGURE 11.7 Model output (lines) compared to field data (symbols). Dashed lines indicate depth and volume averaged surface water concentrations, and solid lines indicate depth and volume averaged bottom water concentrations. Crosses show measured bottom water concentrations, and circles show measured surface water concentrations. The addition of gibbsite solubility equilibrium control to the model improved its ability to simulate the response of Al concentrations and pH to riverine inflow.

through the promotion of biological alkalinity generation. The model allows exploration of the impact of alkalinity addition on geochemical cycling. Ultimately, this process understanding will enhance remediation efficiency.

The generic process descriptions in the developed model can be applied to almost any aquatic system irrespective of the pH. Examples of systems that can now be modeled include the effect of artificial destratification on algal blooms associated with Fe reduction and PO_4 release; treatment of eutrophic waters or drinking water by addition of Al or Fe salts to remove organic matter and/or phosphorous; and the removal of heavy metals through addition of nutrients to stimulate primary production (organic matter scavenging) and/or sulfate reduction (metal sulfide precipitation). The model also allows study of the response of water quality to altered groundwater and surface water inflow under various climate change scenarios.

Although the initial test field site was a coal mine pit lake, the model is based in sophisticated process description and parameterization. This approach and modeling capability allows immediate application to pit lakes of widely varying water quality. The authors now require a comprehensive validation exercise of model predictions against high-quality data sets from pit lakes because it is essential to increase our confidence in the simulation results. In particular, there is a need for testing model performance against long-term data sets from systems with well-constrained groundwater inflow, sediment fluxes and aquatic food web data, as well as hydrodynamic and geochemical state variables. As with all models, there is also the need to quantify, given the available data, the uncertainty in model simulation results and, alternatively, the minimum data

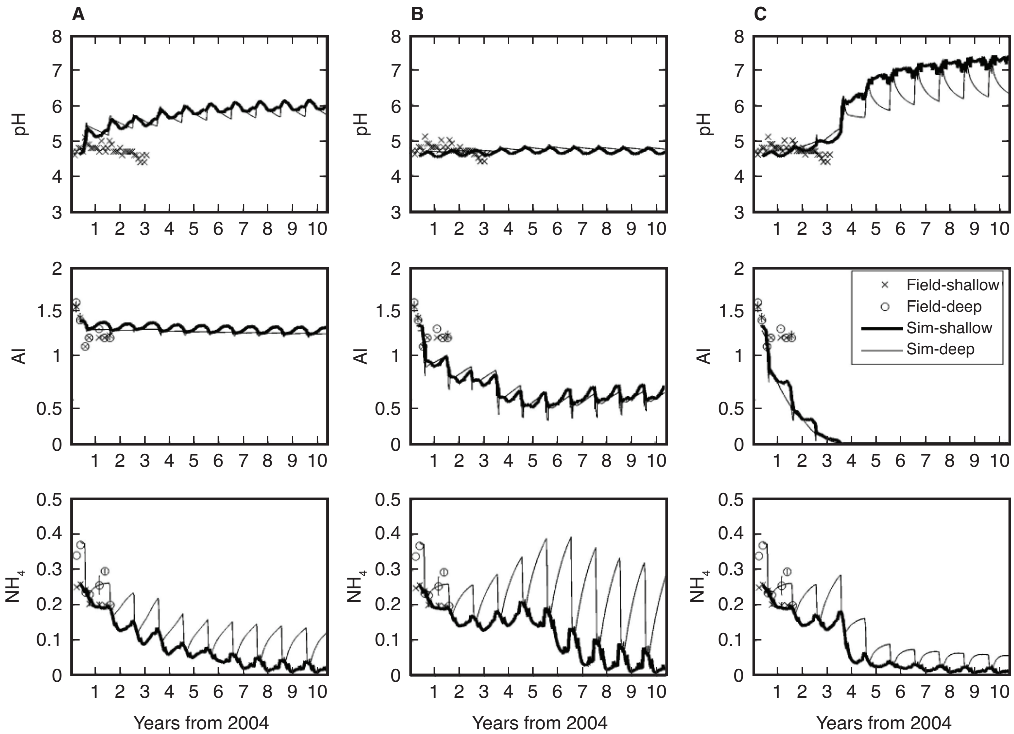


FIGURE 11.8 Model output (lines) of pH, Al, and NH_4, compared to field data (symbols) over a 10-year forward simulation. Note that field data exists only for the first 2 years of the simulation. Dashed lines indicate depth and volume averaged surface water concentrations, and solid lines indicate depth and volume averaged bottom water concentrations. Crosses show measured bottom water concentrations, and circles show measured surface water concentrations. Model setup for panel A output: no gibbsite solubility control, acidic groundwater inflows included; panel B: gibbsite solubility included, acidic groundwater inflows included; panel C: gibbsite solubility included, acidic groundwater inflows included, source of alkalinity included.

required to obtain a result to within a given degree of certainty. The result will be an improved tool for management and remediation of acidic mine lakes, including design optimization for future field and laboratory campaigns.

The model described previously in this chapter would optimally be used in conjunction with the sampling protocol for monitoring required for long-term prediction of water quality in pit lakes. This protocol is currently being developed by the authors, funded by ACMER and a number of industry partners.

In summary, the essential tools for investigating the impact of different closure strategies on future pit lake water quality are

- A sound conceptual knowledge of the system, including current and future water balances;
- A god quality data set (measured and/or modeled) on external sources of contaminants, possible pit configurations/bathymetry, and meteorological conditions; and
- Numerical lake modeling tools, as described in this chapter, to test different closure scenarios.

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CHAPTER 12

Subaqueous Oxidation of Pyrite in Pit Lakes

C. Gammons

INTRODUCTION

A large number of chemical interactions may occur between the water column and submerged mine walls after a pit lake is flooded. Probably one of the most dramatic results of inundation by water is the dissolution of soluble secondary minerals (such as gypsum and other metal-sulfate salts) that have accumulated on the weathered bedrock surfaces during the lifetime of the mine and during the early period of mine flooding. This "first flush" of solutes into the rising lake waters can severely degrade water quality. For example, in a very simple experiment, Newbrough and Gammons (2002) illustrated that interaction of distilled water with crushed, weathered bedrock from the Berkeley pit in Butte, Montana, United States, produced a leachate with a chemistry remarkably similar to that of the Berkeley pit lake. Other water–rock processes that could influence pit lake water quality include adsorption or desorption of metals and metalloids onto clay or hydrous metal oxides, dissolution of primary rock-forming minerals (such as calcite, feldspar, or mafic silicate minerals), and precipitation of secondary minerals, such as jarosite or Fe oxyhydroxides. Precipitation of jarosite is especially common wherever acidic water contacts rock that is rich in potassium-bearing minerals, such as K-feldspar, muscovite, illite, or biotite. Most of these processes are relatively well known and are furthermore covered in other chapters. The main focus of the present chapter is a topic that is much less frequently discussed, that is, the possibility that oxidation of pyrite may continue at a rapid rate *after* a pit lake is flooded.

PYRITE OXIDATION BASICS

The main oxidants for pyrite oxidation are molecular O_2 (gaseous, or dissolved in water) and dissolved ferric iron (Fe^{3+} and other Fe(III) species, such as $Fe(OH)^{2+}$ or $Fe(SO_4)^+$) (Garrels and Thompson 1960; Singer and Stumm 1970; Moses et al. 1987; Williamson and Rimstidt 1994; Nordstrom and Alpers 1999). The following are two example reactions:

$$FeS_2(s) + 7/2O_2 + H_2O = Fe^{2+} + 2SO_4^{2-} + 2H^+ \qquad \text{(EQ 12.1)}$$

$$FeS_2(s) + 14Fe^{3+} + 8H_2O = 15Fe^{2+} + 2SO_4^{2-} + 16H^+ \qquad \text{(EQ 12.2)}$$

Both reactions are themselves the end result of a number of intermediate steps, each of which involves one or at most two electron transfers (Nordstrom 1982; Sand et al. 2001). Although both reactions liberate protons, oxidation of one mole of pyrite by Fe^{3+} produces eight times more acid than oxidation by O_2. In the presence of an excess quantity of O_2, the Fe^{2+} produced from Reactions 12.1 or 12.2 may be oxidized to Fe^{3+}, a process that consumes acid:

$$Fe^{2+} + \tfrac{1}{4}O_2 + H^+ = Fe^{3+} + \tfrac{1}{2}H_2O \qquad \text{(EQ 12.3)}$$

The Fe^{3+} so formed may then undergo hydrolysis and subsequent precipitation as amorphous ferric hydroxide, ferrihydrite, or schwertmannite, with further acid liberation:

$$Fe^{3+} + 3H_2O = Fe(OH)_3(s) + 3H^+ \qquad \text{(EQ 12.4)}$$

$$5Fe^{3+} + 12H_2O = Fe_5O_3(OH)_9(s) + 15H^+ \qquad \text{(EQ 12.5)}$$

$$8Fe^{3+} + 14H_2O + SO_4^{2-} = Fe_8O_8(OH)_6SO_4(s) + 22H^+ \qquad \text{(EQ 12.6)}$$

Reactions 12.1–12.6 are believed to be responsible for the majority of acid generation on mine walls, in waste-rock piles, and in tailings impoundments (Nordstrom and Alpers 1999). The questions that are most relevant to the current discussion are (1) do these reactions continue after a pit lake is flooded; and (2) if so, what would be the possible impacts to water quality?

Rate of Subaqueous Oxidation of Pyrite by Dissolved Oxygen

Beneath the water line, Reaction 12.1 depends on an adequate supply of dissolved O_2. In most pit lakes, the epilimnion (i.e., the upper layer of water above the thermocline) is well mixed and will contain dissolved O_2 at or near equilibrium saturation with atmospheric O_2. At sea level and 25°C, this corresponds to a dissolved O_2 concentration of approximately 8 mg/L O_2, or 0.00025 molal. According to Williamson and Rimstidt (1994), the rate of pyrite oxidation by dissolved oxygen at temperatures near 25°C can be described by the following equation:

$$r(\text{mol/m}^2\text{/s}) = 10^{-8.10} \times \frac{(m\mathrm{O_2})^{0.5}}{(m\mathrm{H^+})^{0.11}} \qquad \text{(EQ 12.7)}$$

where m denotes molal concentration. For the above conditions, the calculated oxidation rates range from $10^{-9.80}$ mol/m^2/s at pH = 0 to $10^{-8.82}$ mol/m^2/s at pH = 9. In 1 year, the maximum amount of pyrite oxidized by Reaction 12.1 would be 0.005 mol/m^2 at pH = 0, increasing to 0.05 mol/m^2 at pH = 9. Thus, pyrite oxidation rates increase with an increase in pH, although the effect is not dramatic. It should be emphasized that the above rates are inorganic and do not take into account the possible catalytic role of microbes. This is discussed in greater detail in the following paragraph.

Are these rates fast enough to influence water quality? The answer depends to a large extent on the amount of pyrite surface area that is exposed on the submerged mine walls. For a back-of-the-envelope calculation, assume that a pit lake can be approximated by a right circular cone with a 1:1 slope and maximum lake depth of 100 m. The total submerged surface area is then 44,000 m^2, and the total lake volume is roughly 1 million m^3. If it is assumed that pyrite is present on approximately 5% of the exposed surface area of the pit, then the total pyrite surface area is 2,200 m^2. Dividing through by the total volume of the lake, this gives 2.2×10^{-6} m^2 of pyrite per liter of water in the lake. Multiplying this by the rate of pyrite oxidation gives $10^{-7.96}$ to $10^{-6.96}$ moles of pyrite oxidized in 1 year per liter of water, assuming that pH is in the range of 0 to 9. If each mole of pyrite oxidized by Reaction 12.1 produces 2 moles of acid, this would result in $10^{-7.66}$ to $10^{-6.66}$ moles/L of H^+. Even if the lake was initially filled with distilled water, this quantity of acid is not enough to drop the lake pH below 6.5.

Although the parameters in the previous calculation are subject to potentially large uncertainties, it is clear that subaqueous oxidation of pyrite by dissolved O_2 is a relatively slow process and should not result in severe water quality degradation. This is particularly true when one considers that pyrite oxidation via Reaction 12.1 consumes dissolved O_2, which would further slow down the oxidation rate. Although dissolved O_2 can be replenished in the epilimnion by diffusion from air, this is not normally possible in deeper portions of the lake, depending on turnover frequency and depth of mixing during turnover. In the absence of any replenishment, Reaction 12.1 would proceed until the initial supply of dissolved O_2 is consumed, at which point the reaction would cease.

Rate of Subaqueous Oxidation of Pyrite by Ferric Iron

Considering Reaction 12.2, Langmuir (1997) provides two equations to estimate the oxidation rate of pyrite by dissolved ferric iron:

$$r(\text{mol/m}^2\text{/s}) = 10^{-8.580} \times \frac{(m\text{Fe}^{3+})^{0.3}}{(m\text{Fe}^{2+})^{0.47}(m\text{H}^+)^{0.32}} \qquad \text{(EQ 12.8)}$$

$$r(\text{mol/m}^2\text{/s}) = 10^{-6.71} \times \text{Eh}^{7.96} \times \text{pH}^{1.06} \qquad \text{(EQ 12.9)}$$

where the temperature is again assumed to be near 25°C. To use Equation 12.8, one needs to know pH and the concentrations of dissolved Fe^{2+} and Fe^{3+}. Redox (oxidation–reduction) speciation of dissolved Fe is relatively straightforward, and a number of rapid colorimetric methods have been developed that can be performed in the field (e.g., To et al. 1999). To use Equation 12.9, one simply needs to know the pH and Eh (volts, corrected to the standard hydrogen electrode) of the mine water. However, Equation 12.9 assumes equilibrium with amorphous ferric hydroxide, which may not be true for a particular sample. For this reason, Equation 12.8 has more general applicability.

According to Equation 12.8, the rate of pyrite oxidation increases with an increase in $m\text{Fe}^{3+}$, decrease in $m\text{Fe}^{2+}$, or increase in pH. Because of the pH dependence of the solubility of ferrihydrite and other secondary ferric minerals, pit lakes that are acidic will contain far more dissolved Fe^{3+} than neutral or alkaline pit lakes (see below). Using data from the deep Berkeley pit lake (Pellicori et al. 2005) as an example of a highly acidic, Fe-rich lake, one can select dissolved Fe^{3+} = 300 mg/L and dissolved Fe^{2+} = 700 mg/L, and a pH of 2.5. Plugging these values into Equation 12.8, the calculated pyrite oxidation rate is $10^{-7.57}$ mol/m^2/s. Following the same assumptions in the preceding hypothetical example, this translates to approximately $10^{-5.75}$ moles of pyrite oxidized per liter of water per year (this assumes that Fe^{3+} concentrations remain buffered at 300 mg/L and do not decrease with reaction progress). Since each mole of pyrite oxidized by Reaction 12.2 produces 16 moles of H^+, the calculated H^+ production rate is $10^{-4.53}$ mol/L/yr. This rate is two to three orders of magnitude greater than that calculated from Equation 12.7 for oxidation by dissolved O_2. This comparatively greater amount of H^+ would be sufficient to lower the pH of distilled water with an initial pH of 7.0 to about 4.5 in 1 year. In 10 years, the pH would be lowered to 3.5.

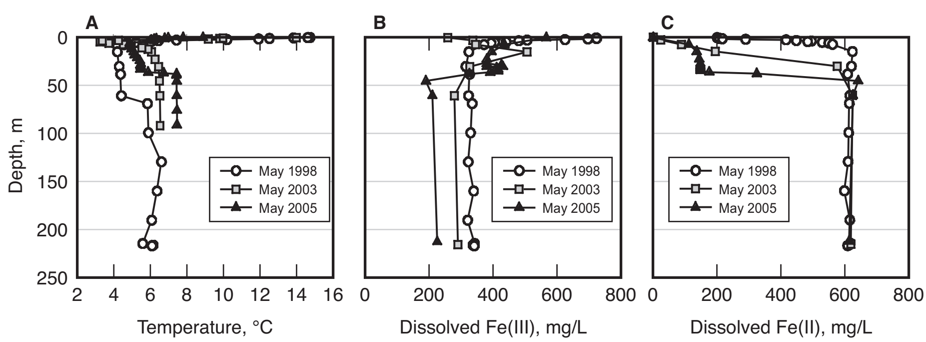
The previous calculation shows that subaqueous pyrite oxidation is much faster when Fe^{3+} is the primary oxidant, and furthermore that the quantity of acid produced could lead to a significant degradation in pit lake water quality. Indeed, Madison et al. (2003) performed detailed calculations specific to the Berkeley pit lake and concluded that 5.9 million to 590 million moles of H^+ could be generated each year by this process. The large range in their calculation was due to uncertainty in the pyrite surface area (see below). Madison et al. (2003) concluded that oxidation of pyrite by dissolved Fe(III) is potentially a very important process in the Berkeley pit lake and could help to explain why the lake has not seen any improvement in water quality since it began flooding in 1983, despite the influx of 11,360 to 22,710 m^3/d (3 to 6 million gpd) of less polluted groundwater and surface water. Stable isotope data presented by Pellicori et al. (2005) support the idea that most of the dissolved sulfate in the pit lake was derived in situ from oxidation of pyrite under anaerobic conditions. More recently, Gammons and Duaime (2006) documented a steady decline in the concentration of Fe^{3+} in the deep pit lake during 1998 to 2005 (Figure 12.1b) and suggested that this decline could be the result of subaqueous pyrite oxidation reactions. Taken together, the body of evidence suggests that oxidation of pyrite by Fe^{3+} below the water line is an important process that has had an impact on the evolution of the chemistry of the Berkeley pit lake. By analogy, it may also be important in other acidic pit lakes that contain elevated concentrations of Fe^{3+}.

Sources of Uncertainty in the Rate Calculations

The preceding calculations are estimates because of uncertainties in the following parameters:

- Specific surface area of pyrite,
- Extent of mixing of water in the pit lake,
- Influence of microbes on pyrite oxidation rates, and
- Changes in reaction rates as a function of temperature.

Regarding the first source of error, it is important to realize that true mineral surface areas (as obtained for example from BET surface area measurements) are invariably at least one to two orders of magnitude higher than what is estimated from simple geometric calculations, such as the circular cone example given previously (White 1995). This is because of the very high degree of surface roughness displayed by minerals undergoing weathering at both macroscopic



Source: Adapted from Gammons and Duaime 2006.

FIGURE 12.1 Vertical profiles in temperature and dissolved Fe(III) and Fe(II) concentration in the Berkeley pit lake in the month of May, collected over the time span 1998 to 2005

(fractures, grain boundaries) and microscopic (etch pits) scales (Rowe and Brantley 1993). On the walls of an open pit, there is a continuum between surfaces that are nearly planar (e.g., freshly blasted bedrock on a steep face) to surfaces that are disaggregated and soil-like (e.g., highly weathered bedrock or unconsolidated material sloughed from above that has settled on a bench). The latter have much higher specific surface areas compared to the former. As well, pyrite and other sulfide ore minerals are often concentrated along the same structures (e.g., veins, fractures, faults) that control the way that rock breaks during blasting. For this reason, the exposed planar surfaces of a mine wall may contain a higher percentage of pyrite than the total volume of unmined rock as a whole. On the other hand, it is also possible that pyrite surfaces may become armored by secondary minerals (such as elemental sulfur, jarosite, or ferric hydroxide) that are metastable intermediates or end products of the oxidation reactions. Although this would tend to decrease the specific surface area of pyrite available for reaction, it is likely that fresh pyrite surfaces are continuously generated on mine walls by slope failures ranging in scale from individual grains salting down the slope to large-scale slumps or landslides. In summary, it is likely that the total surface area of pyrite available for oxidation on submerged pit walls is much higher, and possibly orders of magnitude higher, than what would be estimated from simple geometric constraints. Because the reaction rate is directly proportional to surface area, this means that the above calculation may seriously underestimate the true pyrite oxidation rate.

The second source of error has to do with how frequently and to what extent water near the edge of a pit lake mixes with water in the middle of the lake. Horizontal and vertical mixing occurs in the shallow epilimnion of a lake during any period of sustained wind, which drags water across the lake surface. Less is known regarding vertical or horizontal mixing in the deeper parts of permanently or seasonally stratified pit lakes, although important insights have been gained through detailed research on the limnology of the Island Copper pit lake in British Columbia, Canada (Fisher and Lawrence 2000; Fisher 2002). In a study of the Berkeley pit lake conducted in the late 1990s (Jonas 2000), nearly identical vertical profiles in temperature and chemistry were obtained at three widely spaced locations in the lake. As well, in all depth profiles from the Berkeley pit lake obtained in the past 10 years, the chemistry of the lake has been invariant below the chemocline (i.e., from 10- to 20-m depth all the way to the bottom of the lake at >200 m). This is true, despite the fact that long-term changes in the chemistry of the deeper waters have been observed, such as the decrease in Fe^{3+}/Fe^{2+} ratio of the monimolimnion mentioned previously (Figure 12.1). This author believes that the monimolimnion of the Berkeley pit lake is continuously convecting to minimize temperature gradients between the relatively warm granite bedrock beneath the lake and the colder mixolimnion above the chemocline. Convection of the deep lake is no doubt enhanced by the continual influx of approximately 7,570 to 11,360 m^3/d (2 to 3 million gpd) of warm (>15°C) water from the adjacent and underlying flooded underground mine complex. Other mechanisms that could promote mixing at deep levels in a pit lake include turbidity flows and eddy diffusion caused by episodic landslides, or by the influx of wastewater that sinks to the bottom of the lake, such as mill tailings or high-density sludge (Stevens and Lawrence 1997).

The third source of uncertainty in the rate calculations has to do with the possible catalytic effect of microbes. Many studies have shown that pyrite oxidation is faster in the presence of microorganisms, and this is the basis of the science of bioleaching (e.g., Sand et al. 2001). However, exactly how the microbes speed up the process is a complex topic and is still being researched. It turns out that Reaction 12.2, the initial attack of pyrite by Fe^{3+}, is fairly fast even in the complete absence of bacteria, as long as an ample supply of dissolved ferric iron is present (Williamson and Rimstidt 1994; Langmuir 1997). Microbes may speed up the initial attack but probably not by more than a factor of 2 (Fowler et al. 1999). Of course, Reaction 12.2 will eventually slow down or stop after the supply of ferric iron is used up. If molecular oxygen is present, ferric iron can be regenerated by reoxidation of Fe^{2+} (Reaction 12.3). However, this simple reaction is very slow at low pH in the absence of microbes and therefore is often cited as the rate-limiting step in sustaining pyrite oxidation over the long term (Singer and Stumm 1970). This is where microbes enter the picture, as Reaction 12.3 is sped up many orders of magnitude in the presence of bacteria such as *Acidithiobacillus ferrooxidans* (Nordstrom 2003). Although this discussion is greatly simplified, the result is that microbes will always increase pyrite oxidation rates, but the magnitude of this effect may be as little as a factor of 2 (if Reaction 12.2 is rate limiting) or as much as a factor of more than 10,000 (if Reaction 12.3 is rate limiting). Because the oxidation rate calculations in the preceding section were based on Reaction 12.2 and assumed an ample supply of ferric iron, the overall effect of microbes in the deep Berkeley pit example is probably minor.

Regarding the final source of error, pyrite oxidation rates should be slower in cold water as compared to warm water. The magnitude of this effect depends on the enthalpy of the rate-limiting step in the overall reaction. In a series of bench-top experiments using pyrite from the Butte ore body and synthetic $Fe_2(SO_4)_3$ (ferric sulfate) or $FeCl_3$ (ferric chloride) solutions, Pellicori (2004) found that the rate of anaerobic pyrite oxidation by Fe^{3+} was roughly 2.3 times faster at 20°C than at 5°C, and estimated an enthalpy of +41 kJ/mol for the rate-limiting step. This compares reasonably well with similar estimates of +46.2 to +56.3 kJ/mol published by Schoonen et al. (2000) for pyrite oxidized by Fe^{3+} in the pH range 2 to 6. Thus, subaqueous pyrite oxidation by Fe^{3+} is indeed slower in cold water, although the magnitude of the temperature effect is probably less than other uncertainties, such as the estimation of mineral surface area or the possible influence of microbes.

What Happens When the Supply of Soluble Ferric Iron Runs Out?

Reaction 12.2 requires a renewable source of dissolved Fe^{3+}. Without it, the reaction will slow down and eventually cease. The concentration of Fe^{3+} can be regenerated in two ways: (1) by oxidation of Fe^{2+}, and (2) by dissolution of ferric minerals present on the mine walls or suspended in the water column. The first mechanism (Reaction 12.3) and the important role that bacteria play in catalyzing this otherwise slow reaction have already been discussed. If the only supply of Fe^{3+} comes from oxidation of Fe^{2+}, then a combination of Reactions 12.3 and 12.2 reduces to Reaction 12.1. Thus, although Fe^{3+} may still play a lead role in breaking down the pyrite structure, the overall rate of pyrite oxidation in the long term is still controlled by the availability of dissolved O_2. If dissolved O_2 is absent (as may be the case in deeper portions of the lake), then oxidation of Fe^{2+} cannot take place, and any Fe(III) must be regenerated by dissolution of ferric solids, such as ferric hydroxide, ferrihydrite, or schwertmannite (the reverse of Reactions 12.4 to 12.6). The following reaction describes the oxidation of pyrite using ferric hydroxide as the source of Fe(III):

$$FeS_2(s) + 14Fe(OH)_3(s) + 26H^+ = 15Fe^{2+} + 2SO_4^{2-} + 34H_2O \qquad \text{(EQ 12.10)}$$

Note that the above reaction consumes a large number of proton equivalents. What this means is that if water in deeper portions of the lake has no access to dissolved O_2, then continued pyrite oxidation by Reaction 12.2 must eventually result in dissolution of ferric minerals on the mine walls, and this process in turn will cause an increase in pH. Paradoxically, this does not mean that the quality of the water will improve. In fact, because Fe^{2+} carries two equivalents of metal acidity, Reaction 12.10 causes the total acidity of the water to increase by 4 moles for each mole of pyrite oxidized. In other words, the acidity of the water is transferred from protons to Fe^{2+}

with a net degradation in water quality from the standpoint of the quantity of lime that would eventually be needed to treat the water.

Whether subaqueous pyrite oxidation drives pH down by Reaction 12.1 or drives pH up by Reaction 12.10 depends on the availability of dissolved O_2. Thus, it is more likely that Reaction 12.1 will be dominant in the epilimnion of pit lakes, whereas Reaction 12.10 may be dominant in the hypolimnion or monimolimnion. This author is not aware of a pit lake that provides a compelling case for Reaction 12.10. However, adjacent to the Berkeley pit in Butte, there is a flooded mine shaft that has the exact chemistry that one would predict from Reaction 12.10. This is the Kelley mine shaft, a large underground copper producer in its day, which later became the location of the main dewatering pumps that allowed the Anaconda Company to mine the Berkeley open pit. Water in this mine shaft is not strongly acidic (pH ~ 4), but nonetheless has extremely high concentrations of Fe(II) (>2,000 mg/L) (Pellicori et al. 2005; Gammons et al. 2006). Groundwater in the Kelley shaft is anaerobic, and O_2 levels in the overlying column of air within the shaft are depleted for at least 50 m above the static water level (Snyder 2008). The Kelley water is also anomalously warm (>35°C), a phenomenon that could be due to pyrite oxidation (Gammons et al. 2006). All of these lines of evidence suggest that pyrite oxidation is occurring at a rapid rate in the Kelley mine in the complete absence of O_2, assisted by dissolution of secondary ferric compounds stored on the mine walls to provide a continuous source of Fe(III) to keep the oxidation reaction going. Despite the fact that the Kelley mine water has a much higher pH than the Berkeley pit lake (pH 2.5), as well as much lower concentrations of heavy metals (with the exception of Fe), the two waters have a similar total acidity: 105 mmol/L for the Berkeley pit lake vs. 97 mmol/L for the Kelley (based on chemical data in Pellicori et al. 2005).

In summary, mine water in the Kelley shaft could be an analog of how water in the bottom of an initially acidic pit lake may evolve when there is no access to dissolved O_2, but when subaqueous pyrite oxidation is still playing an important role in modifying the pit lake chemistry. If the trend of decreasing Fe^{3+} concentration in the deep Berkeley pit lake (Figure 12.1b) continues, this may initiate dissolution of ferric minerals known to exist in abundance on submerged mine walls and lake sediment (such as schwertmannite and K-jarosite), with the possible onset of a long-term trend of increasing pH. Regulatory agencies should not be delighted by this prospect, however, as the total acidity of the deep water will continue to increase, with the bulk of the acidity shifting from Fe^{3+} and H^+ to Fe^{2+}.

WHAT ABOUT PIT LAKES THAT ARE NOT STRONGLY ACIDIC?

Although it is true that acid pit lakes will contain much more Fe^{3+} than pH-neutral lakes, some simple calculations show that the rates of pyrite oxidation by Fe^{3+} are not necessarily much slower at higher pH. Figure 12.2a shows the calculated concentration of dissolved Fe^{3+} as a function of pH, assuming equilibrium with both ferrihydrite and gypsum (calculations made using Visual MINTEQ). Figure 12.2b shows the calculated pyrite oxidation rates according to Reaction 12.8 as a function of pH, using the same Fe^{3+} concentrations shown in Figure 12.2a. The data are contoured for different Fe^{2+} concentrations, and the composition of the Berkeley pit lake is shown for comparison. Despite the fact that the Fe(III) concentration of a pH 5 water is more than a million times lower than a pH-2 water, the pyrite oxidation rate at any given Fe^{2+} concentration is only about 10 times lower. Thus, pyrite oxidation by ferric iron is not necessarily a phenomenon restricted to low-pH waters. However, because the concentration of Fe^{3+} is so much lower at higher pH, the reactant must be continuously replenished, such as through oxidation of Fe^{2+} or dissolution of secondary ferric compounds.

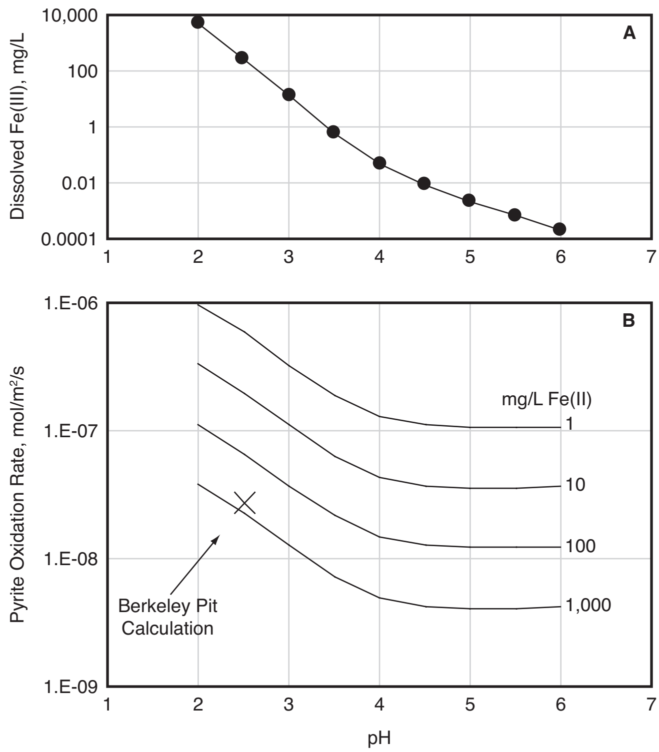


FIGURE 12.2 Geochemical calculations that have a bearing on the rate of oxidation of pyrite by dissolved ferric iron. (a) The dependence of Fe(III) concentration on pH (assumes equilibrium saturation with ferrihydrite and gypsum); and (b) the dependence of pyrite oxidation rate on pH and Fe(II) concentration, assuming the same Fe(III) concentrations given in (a).

CONCLUSIONS

The main purpose of this chapter is to make pit lake researchers aware of the possibility that pyrite oxidation may continue at a moderate to rapid rate after a pit lake is flooded, even in the complete absence of dissolved oxygen. Previously published equations are presented that can be used to estimate pyrite oxidation rates for any lake, although the uncertainties in such calculations are rather large. If a renewable supply of dissolved O_2 is available, then subaqueous pyrite oxidation will lower pH. However, if dissolved O_2 is absent, consumption of Fe^{3+} may eventually cause dissolution of secondary ferric oxyhydroxides present on the mine walls or descending by gravity through the water column, with a subsequent increase in pH. The end result of such a situation could be a deep pit lake with moderately acidic pH (~4) but with extremely high Fe^{2+} concentration and therefore very high total acidity. Although most of the discussion in this chapter has focused on strongly acidic pit lakes, such as the Berkeley pit, subaqueous pyrite oxidation may also be important for lakes that are only slightly acidic.

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CHAPTER 13

Microbial Reactions

B. Wielinga

INTRODUCTION

In recent years there has been a growing awareness that microbial activity and processes have a fundamental role in numerous geological and geochemical processes and in shaping the habitable part of our planet (Ehrlich 2002). As such, consideration of the role that microorganisms may play in the evolution of pit lakes is appropriate. The purpose of this chapter is to provide an overview of some of the potentially important biogeochemical reactions that could affect pit lake water quality and review the current state of knowledge concerning these processes in the pit lake environment.

MICROBIAL REACTIONS AN OVERVIEW

Numerous microorganisms, which include prokaryotes (organisms that lack a true nucleus) and eukaryotes (organisms that have a true nucleus), play a significant role in geological and geochemical processes. Prokaryotic microorganisms include all types of bacteria (e.g., members of the domains Archaea and Bacteria), while eukaryotic microorganisms include the algae, fungi, protozoa, and slime molds (Ehrlich 2002). Until recently, the importance of microbial activity in various geological processes was not sufficiently appreciated by microbiologists and geologists/geochemists. Thus, it is probably safe to say that our understanding of the significance of these various processes in the pit lake environment is also currently limited. However, it has been demonstrated that microbial processes play a role in the cycles of many elements that can be significant to pit lake chemistry, including carbonates, silicon, aluminum, phosphorus, nitrogen, arsenic, antimony, mercury, iron, manganese, chromium, molybdenum, sulfur, selenium, tellurium, uranium, technetium, vanadium, neptunium, gold, silver, and cyanide (Ehrlich 2002; Lovley 1993). Although a comprehensive discussion of all of the biological reactions that may be pertinent to pit lake chemistry is beyond the scope of this chapter, those that are currently thought to be of greatest importance will be presented here.

The energy that supports all life on earth is obtained through a complex series of oxidation–reduction (redox) reactions in which the oxidation of a reduced inorganic or organic substrate is coupled to the reduction of a more oxidized compound. In the oxidation–reduction process, the compound that is oxidized gives up electrons and the compound that is reduced gains electrons. Microorganisms are very efficient at catalyzing these electron transfers and thereby gain energy for growth through these reactions. This concept is illustrated in Figure 13.1 with the important cycle of carbon fixation, in which inorganic carbon (CO_2) is converted to organic carbon (represented by CH_2O) in a process called carbon fixation or primary production, and organic carbon is oxidized back to carbon dioxide with the reduction of molecular oxygen in a process referred to as aerobic respiration.

$$\underset{\text{(oxidized)}}{CO_2} + \underset{\text{(reduced)}}{H_2O} \xrightleftharpoons[\text{Carbon Oxidation/Aerobic Respiration (heterotrophy)}]{\text{Carbon Fixation/Primary Production (photosynthesis, chemolithotrophy)}} \underset{\text{(reduced)}}{CH_2O} + \underset{\text{(oxidized)}}{O_2}$$

FIGURE 13.1 Generalized reactions describing carbon fixation and organic carbon oxidation

TABLE 13.1 Classification scheme for defining physiological bacterial groups

Physiological Group	Energy Generating Process	Source of Organic Carbon
Photolithotrophs	Photosynthesis (convert solar energy to chemical energy)	Inorganic carbon fixation
Chemolithotrophs	Oxidation of reduced inorganic compounds	Inorganic carbon fixation
Mixotrophs	Oxidation of reduced inorganic and/or organic compounds	Organic carbon sources
Photoheterotrophs	All or part of energy from the sun (solar energy)	Organic carbon sources
Heterotrophs	Oxidation of organic carbon compounds	Organic carbon sources

Numerous oxidation–reduction reactions are catalyzed by microorganisms, which may have a significant impact on pit lake chemistry and evolution. However, before these reactions are discussed in more detail, it will be helpful to define a few terms common to microbiology.

Prokaryotes can be divided into specific physiological groups such as chemolithotrophs, photolithotrophs, mixotrophs, photoheterotrophs, and heterotrophs based on how they obtain energy for growth and metabolism and where the carbon that gets incorporated into cellular biomass originates (Ehrlich 2002). Table 13.1 provides a summary for how these groups are defined.

The fixation of carbon as shown in Figure 13.1 requires energy and is accomplished by select groups of organisms, either photosynthetic plants and bacteria or chemolithotrophic bacteria. Photosynthetic processes are conducted by green plants, green algae, cyanobacteria (blue-green algae), and the purple and green bacteria, which derive the energy required for carbon fixation from the sun. Chemolithotrophic bacteria gain energy for carbon fixation through the oxidation of various reduced inorganic compounds, which include H_2, HS^-, S^0, NH_4^+, NO_2^-, and Fe^{2+}. The importance of these biological reactions is the addition of organic material to the system, which can in turn be utilized by other organisms.

In the reverse reaction, the reduced carbon produced from carbon fixation is oxidized in a coupled reaction in which an inorganic or organic compound is reduced. In the reaction shown in Figure 13.1, the oxidation of organic carbon is coupled to the reduction of molecular oxygen, a process termed aerobic respiration in which oxygen is used as the terminal electron acceptor in the terminal electron accepting process. Numerous bacteria are able to couple the oxidation of organic carbon to the reduction of alternative electron acceptors in a process termed anaerobic respiration. Some of the most important of these reactions are shown in Table 13.2. With this very brief overview of biological reactions and processes, the potential effects on pit lake systems will be explored.

TABLE 13.2 Oxidation–reduction reactions catalyzed by microorganisms in respiratory pathways

Reaction	Redox Couple	Eh, volts
Aerobic respiration: $CH_2O + O_{2(g)} \rightarrow HCO_3^- + H^+$	$O_{2(g)}/H_2O$	0.816
Denitrification: $5CH_2O + 4NO_3^- \rightarrow 2N_{2(g)} + 5HCO_3^- + H^+ + 2H_2O$	$NO_3^-/N_{2(aq)}$	0.713
Managanese reduction: $2CH_2O + 2MnO_{2(s)} + 3H^+ \rightarrow 2Mn^{2+}_{(aq)} + 2HCO_3^- + 2H_2O$	MnO_2/Mn^+	0.544
Iron reduction: $CH_2O + 4Fe(OH)_{3(s)} + 7H^+ \rightarrow 4Fe^{2+}_{(aq)} + HCO_3^- + 10H_2O$	$Fe(OH)_3/Fe^{2+}$	0.014
Sulfate reduction: $2CH_2O + SO_4^{2-} + 2H^+ \rightarrow H_2S_{(aq)} + CO_2 + 2H_2O$	$SO_4^{2-}/H_2S_{(aq)}$	–0.217

*Table is organized in order of energy available from the reactions (most energetic on top) and the approximate redox conditions at which the reactions are expected to occur.

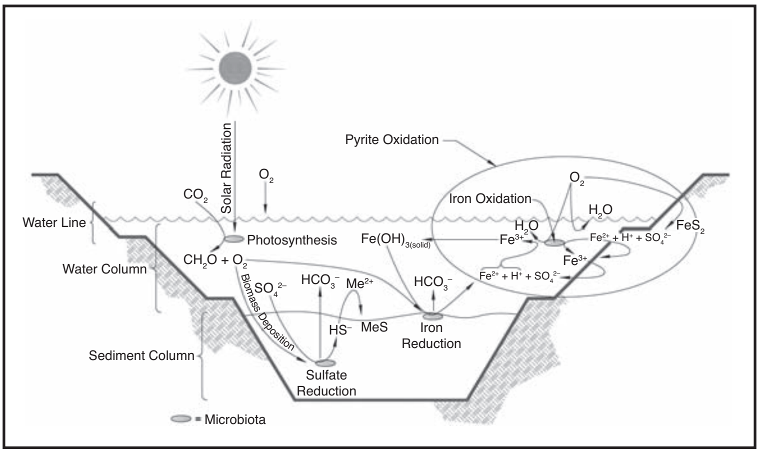

FIGURE 13.2 Conceptual illustration of potentially significant biogeochemical reactions in pit lakes

MICROBIAL REACTIONS IN PIT LAKES

In general, little is known about the microorganisms that inhabit mine pit lake environments, and a limited number of reports exist on the ecology of pit lakes (Kalin et al. 2001; Mitman 1999). Of the physiological groups briefly described previously, the following three have been the best characterized in pit lake environments:

- Chemolithotrophs—the sulfur- and iron-oxidizing bacteria;
- Heterotrophs—dissimilatory metal- and sulfate-reducing bacteria; and
- Photolithotrophs—algae and cyanobacteria (blue-green algae).

In this section, the effects that these microbial processes may have on pit lake water quality are considered. For the discussion of these processes, the pit lake environment has been divided into three areas, which include the water line (air/water/wall-rock interface), the water column, and the sediment column. The primary reactions that will be discussed in the following sections are illustrated schematically in Figure 13.2.

Reactions at the Water Line

The oxidation of pyrite associated with pit wall rock can have a significant effect on pit lake water quality. The process of metal sulfide and/or pyrite oxidation is complex and involves both chemical and biological pathways. The process of pyrite oxidation is described by the reactions that follow. The initial reaction is the chemical oxidation of pyrite by molecular oxygen, which results in the formation of ferrous iron and sulfuric acid:

$$FeS_2 + 3.5O_2 + H_2O \rightarrow Fe^{2+} + 2H^+ + 2SO_4^{2-} \qquad \text{(EQ 13.1)}$$

The ferrous iron generated in this reaction is subsequently oxidized by numerous iron-oxidizing bacteria (Ehrlich 2002) according to the following reaction:

$$2Fe^{2+} + 0.5O_2 + 2H^+ \rightarrow 2Fe^{3+} + H_2O \qquad \text{(EQ 13.2)}$$

followed by the rapid chemical oxidation of residual pyrite by the resultant ferric iron according to the reaction:

$$FeS_2 + 14Fe^{3+} + 8H_2O \rightarrow 15Fe^{2+} + 2SO_4^{2-} + 16H^+ \qquad \text{(EQ 13.3)}$$

whereby additional ferrous iron is regenerated.

The oxidation of pyrite by ferric iron is kinetically much faster compared to the oxidation of pyrite by molecular oxygen (Williamson and Rimstidt 1994). Thus, the supply of ferric iron from ferrous iron is a critical factor limiting the rate of pyrite oxidation (see Chapter 12). At high pH values (e.g., pH >6), ferrous iron is rapidly oxidized chemically without biological involvement (Williamson et al. 2006). At low pH, microbially mediated ferrous iron oxidation by bacteria predominates (Pronk and Johnson 1992). At low pH (pH <3), microbial activity can significantly speed the conversion of ferrous to ferric iron and thus increase the rate of iron oxidation by a factor of 106 (Williamson et al. 2006). Rates of iron sulfide oxidation reported for ore tailings and coal dumps range from 0.43 to 1,040 mol $S \cdot m^{-2} \cdot yr^{-1}$ (Blodau 2006). In situ measurements and reactor studies have demonstrated iron oxidation rates ranging from 0.03 $mol \cdot L^{-1} \cdot yr^{-1}$ to 350 $mol \cdot L^{-1} \cdot yr^{-1}$ (Blodau 2006; Foos 1997; Kirby and Elderbrady 1998; Noike et al. 1983) at pH values that range from 1.3 to 7.7. These rates approximately bind those measured in tailings and waste rock facilities (Blodau 2006).

Pyrite oxidation may be of greatest importance at the water line (the air/water/wall-rock interface), where both oxygen and water are available. Above the water line, especially in arid regions, the lack of moisture will limit microbial activity, and at depth in the pit lake, oxygen is expected to be limiting.

Reactions in the Water Column

Biological reactions in the water column shown in Figure 13.2 include photosynthesis and iron oxidation. Photosynthetic reactions can be especially important because they are primary production reactions that add organic carbon to the system, which in turn can fuel other important heterotrophic processes. In addition, the sedimentation of these organisms when they die off can be an important process for removing metals and metalloids from the water column (see Chapter 19).

Primary production and thus the accumulation of organic matter in pit lakes are often limited by phosphate adsorption and co-precipitation with ferric iron and degassing of dissolved inorganic carbon (DIC) at low pH (Nixdorf and Kapfer 1998) as shown in the following reactions:

$$PO_4^{3-} + Fe^{3+} \rightarrow FePO_{4(solid)} \quad \text{(EQ 13.4)}$$

$$HCO_3^- + H^+ \rightarrow H_2O + CO_{2(gas)} \quad \text{(EQ 13.5)}$$

Water turbidity, which can limit light penetration, can also limit primary productivity of photosynthetic organisms in the limnetic zone (the well-lit open surface waters away from the shore).

Primary productivity in some pit lakes has been found to be spatially limited to the benthic regions, metalimnion, and the littoral zones, where phosphorus and DIC are available (Koschorreck and Tittel 2002; Nixdorf and Kapfer 1998). Since many pit lakes associated with hard-rock mining are relatively deep, with steep side walls and a limited littoral zone, limnetic productivity will dominate. In a 7-year study looking at the evolution of a pit lake in Northern Saskatchewan, Canada, Kalin et al. (2001) used principal component analysis to identify the three key factors, total suspended solids (TSS), total-P, and arsenic driving changes in phytoplankton community composition. However, in an acidic mining lake (pH 2.6), benthic algae predominantly *Eunotia* spp. and *Pinnularia obscura*, were found to be adapted to low light intensities, which may have been explained by the efficient adsorption of red light, the dominant wavelength available in the ferric iron–rich lake (Koschorreck and Tittel 2002). Similar populations adapted to low light conditions may therefore also enhance primary productivity in the limnetic zone.

Rates of primary productivity measured in three pit lakes in Germany were found to be low and ranged from 0 to 12.5 milligrams organic carbon per cubic meter per hour ($mg\ C{\cdot}m^{-3}{\cdot}h^{-1}$) as compared to 27 to 230 $mg\ C{\cdot}m^{-3}{\cdot}h^{-1}$ measured in natural lakes in the same region (Beulker et al. 2003; Nixdorf et al. 2003). In contrast, blooms of algae, *Chlamydomonas botryopara*, with cell densities of 6.5 × 106 mL−1 corresponding to 700 $mg{\cdot}L^{-1}$ fresh weight and 2,660 $\mu g{\cdot}L^{-1}$ chlorophyll-a have been observed in an extremely acidic coal mining pond (pH 2.5) when appropriate nutrients were available (Woelfl et al. 2000). Conditions that favor phytoplankton growth can facilitate the removal of metals/metalloids from the water column by adsorption to cell biomass and subsequent deposition to the sediment (Dessouki et al. 2005; Mitman 1999; Chapter 17).

The oxidation of ferrous iron in the water column can lead to the formation of various iron precipitates, such as goethite, jarosite, ferrihydrite, and schwertmannite dependent on the geochemical conditions in the lake (Bigham et al. 1990; Bigham et al. 1996; Peine et al. 2000; Regenspurg et al. 2004). The poorly crystalline oxyhydroxysulfate mineral schwertmannite may be the primary iron precipitate in acidic mine lakes (Bigham et al. 1990; Peine et al. 2000; Regenspurg et al. 2004). When precipitated to the lakes sediment, the iron oxyhydroxides are subject to both chemical and biological transformations.

Reactions in the Sediment Column

Reactions expected in the sediment column are those associated with heterotrophic processes, such as fermentation of organic material, and metal and sulfate reduction. Heterotrophic microbial activity, such as iron and sulfate reduction, can be very important reactions as they can add alkalinity to the pit lake system and remove metals via precipitation of metal sulfide minerals (Blodau 2006; Kusel 2003) as shown in Figure 13.2 and the reactions in Table 13.2. While these reactions could also occur in the water column in anoxic water, because in natural systems most bacteria are found attached to surfaces rather than free swimming, it is expected that the sediment column will support much of this activity. In addition, within the sediment column, oxygen can be consumed rapidly and depleted within several centimeters of the sediment–water interface, and therefore these reactions can proceed in the sediment column within holomictic or meromictic pit lakes with oxic or anoxic bottom waters, respectively. A fundamental requirement for this heterotrophic activity, in addition to reducible iron and/or sulfate, is the availability of an organic carbon substrate. In addition to the organic material generated by primary productivity, mine pit lakes may contain organic carbon derived from soil percolates, soil organic matter, coal, and bituminous materials that can function as electron donors for microbial respiration. Conditions that limit carbon fixation and slow the decomposition of natural organic matter in mining influenced waters (MIWs) can limit the availability of electron donors in mine pit lakes. The availability of simple and readily usable organic carbon compounds is often the controlling factor for these reductive reactions in pit lakes.

Recent literature indicates that the rates of these reactions and the generation of alkalinity can vary considerably under natural conditions (Blodau 2006), and cycling between oxidized and reduced species can limit system changes (Blodau 2006; Knoller et al. 2004; Kusel 2003). In sediments of coal mining pit lakes in Germany, Kusel (2003) and Peine et al. (2000) found the reduction of Fe(III), apparently mediated by *Acidiphillium* species, to be the dominant electron-accepting process for the oxidation of organic matter at low pH. However, on account of the absence of sulfate reduction and therefore the lack of sulfide in the upper 6 cm of sediment, the Fe(II) formed in the sediment diffuses to the oxic zones in the water column where it is reoxidized by *Acidithiobacillus* species. Presumably, the iron cycling established along with the acidity generated by the transformation of schwertmannite by dissolution to goethite stabilizes the acid conditions in the upper portion of the sediments, which favor acidophilic iron reduction at the expense of fermentation and sulfate reduction (Kusel 2003; Peine et al. 2000). A conceptual model for the proposed iron cycling is shown in Figure 13.3. At greater depth in the sediment, pH values >5 were measured, and fermentation and sulfate reduction was observed.

In large (10-m-diameter) sediment enclosure experiments in which sediments were amended with Carbokalk, a waste product from the sugar industry that contains organic carbon and lime, similar reactions were observed (Wendt-Potthoff et al. 2002). In these experiments, iron reduction started within 1 week of sediment amendment and thus was the initial step in microbial alkalinity generation. As the experiments continued, Fe(III) and sulfate reduction were shown to occur simultaneously, but the rate of iron reduction was 3.5 times greater than that of sulfate reduction. The absence of carbonate and sulfide to precipitate the Fe(II) and form siderite ($FeCO_3$) and mackinawite (FeS) allowed the reduced iron to move through the water column where it could be reoxidized. Field measured rates of iron and sulfate reduction reported from these studies ranged from 3.5 to 11.3 $mol{\cdot}m^{-2}{\cdot}yr^{-1}$ and 0.9 to 1.0 $mol{\cdot}m^{-2}{\cdot}yr{\cdot}$, respectively (Wendt-Potthoff et al. 2002).

While Lake Anna, Virginia (United States) is not a pit lake, it does receive MIW from the adjacent mineral district and thus provides some documentation for the potential contribution of sulfate reduction in sediments of lakes impacted by MIW. Under moderately acidic (pH ~4) and eutrophic conditions found in Lake Anna, significantly greater rates of sulfate reduction were observed (Herlihy and Mills 1985). Sulfate reduction rates in the portion of Lake Anna impacted by acidic mine water reached a maximum of 82.5 $mol{\cdot}m^{-2}{\cdot}yr^{-1}$ compared to 4.9 $mol{\cdot}m^{-2}{\cdot}yr^{-1}$ at an unpolluted control site. The sulfate reduction rates observed here during summer months were higher than those observed in several marine sediments (Canfield et al. 1993; Jorgensen 1977) and were comparable to the high rates observed in salt marshes (Howarth and Teal 1979).

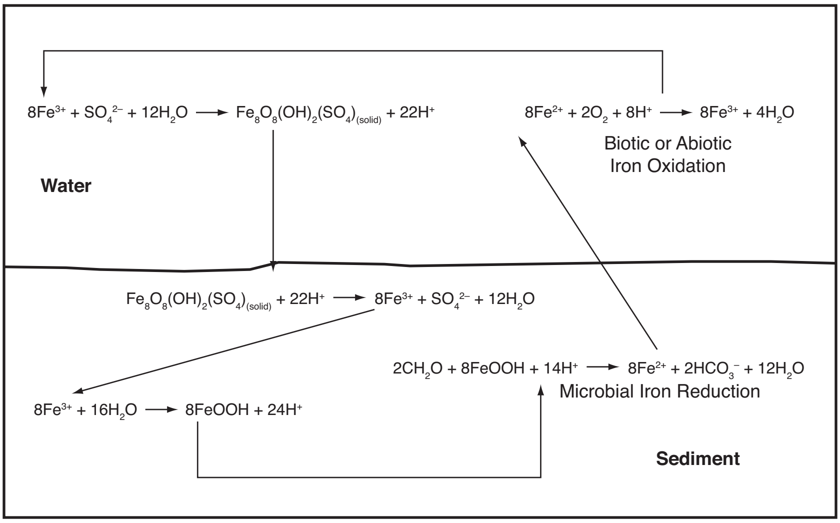

FIGURE 13.3 Conceptual model of iron cycling through lake sediment resulting in net acidity generation

However, measured sulfate reduction rates were highly temperature dependent, averaging about 0.515 mol·m^{-2}·yr^{-1} during fall and winter months. These higher rates of sulfate reduction resulted in a 50% reduction in sulfate concentration within 2 km of the inflow of MIW and an increase in the pH of the surface water from 3.9–4.3 to 6.1–6.3.

MICROBIAL REACTIONS AND PIT LAKE PREDICTIVE MODELS

Current understanding of the microbial ecology and the significance of the biogeochemical reactions that microorganisms mediate in various pit lake environments is limited. It is now known from a few published ecological studies that diverse groups of organisms including green algae, bacteria, archaea, yeasts, and fungi are encountered in acid mine waters and pit lakes (Mitman 1999; Nixdorf et al. 1998). An important question then is how might the knowledge of microbiological reactions that are now posed be used predicatively and incorporated into pit lake water quality models?

Water Line

Numerous reports of iron sulfide oxidation rates are available in the literature and have recently been summarized by Blodau (2006). Coupled with an understanding of the distribution of pyrite or other sulfide-bearing rock in the pit walls, and estimates for the rate of pit filling, rate expressions for pyrite oxidation can be incorporated into dynamic systems models to help predict pit lake water quality. Microbial iron oxidation at the water line can contribute to the sustained oxidation of sulfide, and thus incorporation of rates for microbial oxidation of Fe^{2+} is important. A discussion of iron dynamics under the acidic conditions commonly found in MIW and appropriate rate laws can be found in Williamson et al. (2006). A better understanding of variation of oxidation rates at the water line and how periodic wetting and drying cycles of wall rock resulting from wave action would be useful for incorporation into pit lake models.

Water Column

In the water column, key biological reactions that have been evaluated are iron oxidation, primary production by photosynthetic organisms, and sulfate reduction. Consideration of rates of microbial iron oxidation was discussed previously, and the potential effect from heterotrophic activity such as sulfate and iron reduction are discussed in Chapter 19.

Our knowledge concerning the physical and chemical controls on primary production in pit lakes is limited, and additional research in this area is needed. These reactions can be significant for pit lake evolution because they may provide much of the organic carbon to fuel heterotrophic activity and the sedimentation of algal biomass has the potential to remove contaminants from the water column.

Available literature indicates that the main controls on primary production are TSS (e.g., water turbidity) and nutrient (N and P) availability, and potentially metal toxicity, which inhibits phytoplankton growth (Kalin et al. 2001). Geochemical and climactic conditions that may enhance or limit primary production can be evaluated and could include

- Chemical conditions that may affect TSS and water turbidity (such as groundwater pCO_2);
- Potential sources (groundwater and/or surface runoff inputs) of macronutrients (N and P);
- Iron concentrations and factors affecting phosphate co-precipitation;
- Amounts of littoral zones that will be available to support primary production.

A conceptual site model that includes these considerations would help to estimate the probable level of primary production for a given pit lake system. Published values for primary production from pit lakes considered to be similar to the lake being modeled could be input into the model to estimate organic carbon loading from primary production.

Sediment Column

Anaerobic microbial processes, such as iron and sulfate reduction, which add alkalinity and can remove metals from the aqueous phase, need to be more fully understood in pit lake environments. Several studies discussed previously in this chapter indicate that although these processes are occurring, the rapid cycling between reduced and oxidized species may diminish the overall impact of these reactions on the lake chemistry. In contrast, under eutrophic conditions where organic carbon is not limiting, sulfate reduction can have a significant impact on pit lake chemistry (Herlihy et al. 1987). A clearer understanding of the levels of labile organic carbon that are required to promote system changes will be required to adequately predict the effects of these processes. Recently a new rate law has been introduced for predicting microbial respiration in varied geochemical environments (Jin and Bethke 2003, 2005). The new rate law is reported to account for available energy to the microbial populations, and therefore can be applied over a spectrum of conditions from eutrophic to oligotrophic. Such a rate law coupled to estimates for diffusion of alkalinity out of the sediments and metal to the sediments will help to better define long-term water quality.

CONCLUSIONS

Numerous microbial mediated reactions have the potential to affect long-term pit lake water quality. Our understanding of the basic microbial processes such as iron oxidation, iron and sulfate reduction, and photosynthetic primary production is well established. Knowledge of microbial reaction kinetics and dynamics in the pit lake environment and ability to predict the effects of microbial processes on future water chemistry in pit lakes seems to be less well developed. In part, this stems from the difficulty in measuring microbial processes in situ. While methods for

TABLE 13.3 Case studies on microbial ecology and reactions in pit lakes

Pit Lake Name and Location	Reference
Berkeley pit lake, Butte, Montana, United States	Mitman 1999
B-Zone pit lake, Saskatchewan, Canada	Kalin et al. 2001
Mine Lake 111, Germany	Knöller et al. 2004

TABLE 13.4 General papers on biogeochemistry of pit lakes and reaction rates

Title	Reference
A review of acidity generation and consumption in acidic coal mine lakes and their water sheds	Blodau 2006
Pit lakes: their characteristics and the potential for their remediation	Castro and Moore 2000
Iron dynamics in acid mine drainage	Williamson et al. 2006
Predicting the rate of microbial respiration in geochemical environments	Jin and Bethke 2005

measuring the various inorganic parameters are generally well established, microbial enumeration and measurement of microbial activity in situ can be problematic. In addition, there is a current need to develop geochemical models that can more fully integrate microbial reactions. Current models (e.g., MINTEQA2, PHREEQC) are based on the principles of thermodynamic equilibrium that typically do not occur in natural waters between redox pairs, meaning our current modes of geochemical prediction do not accurately address these reactions.

An understanding of the various physiological groups of organisms and those environmental factors that control their activity, however, can nonetheless guide the development of predictive models. For example, initial geochemical modeling will allow prediction of whether a pit lake will likely evolve to be acidic or remain neutral or alkaline. If the model predicts the formation of an acidic lake, it is likely that acidophilic iron-oxidizing bacteria will thrive and incorporation of a component for microbial iron oxidation is appropriate. In contrast, under alkaline conditions, these processes will be less important and can possibly be excluded from the model. Thus, initial geochemical model predictions can evaluate which of the microbial processes will likely be important. Established rate laws for the microbial reactions determined to likely be of greatest importance can then be incorporated for model refinement.

USEFUL CASE STUDIES

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CHAPTER 14

Integration of Prediction Models

F. Werner

INTRODUCTION

The objective of this chapter is to give an example for a complex pit lake prediction using the MODGLUE program. MODGLUE integrates three popular limnologic, geochemical, and hydrologic programs to generate a water quality prediction. This model can be applied to similar cases.

Prediction models are different from models that are made to give an interpretation for observed phenomena (Luckner and Schestakow 1991). The reason is that the main impact factors are not known in their future intensity (i.e., storm frequency, wind speed, temperature, solar radiation) and that water management decisions (i.e., groundwater drawdown/rise, surface water input) often depend on factors that cannot be foreseen (i.e., regional planning, mine development, construction and maintenance work, legal permits). Thus, possible scenarios for future impact factors must be considered in predictive modeling.

Generally, modeling is not predominantly determined by the system (pit lake) itself and its complexity but rather by the questions that are asked about specific aspects of the system's performance. These questions are the starting point for the development of an adequate conceptual model. On the basis of a sound conceptual model, it is possible to decide on a solution pathway for the prediction task itself. Thus, modeling may be divided into three steps:

1. Define a question that creates the need for a prediction (objective). In most cases, the planning of an investment or regulative issues creates questions such as
   - When will the steady-state water level of a pit lake be reached, and how will it fluctuate?
   - What will be the future water quality of the pit lake?
   - What will be the effects of a remediation/management measure?
   - What are the effects of specific water management decisions?
2. Perform a system analysis and create a conceptual model. The pit lake characteristics (i.e., geology, climate, hydrology, limnology) are most important in this step but do not automatically lead to a certain conceptual model.
3. Generate a prediction using appropriate tools. These tools will be computer models in most cases. Another way is the extrapolation of trends, but the extrapolation of measured trends is not able to account for expected changes in the surrounding of the lake (i.e., groundwater rise, surface water management, etc.) as well as changes within the pit lake (i.e., when the depth is expected to increase).

The critical evaluation of these steps is an essential part of modeling. Mistakes in or neglecting the first step cannot be corrected by the modeling itself or be improved by high diligence in the third step. It is necessary to ask questions such as

- Is the problem that has to be solved clear and well defined?
- Is the system well understood?
- Is my conceptual model adequate with respect to the objective?
- Which available numerical models (i.e., computer programs) do what job?

These stated remarks on prediction models lead to the perception that no general rules can be given for the construction or integration of the models. Nevertheless, typical objectives do exist and will be addressed in the example that follows. From the experience with pit lakes from lignite mining in Germany, examples of using and coupling of numerical models are presented. An important fact about these pit lakes is that they are highly influenced by groundwater (Werner et al. 2001). If only hydraulic issues are to be addressed, lake internal processes may be neglected. This simplifying assumption may not be true for meromictic lakes where the monimolimnium layer exhibits a different water chemistry than shallower layers (see Chapter 5). A good water balance for a pit lake is a basic precondition for any elaborate model.

MODGLUE DESCRIPTION

Processes that may be considered in predictive pit lake modeling can be ordered into three groups:

1. Lake internal processes
2. Lake external processes
3. Lake-aquifer/soil boundary processes

In 1999, no single model was available that was capable of simultaneously modeling all of these processes. Therefore, a loosely coupled model was created and successively improved upon (DGfZ 2003; Mueller 2004). Changes were necessary to adapt the existing limnology model CE-QUAL-W2 (Cole and Buchak 1995) to handle the problems of acidic pit lakes. The resulting code was named MODGLUE (**MOD**el for Prediction of **G**roundwater and Erosion Influenced **L**ake Water Quality **U**sing **E**xisting Models). It consisted of three computer programs:

1. CE-QUAL-W2 (limnology);
2. PHREEQC (geochemistry); and
3. PCGEOFIM (hydrology).

The major changes to CE-QUAL-W2 were as follows:

- Changed alkalinity from an input to a model parameter;
- Included CO_2 gas exchange with the atmosphere; and
- Included carbon-limitation in nutrient/energy equations for the calculation of primary production.

The versatile geochemical program PHREEQC (Parkhurst and Appelo 1999) did not need to be modified. The problem definition was written using the BASIC-interpreter of PHREEQC and modifying its database. Major characteristics of the PHREEQC problem definition were as follows:

- Redox (oxidation–reduction) equilibrium was defined by the balance between electron donors and acceptors, and
- Iron chemistry was defined by the use of kinetic expressions for oxidation and oxide precipitation and solubility equilibrium with iron phosphate.

PCGEOFIM (Sames and Boy 1999) is a groundwater flow and transport model that was developed for mine water management in Germany. This program was originally used in MODGLUE.

In one application, a leaching module was added to MODGLUE to simulate the release of dissolved constituents from a waste rock dump situated along a pit lake shoreline. The basis for the ability of the waste rock to release dissolved constituents was investigated in lab experiments. The magnitude of release was calculated from the length of the shoreline involved. This parameter decreased significantly when temporary islands within the lake were submerged by rising water. If the water level did not rise above these islands, the specific amount of mass that was leached was eventually reduced to reflect the development of steady-state beach profiles after a certain time (Apel et al. 1980).

The MODGLUE code was initially used to model two lignite pit lakes in Germany, Lake Baerwalde (Werner et al. 2006) and Lake Bockwitz, and was subsequently modified to adapt to new tasks. For a third pit lake in Germany, Lake Runstedt, the groundwater program PCGEOFIM was exchanged with the program MODMST, which was designed to handle density-driven flow, multicomponent transport and reactions, and double porosity (Boy et al. 2001; Häfner and Boy 2005). In this case, groundwater flow and lake water mixing were linked by the possibility of density-driven flow between these two domains. Therefore, both limnology and groundwater programs were coupled to exchange data during the runtime. The structure of the revised MODGLUE program is shown in Figure 14.1.

The water balance for a pit lake is usually based on groundwater flow modeling. In Figure 14.1, the water fluxes between a pit lake and its surroundings are schematically drawn as arrows. The single water fluxes are analyzed for their water quality to calculate mass fluxes going in and out of the lake.

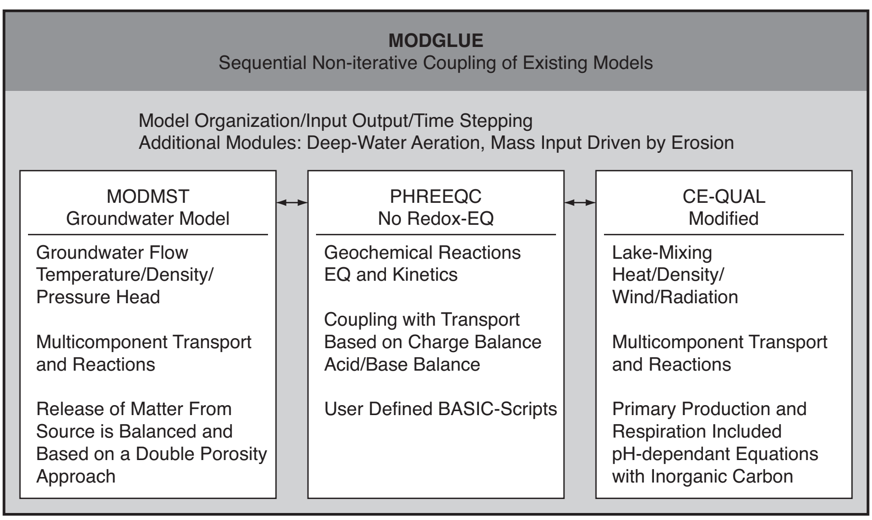

FIGURE 14.1 Structure of the revised MODGLUE program

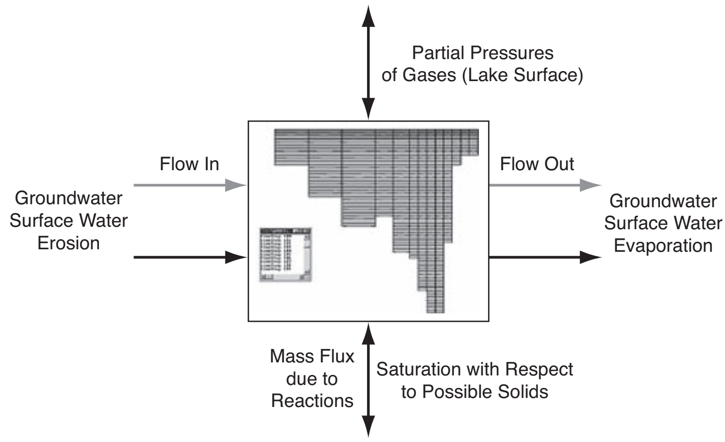

FIGURE 14.2 Representation of a fragmented reactor concept. A 2-D cross section of a pit lake is shown inside the box. Vertical lines separate horizontal fragments defined as sub-basins from lake bathymetry. Gray arrows indicate water fluxes between the pit lake and its surroundings. Black arrows indicate mass fluxes going into and out of the lake.

For the future magnitude and quality of the water fluxes flowing into a pit lake, assumptions have to be made. Figure 14.2 shows a two-dimensional (2-D) cross section of a pit lake divided in small boxes called fragmented reactors. To look at the whole pit lake as a zero-dimensional (0-D) reactor, or a point in space, and pose hydrochemical reactions (i.e., gas exchange, oxidation of metals, mineral precipitation, etc.) upon it often gives a good first-order approximation of the quantitative results of the posed reactions. The term *fragmented reactor* is used to describe conditions of either horizontal fragmentation into several lake basins or the vertical fragmentation into density-stratified layers. This approach refines the 0-D prediction by accounting for 2-D variations in chemistry within the lake.

Coupling of the limnology, geochemistry, and groundwater programs employed software engineering techniques such as object-oriented programming, design patterns, hybrid programming, and object databases. To construct MODGLUE, the existing programs were wrapped in a shell through implementation of the Python programming language. The wrapping utilized the design pattern adapter, turning the existing programs into Python objects. The other objects are pure Python objects. They have been implemented to account for a dual-porosity approach for release of acid mine drainage from aquifers and mine dumps and for the effects of bank erosion. The individual objects can be assembled in container objects. Each container uses the design pattern facade to give them a common interface for communicating with other objects. MODGLUE itself uses this pattern.

MODELING RESULTS FROM LAKE BAERWALDE, GERMANY

The water quality modeling for Lake Baerwalde using MODGLUE is described in work by Werner et al. (2006). The water quality parameters did not differ significantly between the epilimnion and hypolimnion layers. Oxygen was not depleted in the hypolimnion during periods of lake stratification. The water budget of Lake Baerwalde was dominated by surface water input.

The close match between predicted water quality and observed water quality shown in Figures 14.3 and 14.4 is largely dependent on the amount of surface water input specified. The presented

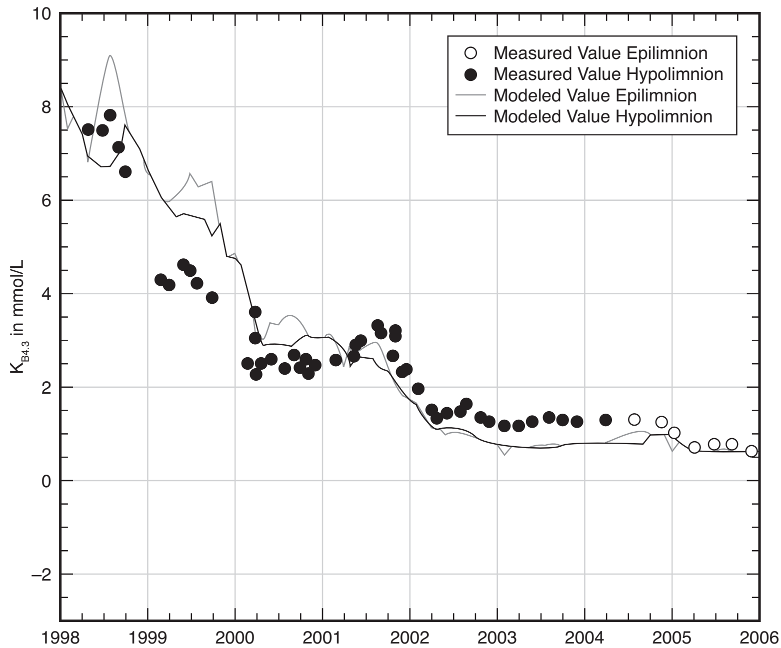

FIGURE 14.3 Predicted and observed acidity in Lake Baerwalde, Germany, from 1998 to 2006. KB4.3 is the acidity of epilimnion and hypolimnion water measured using a titration endpoint of pH 4.3.

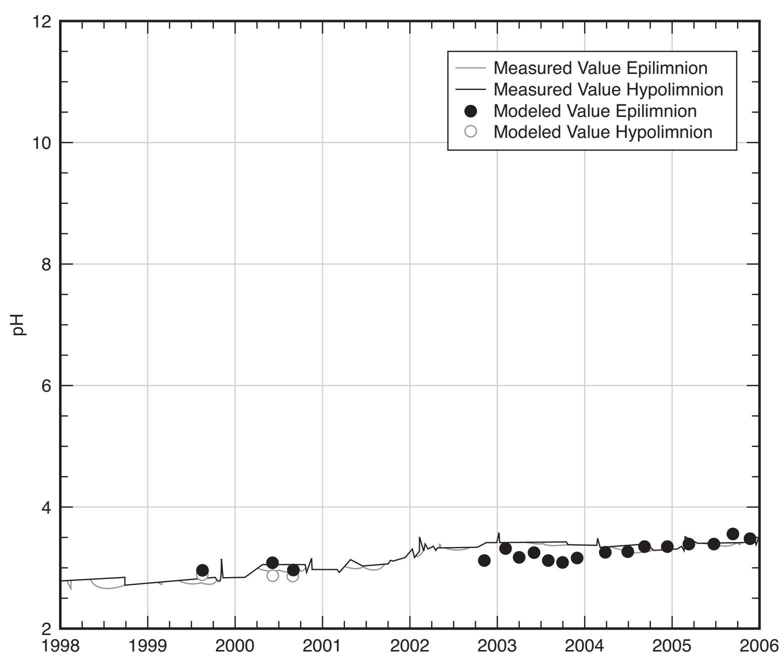

FIGURE 14.4 Predicted and observed pH in epilimnion and hypolimnion layers of Lake Baerwalde, Germany, from 1998 to 2006

TABLE 14.1 Data required to produce reasonably accurate results using MODGLUE

Groundwater	• Groundwater fluxes (flow model) • Inflow concentration • Groundwater transport model
Lake water	• Lake morphometry • Weather (wind, temperature, precipitation) • Algal growth and decay parameters • Possible solids for reactions in lake
Runoff/erosion	• Mass flux due to erosion/leaching • Sediment masses leached • Release of dissolved solids per unit of mass • Length of shoreline

data is a result of a model run using known surface water fluxes. The model was calibrated in 2003 using the soil properties for the release of acid from erosion and leaching as calibration values. The actual mass fluxes from erosion and leaching were calculated by the area of water/soil contact and the time of this contact. These parameters were not changed afterward in a second calibration. Only water fluxes were updated to produce the presented data.

DATA REQUIREMENTS

A considerable amount of data is needed to produce accurate results using MODGLUE, as shown in Table 14.1. Some data can be gathered by measurement and experiment. These include water quality data collected from the lake and from surface waters. Groundwater data is gathered with respect to the distinct recharge areas that are believed to yield distinctive water quality. Based on this knowledge, monitoring wells are positioned. Water fluxes in groundwater cannot be directly measured; rather they are extracted from groundwater flow models. Data for lake modeling also includes lake morphology, which is based on postmining surveys of the open pit and from echo sounding within the lake. Local weather data is gathered to model past conditions and a synthetic data set is created for future conditions (see Chapter 3). Extreme weather events and their effect on lake turnover cannot be predicted. Synthetic future weather data may be created by copying existing data from the past. However, the effect of climate change must be considered for long-range predictions by evaluating possible future weather scenarios (see Chapter 3).

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CHAPTER 15

Approaches for Evaluating the Predictive Reliability of Pit Lake Numerical Models

L.E. Eary and W.M. Schafer

INTRODUCTION

The permitting process for hard-rock, open pit mines that intersect the water table usually requires predicting long-term trends in water quality for the pit lake that will form after mine closure. The methods for predicting water quality in future pit lakes are technically complex and may involve integration of both mechanistic and empirical hydrologic, limnological, and geochemical models that are designed to represent the major processes that add and remove chemicals to and from the lake (Bird et al. 1994; Davis and Eary 1996; Miller et al. 1996; Kempton et al. 1997; Eary 1998). The methods are also often hypothetical because the mine and pit lake may not yet exist at the time of permitting, meaning that there are no site-specific monitoring data to calibrate the prediction. In addition, the definition of what is long term may vary from mine to mine because it is based on the time required for the hydrologic system surrounding the pit lake to reach a steady-state condition plus some additional time period to achieve geochemical steady state. At mines in arid regions of the western United States and other mining districts, rebound of the hydrologic system of a pit lake may take from 50 to 300 years. At lakes in humid climates or where artificial means are used to fill the lake rapidly after mining has ended, the time to reach hydrologic equilibrium may be only a few years.

As a result of the complexities of the systems being modeled and potentially long time frames, it is difficult to know a priori the reliability of a water quality prediction for any particular pit lake at the time of initial permitting and environmental assessment. This poses a problem for both the mines and regulators who have to assess the environmental impacts of pit lakes according to regulations that may vary from place to place (Bolen 2002). The purpose of this chapter is to provide approaches that may be used to assess the reliability of long-term predictions of pit lake water quality.

APPROACHES FOR ASSESSING RELIABILITY

Approaches for assessing the validity of pit lake predictions include the following:

- Model validation through direct comparison of model predictions to observed data;
- Model validation through comparison of model predictions to observed data from empirical batch tests;
- General trends in pit lake chemistry;
- Detailed studies of individual pit lakes; and
- Natural lakes as analogues for pit lakes.

Direct Comparison of Model Predictions to Observed Data

Ideally, the reliability of a model prediction is best tested through direct comparisons to observed data. However, there is not a long history in the scientific literature of making such comparisons for pit lakes for three major reasons. First, predictions may have been made before mining started. Thus, there may be a multidecade time lag between when the prediction was made and when mine closure and environmental monitoring begins. As a result, there are few long-term monitoring records to use for comparison to model predictions. Second, as a result of changes in the mine plan, the size, shape, and geochemical character of the pit walls may differ from the conditions initially modeled. Third, the science of pit lake predictive models is a relatively recent endeavor that did not become part of the published literature until the late 1980s and early 1990s when permitting of large, low-grade gold deposits in Nevada required predictions of future water quality (e.g., Davis and Ashenberg 1989; PTI 1992).

A few studies of direct comparisons of models to observed data are available in the literature. For example, Tempel et al. (2000) compared model predictions over an 8.5-year period to observed data for the North pit at the Getchell mine in Nevada (United States). Although there were only about 1.5 years of monitoring data available, their modeling results for most major solutes (pH, SO_4, HCO_3, total dissolved solids) showed good agreement with the observed data. However, predicted concentrations of arsenic were significantly greater in the epilimnion of the Getchell pit lake than observed. The mechanisms for the lower-than-predicted arsenic concentration were not fully identified by Tempel et al. (2000), but it was speculated that they might include unspecified mineral precipitation and adsorption reactions. In another modeling study of the Getchell mine, Davis et al. (2006) found good agreement between predicted and measured concentrations of As, Fe, SO_4, and pH in the Main pit lake. Davis et al. (2006) used a combination of hydrologic modeling to determine the rates of inflows to the pit lake from different sources and geochemical modeling of mineral equilibria and adsorption to ferric hydroxide for the mixture of inflows representative of the mature pit lake.

As an example for this chapter, a test of model reliability was made by developing a model based on the types of data normally available at the start of the mine permitting process and then comparing the model predictions to a multiyear record of monitoring data for a pit lake located in a former mine. The types of data used were very detailed and included meteorological records, measured groundwater compositions, and estimated chemical compositions for surface runoff and wall-rock leaching based on laboratory leaching data. The model also included mineral solubilities (calcite, gypsum, metal oxyhydroxides) and gas-phase partial pressures [CO_2(g) and O_2(g)] for a terminal, shallow, oxidizing, well-mixed lake based on the data from Eary (1999). The model was constructed using these data without any iterative calibration against observed data.

Comparisons of model predictions to measured concentrations showed good agreement over the period of record for most major ions (Ca, Mg, Na, K, SO_4, HCO_3) and pH (Figure 15.1). The model also correctly predicted the slow rate of increase in the concentrations of major elements on account of evaporation (this particular pit lake is a hydrologic sink). However, the model underpredicted the uranium concentration by a factor of about 10 to 20 (Figure 15.2). Concentrations of Ra and Se were also underpredicted by a factor of about 10 to 50. The primary source of these underpredictions is thought to be an underestimation of the rates of release of these elements from wall-rock leaching during the early years of filling the pit lake. Arbitrary amounts of these elements had to be added to the pit lake model to represent wall-rock leaching and produce a match to the observed trends. This comparison indicates that even in a well-characterized system with a relatively long monitoring record, predictions of major element chemistry can be made with good confidence, but it can be very difficult to quantify rates of minor element release

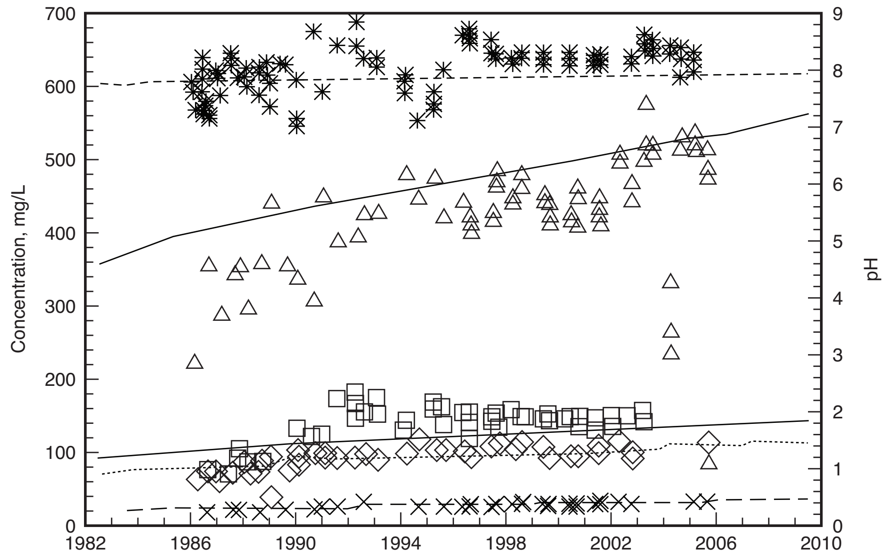


FIGURE 15.1 Comparisons of predicted (lines) concentrations of major solutes in a pit lake to observed (symbols) concentrations

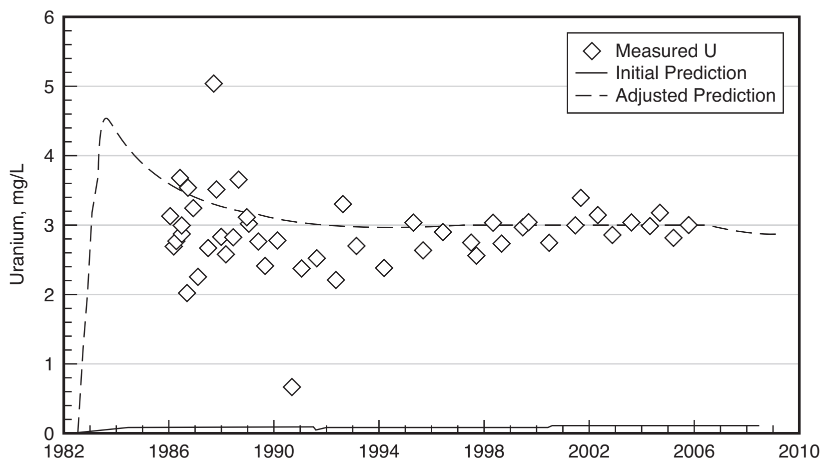


FIGURE 15.2 Comparisons of predicted (lines) concentrations of uranium in a pit lake to observed (symbols) concentrations

from kinetically controlled processes like wall-rock leaching. The extent of initial washoff rates of minor elements is a particularly difficult parameter to predict.

In other studies, Werner et al. (2006) were able to show good agreement between observed and predicted total acidities in a coal pit lake at Baerwalde, Germany, for a short time period (see also Chapter 14). Their model showed that the observed water quality could be explained by considering the effects on solute fluxes because of groundwater inflow, in-lake geochemical processes, and leaching of the pit walls (see Chapter 14). Balistrieri et al. (2006) reported good agreement between observed temperature and salinity depth profiles in the Dexter pit lake, Nevada, and predictions made with the Dynamic Reservoir Simulation Model (DYRESM). However, neither of these studies reported comparisons of predicted concentrations of minor metals (e.g., As, Cd, Cu, Fe, Hg, Mn, Pb, Se, Sb, Zn) to observed data, making it difficult to fully assess the overall reliability of their modeling approaches.

These few direct comparisons between model predictions and observed data provide some useful information on the strengths and weaknesses of numerical models of pit lake water quality. They show that predictions of trace metal concentrations are the least reliable in most cases, whereas predictions for major solutes are usually more reliable. Generally, it is the trace metals that pose the greatest potential risk, indicating that characterization of metal leaching processes and metal cycling in pit lakes are areas where resources should be focused to improve model reliability.

Comparison of Batch Test Simulations and Predicted Water Quality

Additional techniques are available for validating pit lake water quality predictions for existing mines. If representative samples of the solutions that will contribute to the pit lake during filling can be collected from the field or simulated, then a laboratory mixing experiment (batch test) can be used to simulate pit lake water quality. Schafer et al. (2006) used this technique for the Betze open pit located in north-central Nevada. A groundwater and surface hydrology model was used to calculate the proportions of water derived from various sources at a particular time stage of filling. Solution compositions characteristic of the inflow water sources were obtained from column test, natural groundwater, and meteoric water. The solutions were mixed in the laboratory and evaporated according to the proportions and rates predicted by the hydrologic model. After a period of equilibration, the batch solutions were then analyzed for dissolved ion concentrations and precipitated solids were examined by X-ray diffraction and scanning electron microscopy.

In the study of Schafer et al. (2006), major ions that did not precipitate from solution had good agreement between PHREEQC and the batch test (Table 15.1 and Figure 15.3). For virtually all other constituents that were removed from solution by precipitation or sorption, the agreement between measured and predicted concentrations was poor. For example, concentrations of Ca were underpredicted by PHREEQC because the more soluble form of calcium carbonate, aragonite, precipitated from solution rather than calcite. Some trace elements were overpredicted by the model, including Sb, Ni, Ag, Se, Tl, and Zn. Elements that were underpredicted by the model included Al, As, Ba, Cd, Cr, Cu, Fe, Pb, and Mn. Part of the poor agreement was the result of analytical detection methods that were higher than predicted concentrations for Cd, Cr, Fe, and Pb. The lack of thermodynamic data for specific mineral phases that might incorporate or adsorb the trace elements probably also contributed to differences between predicted and measured concentrations. Many of the solids predicted to form by PHREEQC were either absent or were so scarce or amorphous that they could not be detected. Only aragonite, calcite, and rare barite grains were found as crystalline precipitates in the batch tests. An iron-rich amorphous mass was also observed and contained a wide array of trace constituents. For this alkaline pit lake system where the majority of the metals reside as precipitates, improved calibration of geochemical models was required to obtain satisfactory predictions.

Davis (2003) also described a set of detailed batch mixing experiments designed to represent the chemical composition of a pit lake for different stages of filling. The mixing percentages were based on hydrologic model calculations of inflow rates from different sources. The chemical compositions for the hydrologic inflows were derived from a combination of field data and laboratory testing results. Davis (2003) found generally good agreement between the experimental

TABLE 15.1 Comparisons of measured and predicted ion concentrations in a batch test simulation of pit lake water quality

	Batch Laboratory Test, mg/L		PHREEQC Predictions		
Parameter	Inputs*, mg/L	Results for Measured Concentrations, mg/L	Concentrations, mg/L	Percentage of Input Concentrations Mass Balance	Percentage of Measured Results
pH		8.9	8.8		
Calcium	249	10.7	4.1	2	39
Magnesium	99	96	100	101	104
Sodium	196	190	196	100	103
Potassium	53	54.6	54	100	98
Sulfate	550	574	539	98	94
Bicarbonate	1,032	312	300	29	96
Chloride	39	42.5	39	100	92
Fluoride	3.9	3.0	3.9	100	131
Boron	2.0	1.9	2.0	100	105
Silica	117	96	117	100	122
Alkalinity	1,691	508	492	29	97
Aluminum	1.8	0.049	0.032	2	65
Antimony	0.027	0.012	0.025	90	204
Arsenic	0.174	0.016	0.0008	0	5
Barium	0.411	0.0312	0.0054	1	17
Cadmium	0.017	0.002 B	0.0017	10	87
Chromium	0.018	0.006 B	0.0007	4	12
Copper	0.031	0.0077	0.0007	2	8
Iron	30.095	0.02 B	0.0009	0	5
Lead	0.0050	0.001 B	0.00001	0	1
Manganese	3.321	0.0066	0.003	0	46
Nickel	0.429	0.065	0.293	68	451
Selenium	0.013	0.003	0.008	62	278
Silver	0.0135	0.005 B	0.007	54	146
Thallium	0.003	0.001 B	0.003	90	310
Zinc	2.224	0.0052	0.087	4	1,681
Solids:	Barite, calcite, barite, amorphous iron (ppt): +Fe, +SO_4, ±Al, ±Zn, ±Mn, ±SiO_2		Calcite, ferrihydrite, boehmite, barite, manganite, willemite, $Cr(OH)_3$		

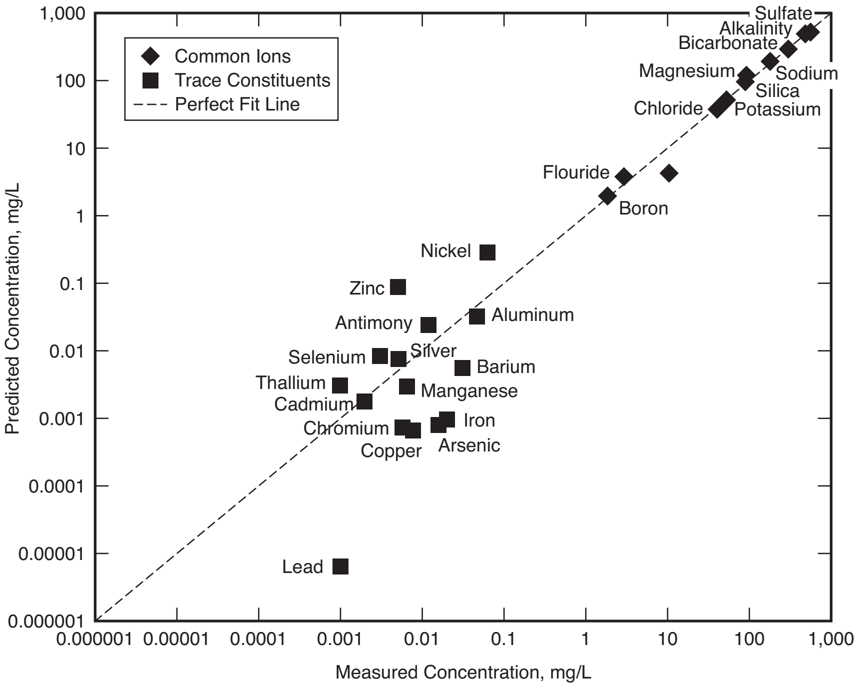
Source: Adapted from Schafer et al. 2006.
*Based on mass balance for the mixture of solutions used in the batch test.
PHREEQC predicted concentrations divided by the input concentrations.
PHREEQC predicted concentrations divided by the results for measured concentrations in the batch test.

results and PHREEQC calculations for most major ions and precipitates. Metal concentrations (Al, Cu, Fe, Mn, Sr, and Zn) predicted with PHREEQC showed a range of agreement with the experimental results.

In other laboratory-based studies, Totsche et al. (2003) and Uhlmann et al. (2004) conducted titration experiments with acidic water samples obtained from lignite pit lakes and compared the results to PHREEQC calculations. These studies found that PHREEQC could be used to identify the principal acid buffering reactions controlling the pH of the pit lake water. The reactions include Fe and Al hydroxide precipitation, Fe and Al hydroxyl-sulfate precipitation, and exchange of SO_4, Ca, Mg, and H^+ on the ferric hydroxide surface. These results provide useful information on the phases controlling solution chemistry and acid-buffering processes in acidic pit lakes.

General Trends in Pit Lake Chemistry

Compilations of water quality observed in numerous pit lakes offer another means to develop guides about what kind of water quality might be expected at any particular pit lake based on



Source: Adapted from Schafer et al. 2006.

FIGURE 15.3 Comparison of measured and predicted ion concentrations in a batch test simulation of pit lake water quality

geology, mineralogy, and local climate. For example, Shevenell et al. (1999) identified factors related to ore deposit geochemistry and mineralogy that are important for affecting water quality at 16 pit lakes in Nevada. They determined that pit lakes formed in quartz-alunite precious metal deposits in volcanic rocks and in porphyry copper-molybdenum deposits in plutonic rocks have the highest potential for developing poor water quality. This trend is due to the tendency of these deposits to have high acid-generation potentials and low amounts of carbonate rocks that might buffer acidity. As a result, there is a greater potential for acidic pH conditions in these deposits, which lead to greater rates of metal leaching.

Although these are general trends that can be overridden by local meteorological, hydrological, and surface geological conditions, they provide a means to evaluate the results of a pit lake modeling effort. For example, if a pit lake model predicts a near-neutral pH and low metal concentrations for a pit lake formed in a porphyry copper-molybdenum deposit, then it may warrant additional examination of the factors controlling water quality to determine why the prediction is counter to the trend observed for that ore deposit type. For example, local mineral zonation and hydrothermal alteration types may be more important for controlling water chemistry than the overall deposit type.

In another study of broad trends, Eary (1999) compared observed concentrations of individual solutes from a variety of pit lakes to concentrations that would be predicted by the solubilities of different minerals that would be expected to exist in pit lake systems. The use of mineral solubilities to represent solute concentrations is a standard geochemical modeling approach for understanding natural aqueous systems (Langmuir 1997; Merkel et al. 2005; Nordstrom and

Alpers 1999; Zhu and Anderson 2002). For pit lakes, this approach was found to be useful for representing observed concentrations of major solutes (Al, Ca, Fe, HCO_3, Mn, pH, and SO_4) but was much less successful for many trace metals (e.g., Cd, Cu, Pb, Zn). In addition, specific mineral phases controlling As and Se were not apparent, although predictions based on adsorption to ferrihydrite roughly approximated the general trends in concentrations of these elements.

These types of broad comparisons show that theoretical calculations of chemical equilibria can provide a reasonable basis for predicting acid–base and major solute chemistry. However, reliable prediction of trace metal concentrations based on theoretical calculations is problematic. If the pH is correctly predicted, then there is a good chance that maximum metal concentrations may also be reasonably well predicted because most mineral solubilities and adsorption reactions are strongly dependent on the pH (Drever 2002). However, if the pH is incorrect, then predictions of metal concentrations may be significantly in error. In addition, it is clear that kinetic processes control metal concentrations in many pit lakes, pointing out the importance of including empirical data on rates of metal leaching from wall rocks in pit lake predictive models.

Detailed Studies of Existing Individual Pit Lakes

Detailed studies are generally focused on a single pit lake or single geochemical aspect of a pit lake. They can provide information on specific geochemical processes that may be occurring in the water column, wall rocks, and sediments. For example, Levy et al. (1997) determined that mineral solubilities (gypsum, goethite, jarosite, and jurbanite) were important for controlling major solute chemistry in an acidic pit lake in a porphyry copper deposit, whereas seasonal cycles of filling and evaporation, and low levels of organic carbon were more important for affecting trace metal concentrations. Triantafyllidis and Skarpelis (2006) reported the presence of a large number of secondary precipitates in the sediments of an acidic pit lake in a high-sulfidation porphyry molybdenum deposit, including anglesite, jarosites, rozenite, melanterite, wroewolfeite, gypsum, bukovskyite, beaverite, scorodite, and goethite. Detailed studies of the acidic Berkeley pit lake (porphyry copper deposit) have also pointed out the importance of secondary mineral precipitation (gypsum, jarosite, and ferrihydrite) and redox (oxidation–reduction) equilibria in combination with density stratification for affecting metal concentrations (Davis and Ashenberg 1989; Gammons and Duaime 2006; Pellicori et al. 2005; Twidwell et al. 2006). In other studies, the importance of limnological stratification for controlling redox chemistry has been demonstrated in studies by Doyle and Runnells (1997), McNee et al. (2003), Ramstedt et al. (2003), and Wilton and Lawrence (1998).

Natural Lakes as Analogues for Pit Lakes

Hydrologic models predict that many pit lakes located in arid regions will become evaporative sinks over the long term. Many natural lakes located in hydrologically closed basins of the western United States are also evaporative sinks with chemistries that have evolved over long periods of time. The water qualities observed in the natural lakes in arid regions can provide analogues of what may be expected for mine pit lakes in arid regions. However, the same analogy may not necessarily be true for pit lakes in humid climates where rainfall exceeds evaporation, resulting in flow-through hydrological conditions.

There are a few reported rare instances of acidic natural lakes, such as in the Andes of northern Chile (Risacher et al. 2002), volcanic terrains (Delmelle and Bernard 1994), and in association with massive sulfide deposits (Kwong and Lawrence 1998). The acidity in these lakes results from the oxidation of sulfide and native sulfur deposits naturally exposed in the lakes or their watersheds, similar to mine pit lakes. Evaporation of these lakes can result in concentrated brines and precipitation of Al and Fe oxyhydroxides, hydroxyl-sulfates, and various sulfate evaporite minerals (Risacher et al. 2002).

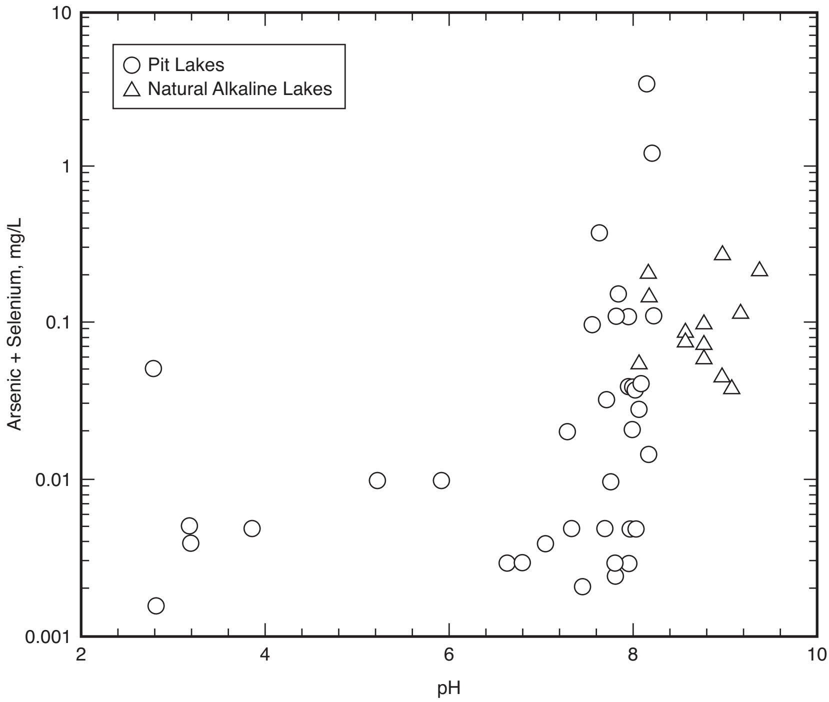
However, most natural evaporative lakes in the western United States are alkaline with compositions dominated by Na-HCO_3-CO_3-($\pm SO_4$)-($\pm$Cl). Similarly, most pit lakes in the western United States are also neutral to alkaline (Eary 1999). Under neutral to alkaline pH conditions, the concentrations of cationic metals (e.g., Cu, Cd, Fe, Mn, Zn) are very low because of solubility constraints and adsorption reactions (Drever 2002). However, anionic metalloids (e.g., As and Se) tend to be naturally elevated because they are not limited by solubility or adsorption reactions under alkaline pH conditions as they are under near-neutral to acidic pH conditions. This effect can be seen in a comparison of As and Se concentrations measured in natural lakes to existing pit lakes reported (Figure 15.4). These trends have important implications for predictive pit lake modeling because the long-term pH evolution of an evaporative pit lake is dependent on its relative concentrations of Ca and alkalinity.

The concept of chemical divides developed by Eugster and Hardie (1978) explains the evolution of evapoconcentration effects in natural lakes and is also useful for understanding how the composition of terminal pit lakes may change over time in arid climates. The chemical divides theory predicts that lakes with 2[Ca] < [alkalinity] (brackets indicate molalities) will evolve to alkaline pH, Na-HCO_3-CO_3-($\pm SO_4$)-($\pm$Cl)-dominated solutions. In contrast, pit lakes with 2[Ca] > [alkalinity] will evolve to near-neutral pH, Ca-Na-SO_4-($\pm$Cl)-dominated solutions. The distinction between these two evapoconcentration paths is important for modeling future pit lake chemistry because it has a fundamental effect on the pH, which in turn affects the prediction of metals concentrations. For example, if a pit lake model predicts that a pit lake will evolve to an alkaline pH condition but with low concentrations of As and Se, then the conceptualizations controlling the release and removal of these metalloids may need to be carefully re-evaluated because the result is inconsistent with observations of natural lakes.

DATA GAPS IN ASSESSING RELIABILITY

In the scientific literature are a few examples where predictions of pit lake water quality have been directly compared to measured data over a time period of more than a few years. This lack of comparisons is due in part to the fact that many mines for which pit lake predictions have been made are still in production. However, other pit lakes are in the process of filling with water and are likely still being monitored on a quarterly to yearly basis as part of mine closure. Where possible, investigations should be conducted to compare what may have been predicted for future water quality for these pit lakes to the monitoring data. Comparisons made for the initial period of filling may be especially useful because they could provide information on the effects of flushing of solutes from the pit walls, an aspect of pit lake modeling that is very difficult to represent in a numerical model. Miller (2002) has pointed out the importance of underestimating solute flushing at the Cove pit lake in Nevada.

Another approach that may be considered for pit lakes that do not have long-term prediction of water quality is to examine characterization data. Shevenell et al. (1999) have shown that pit lake water quality can be related to ore deposit type, mineralogy, and geology. Other data on acid–base characteristics of wall rocks and other testing data on wall-rock reactivity can be related to water quality similar to the approach described by Shevenell et al. (1999). For example, Castendyk et al. (2005) used a detailed spatial characterization of wall-rock mineralogy to evaluate acid–base reactions in a future pit lake. To better define these types of relationships, a database of water quality for existing pit lakes needs to be developed. The compilation and assessment of water



Source: Adapted from Eary 1998.

FIGURE 15.4 Summed concentrations of As and Se for 18 existing pit lakes (38 analyses) and 10 naturally alkaline lakes (19 analyses) in the western United States

quality data for existing pit lakes provides one of the best approaches for providing guidelines on chemical trends for predictive models.

Most of the current state of knowledge of pit lakes has come from generalized observations of different geochemical, biological, hydrologic, and limnologic processes occurring in existing individual pit lakes. These studies have proven their importance for providing information on how to improve the conceptualizations that are incorporated into pit lake models. However, few detailed studies have been conducted on existing hard-rock pit lakes. In particular, studies of sediment mineralogy are needed to provide information on the relative importance of secondary mineral formation on metal cycling. Studies of biological productivity and organic carbon in pit lakes in combination with improved understanding of the effects of density stratification are also needed to provide information of what factors control redox chemistry. Better conceptualizations developed from detailed studies of individual pit lakes should lead to improved reliabilities of pit lake predictive models.

CONCLUSIONS

Investigations of the reliability of pit lake predictive models are scarce in the published literature. A few available studies and general trends in existing pit lakes indicate that trends in major solute chemistry can generally be predicted with reasonable confidence using standard equilibrium relationships incorporated into geochemical models. These types of studies are useful for identifying

TABLE 15.2 Case studies with comparisons of model predictions to observed data

Pit Lake Name, Location	Reference
Coal pit lake, Baerwalde, Germany	Werner et al. (2006)
Lignite lakes, Lusatia, Germany	Totsche et al. (2003), Uhlmann et al. (2004)
Getschell pit lake, Nevada, United States	Tempel et al. (2000)
Getschell pit lake, Nevada, United States	Davis et al. (2006)
Dexter pit lake, Nevada, United States	Balistrieri et al. (2006)
Unknown equatorial lake	Davis (2003)
Betse-Post, Nevada, United States	Schafer et al. (2006)

TABLE 15.3 Studies of broad trends in pit lake water quality

Pit Lake Name, Location	Reference
Hard-rock pit lakes, western United States	Shevenell et al. (1999)
Hard-rock pit lakes, western United States	Eary (1998, 1999)
Hard-rock pit lakes, western United States	Price et al. (1995)
Hard-rock pit lakes, western United States	Miller et al. (1996)
Hard-rock pit lakes, western United States	Davis and Eary (1996)
Coal pit lakes, Germany	Klapper and Schultz (1997)

when a predicted water quality may be consistent with general trends but do not provide information on potential accuracy of a prediction for any particular pit lake. Predictions of metal concentrations, especially for the more mobile metal solutes, such as As, Cd, Cu, Mn, Se, U, and Zn, are problematic because of difficulties in quantifying all the factors that may influence sources and sinks for metals in pit lake systems. There is a critical need for investigations focused at testing the reliability of pit lake predictive models through direct comparisons of model results to monitoring data with the purpose of identifying where models succeed and fail. This need was previously identified by the National Research Council (NRC 1999). To allow these types of investigations to be conducted, there is also a need to monitor existing pit lakes for extended periods of time and allow researchers to access those data to allow pit lake predictive models to be improved.

USEFUL CASE STUDIES

A search of the published literature for articles that have some aspect of comparing model results to measured water qualities yields the references listed in Tables 15.2 and 15.3.

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CHAPTER 16

Significance of Biological Productivity of Pit Lakes for Interpreting Ecological Risks

T.E. Hakonson, V.F. Meyer, and A. Dean

INTRODUCTION

Most ecological risk assessments involving pit lakes focus primarily on the chemical concentration data and the associated toxic risks to resident or migrant biota. Toxic risks are calculated by comparing measured (or calculated) concentrations in aquatic samples with corresponding benchmark concentrations developed from controlled dose-response studies. Benchmark concentrations adopted by most federal and state regulatory agencies generally represent the lowest concentrations (e.g., in water, sediment, or food) that produce toxic effects in the most sensitive populations (e.g., caddis flies). As such, benchmark concentrations are protective for all organisms of a particular kind, including those less sensitive to the chemical of interest.

However, the process of assessing risks to biota in pit lakes requires integration and interpretation of multiple layers of chemical, physical, and biological data (EPA 1997; Owen 1985; Suter et al. 2000). The overall interpretation of risks must include not only estimates of the chemical risks resulting from mining influenced water (MIW), but also data on the (1) characteristics and quantity of the physical habitat containing the MIW, and (2) biological relationships including taxonomic composition, trophic structure, and biomass of species that might be exposed to the MIW (Figure 16.1). The focus of this chapter is to discuss and provide examples of the importance and sometimes dominating role that physical and biological data play in contributing to the final evaluation of overall risks.

Our approach in this chapter is to discuss relevant limnological concepts that apply to many existing pit lakes and to use field measurements from a pit lake in the western United States to illustrate methods and data needed to (1) quantify some of the important physical and biological relationships at a pit lake, and (2) illustrate, through energetic calculations (based on food consumption rates), the potential impact of habitat and biological relationships on the interpretation and overall significance of the chemical risk assessment.

IMPORTANT LIMNOLOGICAL CONCEPTS THAT APPLY TO PIT LAKES

Many pit lakes, particularly in the dry, cold western United States, are relatively young (i.e., a few decades old) and in an early stage of biological development. Many of these lakes, which are still in the process of filling, are classified by biological productivity as oligotrophic. Oligotrophic lakes are characterized as containing very low concentrations of nutrients, such as phosphorus,

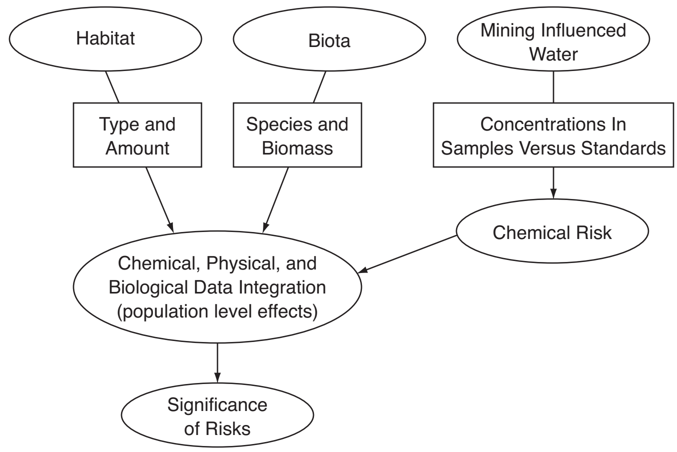

FIGURE 16.1 Data layers required to fully assess ecological risks to pit lake biota

required for plant growth so that the overall productivity of these lakes is low (Wetzel 1983; Brunberg et al. 2002).

Nutrients present in pit lakes may have several sources. These include external sources such as runoff and erosion into the lake and internal sources such as groundwater intrusion, decaying biological materials, and chemicals and metals associated with the mining operations, including blasting residues. The relative importance of these various sources of nutrients is not well known but, in any event, will be dependent on a variety of site-specific conditions. However, in the case of oligotrophic lakes, the low nutrient levels that are typical of these lakes suggest that nutrients additions, regardless of source, are low.

Because only a small quantity of organic matter is produced in an oligotrophic lake, phytoplankton, zooplankton, attached algae, macrophytes (aquatic weeds), bacteria, and fish may be present but only in limited numbers of individuals. There may be many species and types of organisms but very little biomass of each species or type.

With so little production of organic matter, there is very little accumulation of organic sediment on the bottom of oligotrophic lakes (Wetzel 1983; Brunberg et al. 2002). Thus, with little organic food, only small populations of bacteria and other microscopic organisms exist that can serve as food for some species of zooplankton. Moreover, with only small numbers of plankton and bacteria, there is very little consumption of oxygen from the water. One typical trait of an oligotrophic lake is that it contains high oxygen concentrations from surface to bottom.

The physical shape of a lake, including pit lakes, is an especially important attribute that has a major impact on the type and amount of habitat available for various kinds of biological productivity. Oligotrophic lakes are generally steep (to very steep) sided with a very limited band of littoral zone for supporting aquatic vegetation (Wetzel 1983; Brunberg et al. 2002; Welch et al. 2004). The littoral zone is defined as that area close to shore where light reaches to the bottom (Figure 16.2). The primary producers in the littoral zone are plants rooted to the bottom and algae attached to the plants and to any other solid substrate.

The limnetic zone (Figure 16.2) is the layer of open water where rooted vegetation cannot become established on account of the depth of the water column. The upper layer in the limnetic zone, where sunlight can penetrate to drive photosynthesis, is called the euphotic zone. Life in the

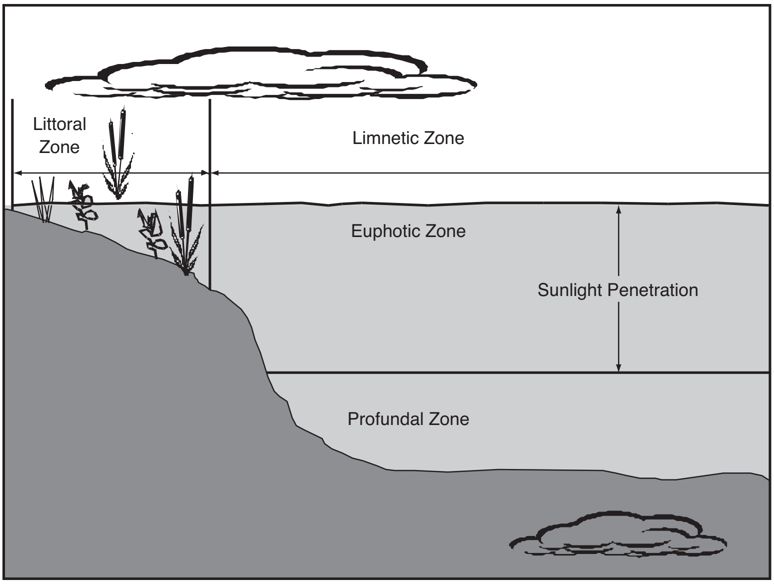

FIGURE 16.2 Conceptual limnological zones associated with biological productivity in lakes

euphotic zone is dominated by floating microorganisms called plankton, which includes actively swimming animals. The primary producers in the limnetic zone are planktonic algae.

The primary consumers may include such animals as microscopic crustaceans and rotifers called zooplankton. The secondary (and higher) trophic-level consumers may include swimming insects and fish. The euphotic zone is shallower in turbid water than in clear because of decreased light penetration.

As one descends deeper in the limnetic zone, the amount of light decreases until a depth is reached where the rate of photosynthesis becomes equal to the rate of respiration. At this level, net primary production no longer occurs. The profundal zone is the region below effective light penetration wherein photosynthetic processes contributing to primary productivity are very low. This zone depends on the contribution of organic matter from the littoral and limnetic zones for its source of food energy. The profundal zone is chiefly inhabited by primary consumers that are either attached to or crawl along the sediments at the bottom of the lake. In oligotrophic lakes, where production of organic matter is limited, biological activity in the profundal zone is also limited.

Conclusions that can be drawn about the characteristics of oligotrophic lakes are as follows:

- Generally deep,
- Generally clear water,
- Relatively high saturated oxygen levels,
- Steep sided with a low ratio of littoral zone to limnetic zone,
- Nutrient poor, and
- Low plant and animal productivity.

Most of the characteristics of oligotrophic lakes apply to many mine pit lakes (Horne and Goldman 1994). These commonalities have a major bearing on the degree and significance of contaminant transport in pit lakes and the associated ecological risks to resident and migratory wildlife populations. To illustrate the importance of these similarities in evaluating ecological risks, the following sections describe habitat and biological data that were collected at an oligotrophic, slightly alkaline pit lake in the western United States and the use of that data in energetic calculations to support interpretation of ecological risks.

EXAMPLE OF ECOLOGICAL RISK ASSESSMENT

Conceptual Models for Risk Assessment

In assessing ecological risks at a pit lake, a conceptual model is usually developed to visually represent the biological components potentially at risk from MIW as well as the mechanisms whereby these organisms could be exposed (Figure 16.3). Conceptualization of the structural and functional relationships, including exposure pathways, is used to identify data needed to evaluate the potential risks from MIW to pit lake biota. From the model in Figure 16.3, data needs would include

1. Specific metals and anions and applicable standards for chemicals of interest;
2. Biota that might be exposed to the MIW, including exposure pathways;
3. Concentrations of metals and anions in physical and biological components;
4. Physical habitat and biological attributes of the lake ecosystem, such as species composition, biomass, surface area, or volume of the various lake components; and
5. A methodology for integrating all the information and data to estimate risks to biota (and humans) that might be exposed to MIW at the pit lake (see Figure 16.1).

Habitat Data

Bathymetric data and field measurements were made on a pit lake in the western United States to evaluate the kinds and amounts of habitat available to support resident and migratory wildlife. Results showed that the pit lake, which was still filling with water, had a shoreline comprised of very steep slopes ranging from near vertical around most of the perimeter to a very small area of littoral zone with more gentle slopes of about 10% to 15%.

For characterization purposes, the littoral zone at this pit lake was defined as the zone from shore to a maximum water depth of 1.5 m. This definition was based on the fact that water depths increased dramatically beyond the 1.5-m depth in the littoral zone owing to the near-vertical side walls associated with the pit-shaped configuration of this lake. For example, water depths exceeding 1.5 m were achieved within 1 m of the shore for all but a small portion of the shoreline.

The total area of littoral zone habitat, as defined previously, comprised only about 0.4% of the total lake volume and 4% of the surface area. However, because of the steep sides of the lake, only 1.5% of the littoral zone contained rooted and submerged macrophytes. This vegetated area covered only about 0.08% of the total lake surface area. Based on future projected lake level and the topography to be covered by the rising water, the amount of littoral zone is not expected to change appreciably from conditions at the time these measurements were made.

The open water zones (i.e., water depths exceeding 1.5 m) comprised about 99% of the volume and 96% of the surface area of the lake. The lake had a maximum depth of more than 30 m during the study and was well oxygenated from top to bottom (oxygen levels exceeded 70% of saturation throughout the year). In contrast to the clear water that is associated with most oligotrophic lakes, light penetration through the water column of this pit lake was limited to about 1.5 m as measured with a Secchi disk. Although the reasons for the turbidity of the lake water are unclear (may be related to CO_2 levels), it is certain that the low light penetration limits primary production requiring photosynthesis.

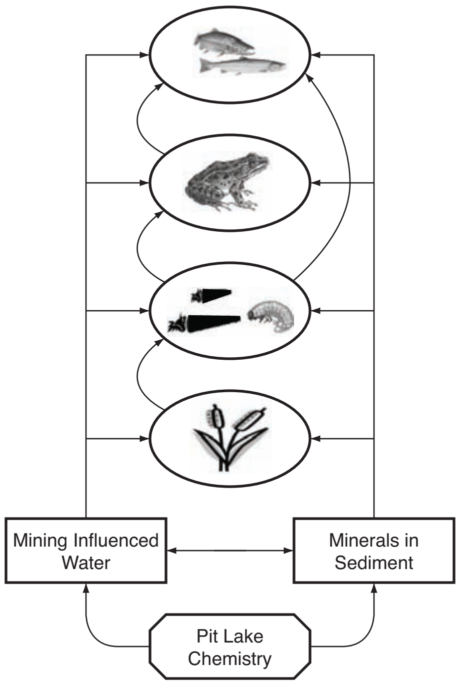

FIGURE 16.3 Conceptual transport pathways to biota from pit lake chemistry

Results of measurements of the physical habitat at the pit lake demonstrated that very little littoral or limnetic zone habitat existed to support biological production in keeping with the definition of oligotrophic lakes. The assumption of low biological productivity is demonstrated in the following sections using quantitative biomass estimates for benthic invertebrates and copepods (a small crustacean zooplankton) to measure standing crops of these organisms at various times during the year.

Biological Data

It is noteworthy that resident aquatic species that come to inhabit a pit lake have either been introduced from other areas by natural processes or have been introduced by humans. If these introduced species survive in a pit lake and populations develop, then it could be concluded that these surviving species are tolerant of the ambient chemical conditions in the lake. The standing crop of any one species or assemblage of species in a pit lake is determined by many factors in addition to chemical/metal loads. Important additional factors would include food and habitat availability.

Zooplankton. At the pit lake under study, a small (i.e, 1 to 2 mm) free swimming copepod (*Cyclops* sp.) was the primary consumer in the open water zone. Copepods, depending on species, feed on microscopic plants and animals including algae, rotifers, and bacteria and can serve as food sources to a variety of resident fish and migratory waterfowl. Copepod biomass was estimated at various times of the year using a 150-µm mesh haul net retrieved vertically in the water column. Multiple discrete samples were taken at each of several locations across the lake at each sampling to obtain spatially integrated copepod biomass estimates. Because the dimensions of the haul net and depth to which it was lowered were known, the mass of copepods in the haul net

TABLE 16.1 Summary statistics for total copepod biomass estimates (in kilograms) at an oligotrophic pit lake in the western United States

Sample Date	Mean*	SD	Mean–95% UCL	Mean +95% UCL	n
February	298	166	147	450	6
March	400	82	318	482	5
April	295	0			1
July	728	403	206	1,249	3

*Samples were obtained with a 150-µm vertical haul net.
SD = Standard deviation.

TABLE 16.2 Summary statistics for benthic invertebrate biomass (in kilograms) estimates in the littoral zone of a western United States pit lake

Sample Date	Mean*	SD	Mean –95% UCL	Mean +95% UCL	n
February	25	32	13	45	21
March	20	28	9	37	21
July	36	36	22	59	23

*All of this biomass was associated with the 0.9 ha of the pit lake containing rooted macrophytes.
SD = Standard deviation.

could be converted to total biomass in the lake. This was accomplished by multiplying copepod mass per liter of water sampled by the total liters of the pit lake water column that contained copepods. Special studies were done to determine the lowest depth at which copepods were found in the lake in order to estimate the volume of the lake occupied by copepods.

Mean total copepod biomass estimates for the entire lake ranged from a minimum of about 300 kg to a maximum of slightly more than 700 kg (Table 16.1). The 95% UCL (upper confidence limit) estimates ranged from 450–1,249 kg. The highest biomass estimates were measured during mid-summer, as would be expected with the warmer water temperatures.

Benthic invertebrates. Benthic invertebrate biomass was estimated using an Ekman dredge to sample a given area on the sediment surface in the littoral zone. The open face of the dredge was lowered to the sediment surface and the closure jaws activated to collect sediment and benthic organisms contained in the sediment. Each sample was transferred to a 60-mesh screen and hand sorted to retrieve all benthic invertebrates. Samples were weighed, oven dried, and then reweighed. Voucher specimens were also taken to identify the species collected.

Results showed that although species richness was relatively high, the total benthic invertebrate biomass in the entire littoral zone averaged only 25–36 kg based on measurements made on three different sampling dates and for sample sizes for each date ranging from 21 to 23 (Table 16.2). The 95% UCL biomass estimates ranged from 37 to 59 kg. Virtually all of the benthic invertebrate biomass collected during the sampling was associated with the small area of the lake containing rooted macrophytes.

IMPLICATIONS OF BIOLOGICAL PRODUCTIVITY FOR RISK ASSESSMENTS

In assessing ecological risks at a pit lake, chemical concentration data and relevant regulatory standards must be evaluated in light of the number of individuals of the potentially exposed population that can be supported by the available habitat and food sources. As discussed previously, pit lakes that exhibit characteristics typical of oligotrophic lakes may provide little habitat for

TABLE 16.3 Energetics calculations to show the number of consumer organisms that could be supported by benthic and zooplankton food sources at a pit lake in the western United States

Consumer	Consumer Weight, kg	Food Ingestion Rate, kg/yr* (1)	Avialiable Prey Biomass , kg (2)	No. Consumers Supported/year (1) (2)	No. Consumers Supported/month [illegible]
Mallard duck	1.0	18 (100% benthic invertebrates)	12–17	0.7–0.95	8–11
Fish	1	18 (100% copepods)	135–375	8–21	96–250

[illegible]
Assumes 30% of 95 UCL biomass of copepods (Table 16.1) or benthic invertebrates (Table 16.2) available for consumption.

primary and secondary biological productivity. This low productivity, as measured by plant and animal biomass, provides limited opportunity for lake contaminants to be transferred via food to more than a few individual organisms.

To illustrate those relationships, assume that a mallard duck spends all or part of the year at the pit lake. Also assume that the duck weighs 1 kg and that its diet consists entirely of benthic invertebrates. Given that a 1-kg duck consumes about 5% of its body weight per day (adapted from EPA 1993), the duck would need to ingest 50 g of benthic invertebrates per day or about 18 kg over a year (Table 16.3). Based on the 95% UCL benthic invertebrate biomass estimates ranging from 39 to 57 kg and a benthic invertebrate population that was subjected to a 30% predation rate (i.e., predation must be low enough to maintain a viable benthic invertebrate population), the existing benthic invertebrate population in the lake would annually yield 12 to 17 kg of available food for a duck. Given these assumptions, calculations showed that the benthic invertebrate population at the pit lake would support less than 1 duck per year or about 10 ducks for a month. The conclusion that can be drawn from these calculations is that the pit lake provides a very limited food base to support more than a few ducks for a limited time.

Another illustration uses rainbow trout assumed to consume a diet of 100% copepods. If one trout weighs 1 kg and consumes 5% of its body weight in food per day (adapted from EPA 1993), then 18 kg of copepods would be required as a trout food source per year. Given that the 95% UCL copepod biomass for the pit lake ranged from 450 to 1,250 kg for the entire lake and assuming a 30% predation rate on copepods, from 135 to 375 kg of copepod biomass would be available for consumption by the single trout. Over the course of a year, copepods would support from 8 to 20 trout. However, given natural mortality and possible predation by fish-eating predators (i.e., osprey, mink), it is unlikely that a viable trout population could be maintained by the amount of food provided by the existing zooplankton population in this pit lake.

Other pathways, such as direct contact and drinking water exposures may become important in pit lakes where contaminant concentrations in water are highly toxic. However, this is not the usual situation so that under most conditions, food chain exposure pathways usually dominant in contributing risks to aquatic biota. When food chain transfer of contaminants to biota dominate, habitat and biological productivity become important contributors to overall ecological risk for pit lake contaminants.

CONCLUSIONS

Calculations, based on energetic requirements of representative consumers, suggest the following:

1. The lack of aquatic habitat and associated biological productivity at an oligotrophic pit lake may provide a very small potential for transfer of pit lake contaminants to significant numbers of migrant or resident wildlife including waterfowl and fish.
2. Depending on topographical features of the shoreline to be inundated as the lake level rises, the amount of littoral zone and associated plant and animal productivity may not change with time.
3. Although a pit lake and surrounding area may be used extensively by wildlife as resting areas for migrant waterfowl, summer range for mule deer and other species, and seasonal hunting habitat for avian and mammalian predators, the importance of this use on the toxic risks from pit lake chemistry to these populations will depend on the amount of habitat and biomass available as food sources.

It is important to repeat that resident populations of aquatic species at most pit lakes have invaded these lakes as they filled and developed resident populations under the existing and prior chemical conditions in water and sediments. Therefore, these species have tolerated chronic exposures to the solutes present in pit lake and other pit lake–related conditions.

The results presented in this chapter suggest that a realistic evaluation of the overall risks at a pit lake must account for trophic status of the lake including food availability and the size of consumer populations that might be affected. Attention to the role of biological productivity in the risk assessment process is particularly important for pit lakes that exhibit oligotrophic characteristics where low biological productivity can result in very low numbers of individuals of a particular species that might be exposed to pit lake contaminants. Finally, reliance on only chemical data in assessing risks to biota in oligotrophic pit lakes can lead to excessive conservatism in risk estimates and remedial actions to mitigate calculated chemical risks.

This leads to the conclusion that the trophic status of a pit lake needs to be evaluated as a part of the risk assessment process. Risks in pit lakes with low biological productivity and less than acutely lethal contaminant concentrations may have insignificant impact on populations of resident and migrant species that are associated with the lake.

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CHAPTER 17

Flooding Pit Lakes with Surface Water

C.A. Pelletier, M.E. Wen, and G.W. Poling

INTRODUCTION

Flooding of pit lakes with surface water can sometimes produce relatively uncontaminated lake surface waters and discharge waters with qualities that exceed stringent discharge or effluent permits. Evaluation of this alternative for open pit mine closure requires detailed knowledge of water balances and all metal inputs to the water column plus metal removal processes that can be developed within the pit lake itself. In certain instances, both biological and geochemical mechanisms involving adsorption onto particulates and/or precipitation coupled with sedimentation of the metal-carrying particulates can provide effective metal removal mechanisms. An example of this type of pit lake treatment system, which is continuously evolving at the Island Copper Mine, is provided in this chapter. Many details of the Island Copper pit lake system in British Columbia, Canada, are provided in this chapter to illustrate the critical importance of monitoring physical, chemical, and biological parameters and being prepared to apply innovative technologies to cope with the near-continuous evolutions that can occur in a pit lake system (Wen and Pelletier 2006). Advantages and disadvantages of this type of system will be provided together with costs and cost comparisons to more traditional methods of coping with metal contaminants in mine waters.

INFORMATION NEEDED FOR DESIGN AND OPERATION

Pit lakes can be "permitted" as treatment ponds designed for the eventual discharge of excess surface waters to a receiving environment or they might also have to qualify as receiving environments themselves (Dagenais and Poling 1997; Poling et al. 2003). Owing to the geometries of these lakes, stratification is usually pronounced and many are meromictic or do not overturn on a seasonal basis. In these common circumstances, the surface layer might be habitable although lower layers are not. In some instances (such as at the Island Copper pit lake), the lower layers are or will become anoxic and therefore uninhabitable. Most often, pit lakes will be considered as "treatment ponds" and operators will be bound by discharge permits. Typically, the following data will be required for the most successful designs:

- Physical descriptions of pit sizes, shapes, elevations, climatic conditions, local meteorological conditions, rock types exposed within pit and geochemistry of each significant rock type exposed, groundwater conditions surrounding pit, upland watersheds and surrounding waste rock dumps, possibilities of gaseous eruptions within pit, geotechnical stabilities of pit walls, influence of ice or permafrost, and local wildlife concerns (Dunbar and Pieters 2004).
- Permit limits for critical water quality parameters including limits for both dissolved and total metal contaminants.

- Detailed water balance data including surface and groundwater inflows and possible eventual outflows, and accurate precipitation and evaporation records.
- Geochemical predictions of contaminant inflows from run-in waters as well as from mineral dissolution or desorption mechanisms (Morin and Hutt 2001).
- Contaminant removal mechanisms within the pit lake itself, usually by adsorption or precipitation mechanisms, but this might also involve knowledge of sedimentation and redox (oxidation–reduction) mechanisms.
- Predictions of changes in pit lake characteristics with time and seasonality, including changes in limnology, geochemistry, and biology are to be expected.

ILLUSTRATIVE EXAMPLE OF THE ISLAND COPPER PIT LAKE TREATMENT POND

Figure 17.1 shows the Island Copper Mine pit at closure in 1995 and again in 2002 after flooding of this pit in the summer of 1996 (Wen and Pelletier 2006). This pit was flooded mainly with seawater through a cut channel from adjoining Rupert Inlet and resulted in the waterfall as shown in the 1996 photo of Figure 17.1. Table 17.1 presents many of the physical properties and characteristics of this pit. Figure 17.2 shows an overview of the flooded pit, surrounding waste rock dumps, and ditch system for collecting surface waters and dump seeps to the pit lake. The water management system at Island Copper is designed to collect the acid rock drainage (ARD) in this network of ditches that ultimately delivers the water to the pit lake for biogeochemical treatment. The two primary ARD sources are generated within waste rock dumps contained in the North and Southeast catchments. The North catchment water has circumneutral pH and is now placed on the surface of the lake, and the Southeast catchment produces acidic water that is injected into the bottom of the lake's middle layer, at 220 m below the surface of the lake.

Table 17.2 shows typical characteristics of the two ARD sources and the effluent permit limits that must be met before discharge of pit lake surface waters to the receiving environment. Figure 17.3 shows the annual water balances for this pit lake in a schematic north-south section. Note that the more heavily contaminated ARD from the southeast catchment is injected through the south injector system (SIS) at a depth of 220 m while the less contaminated ARD from the north catchment (referred to as the NIS) flows onto the surface brackish layer on the north side of the pit. Also note on Figure 17.3 that the excess pit lake waters (approximately 6.5 million m^3/yr) overflow a submerged concrete weir wall along the southern side of the pit and then exfiltrates nearly 1 km to Rupert Inlet, which is the receiving marine water body.

Figure 17.4 shows a more realistic north-south sectional schematic of the pit lake with its three distinct layers and the exfiltration route through the permeable waste rock, placed mostly below low water, called the "marine beach dump."

Some of the alternatives considered for closure of this pit were

1. Cutting a channel to Rupert Inlet to make the pit lake into a continuously flushed artificial marine inlet. This idea was rejected because modeling predicted that the bottom of this pit lake would eventually become anoxic and that overturning could occur once every 50 years or so. This could result in a fish kill and was therefore unacceptable.
2. Making the pit into a freshwater-filled pit lake. Again, modeling predicted that the bottom would stratify and become anoxic and possibly overturn periodically, bringing anoxic waters to the surface.
3. Flooding the majority of this pit with seawater (screening out fish) and capping it off with a freshwater layer to produce a strong density gradient where no overturns could occur.

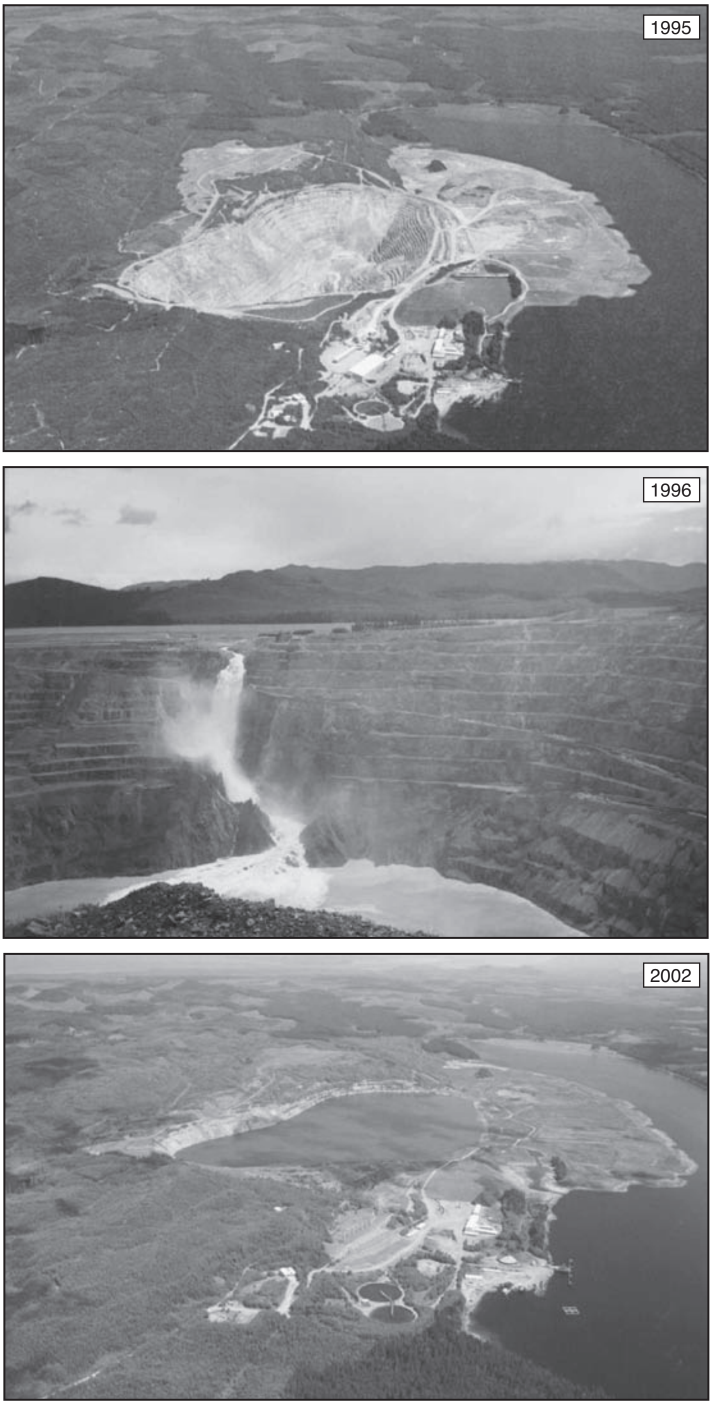

FIGURE 17.1 Island Copper pit prior to, during, and after flooding

TABLE 17.1 Island Copper pit lake physical characteristics

Basic Data	Parameter
Location	North shore of Rupert Inlet, Vancouver Island, BC, Canada
Size and shape of pit	Approximately 2 km long by 1 km wide, oval, maximum depth 353 m
Major rock types in walls	Andesitic pyroclastics, quartz feldspar porphyrys, Bonanza volcanics, minor sulfides
Total volume of pit lake	241,000,000 m^3
Surface area of pit lake	1,735,000 m^2
Annual direct rainfall to pit	1.88 m/yr, equivalent to 3,260,000 m^3
Pit lake watershed area	189 ha direct, 142 ha subterranean flows
ARD directed to pit lake	From 3.2 to 5.2 million m^3/yr
Run-in waters	1 million m^3/yr

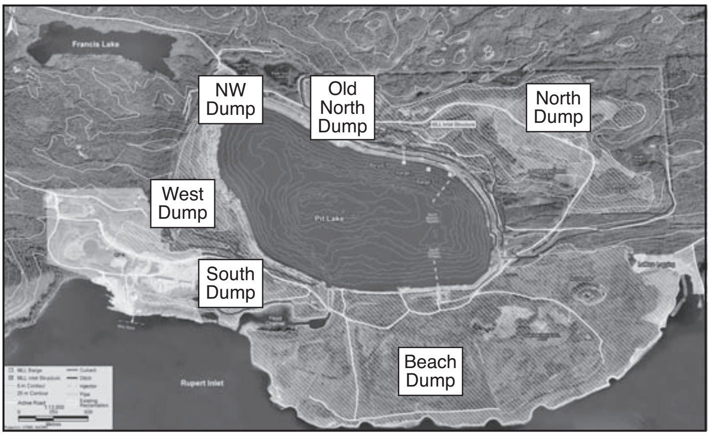

FIGURE 17.2 Flooded pit, uplands with waste rock dumps and ditch systems, and two ARD injection points

TABLE 17.2 ARD qualities and the effluent permit limits that apply to surface waters of the pit lake

Parameter	NIS	SIS	Effluent Permit 00379
pH	7.1–8.4	4.2–6.6	6.5–11.5
SO_4, ppm	400–780	1,300–2,000	
Ca, ppm	140–230	280–410	
Mg, ppm	20–35	60–100	
Cu, ppm	0.010–0.040	0.03–3.3	0.05
Zn, ppm	0.5–3.9	4.2–11.1	1.0
Cd, ppm	0.010–0.030	0.010–0.050	0.01
Mo, ppm	0.008–0.012	0.0005–0.001	0.5

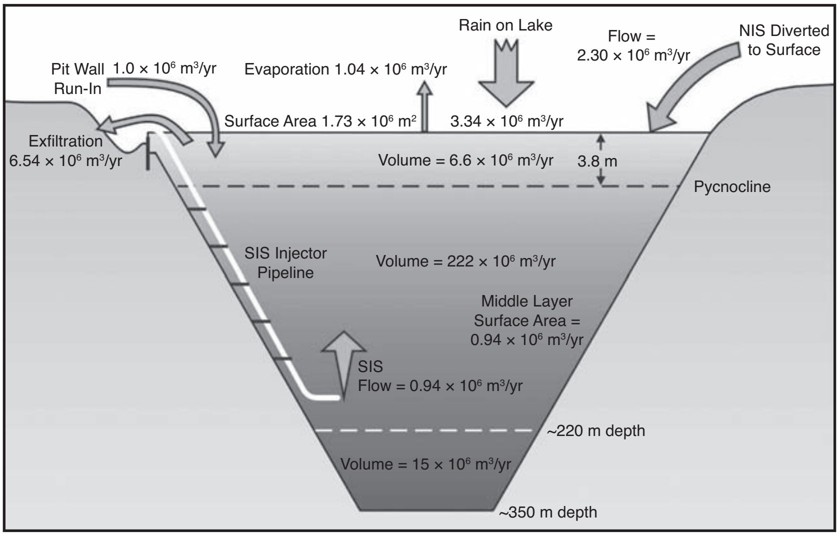

FIGURE 17.3 Island Copper pit lake water balances and volumes of each layer

The pit lake would become a treatment pond to ensure that ARD injected at depth would not overflow the south wall weir and enter the receiving water.

4. Utilizing the pit for a large-capacity garbage storage facility with water treatment, periodic soil coverage, and utilization of methane and other short-chain hydrocarbon off-gases created to produce electricity was examined in detail but eventually rejected because of political uncertainty surrounding sufficient garbage to justify the capital expenditure required.

The decision reached was to follow option 3 and create a meromictic pit lake with a seawater bottom and a freshwater top layer. Initially, the pit was flooded in approximately 30 days via a seawater channel, and the filling level was terminated more than 16 m below the overflow weir height. This left nearly 30 million m^3 to fill the pit lake to overflow and provided ample time for monitoring to gain an improved understanding of the geochemical reactions that would fix the levels of contaminants in the middle layer and also the surface layer water in this pit lake before discharge was necessary.

Figure 17.5 shows the evolution of some physical parameters in the Island Copper pit lake from 1995 to 2005. In order to form a freshwater top layer of significant thickness more quickly, the existing freshwater pipeline from the nearby Marble River was used to pump fresh water onto the top of the seawater. Using the same source of water that had been used for the process plant eliminated the need to reevaluate potential drawdown impacts in the Marble River. Figure 17.5 shows that by augmenting run-in and direct precipitation with Marble River water, the lake's free surface rose at a rate of approximately 9.3 m per year until overflow occurred on February 28, 1999.

The south wall concrete weir was not constructed to provide nonpoint source discharge of the top layer of pit lake water to the receiving environment. This wall was constructed several years earlier to keep Rupert Inlet seawater from entering the open pit. The 3-year delay to overflow was very useful to better understand what turned out to be rapid changes in water qualities within the three layers of this meromictic pit lake. The stable three-layer physical structure of

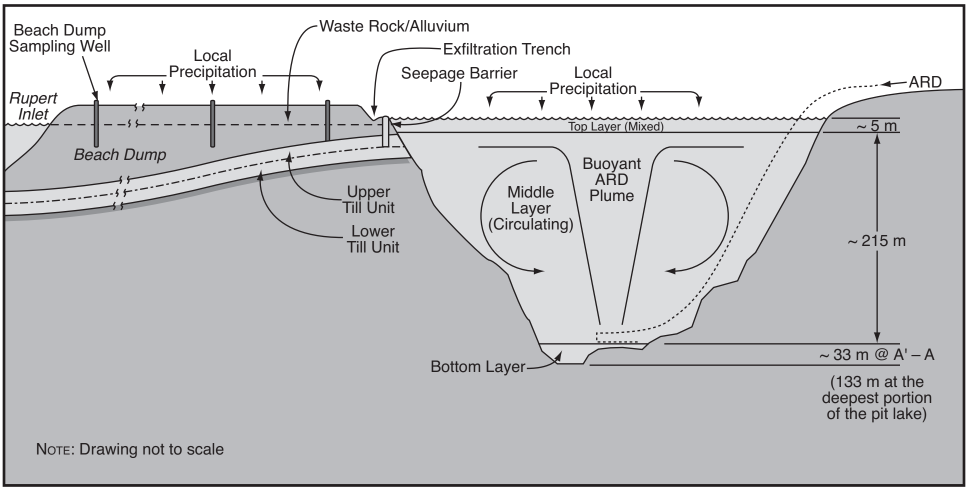

FIGURE 17.4 Schematic north-south cross section of pit lake, barrier wall (weir), and exfiltration route

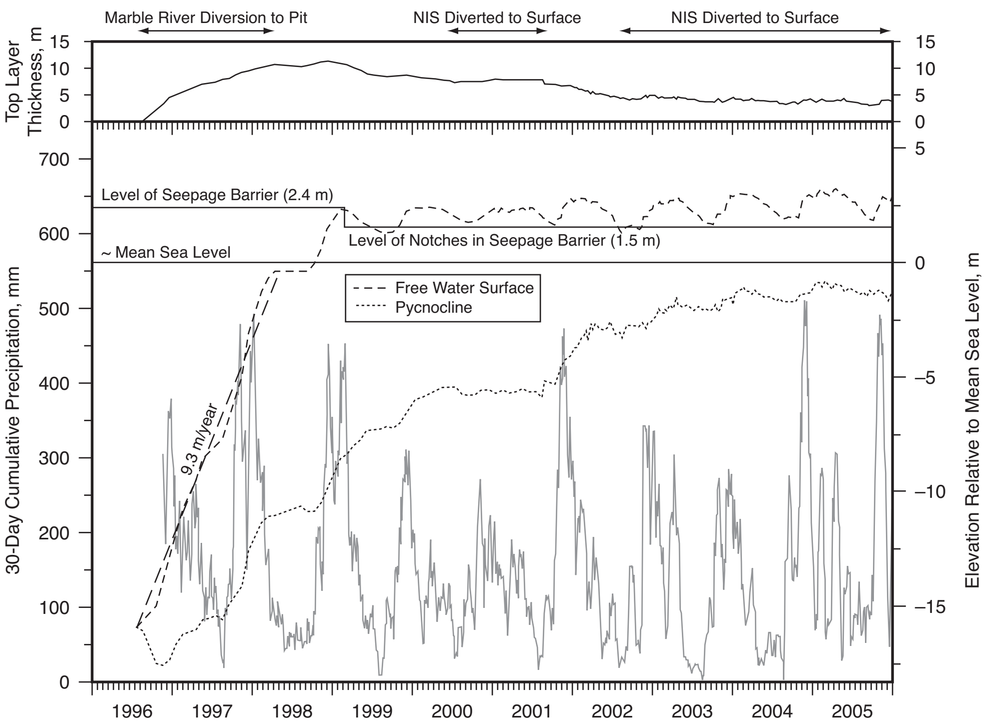

FIGURE 17.5 Evolution of pit lake surface and pycnocline elevations and rainfall record

the pit lake consists of a bottom saline layer (28 practical salinity units, or psu), a middle layer containing 90% of the lake's volume (25 psu), and a brackish top layer (6 psu).

As the pit lake level rose, so did the interface between the surface and middle layers, referred to as the pycnocline. This sharp-density gradient between the top layer at salinity 4–6 psu to initially nearly 35 psu was about 1 m thick and initially rose at a rate of approximately 3.4 m/yr because of the injection of ARD at a depth of 220 m (Muggli et al. 2000; Fisher 2002; Fisher and Lawrence 2000; Wilton and Lawrence 1998). The background of Figure 17.5 shows the record of 30-day cumulative rainfall, which adds approximately 3.3 million m^3 of fresh water to the top layer each year.

Figure 17.6 shows how dissolved oxygen levels in each of the three distinct layers have evolved over the past 10 years. The very bottom layer quite quickly (within 2 years) went anoxic. The middle layer, which was well mixed from the top (pycnocline) to its bottom by the density currents generated from admixing the "fresh oxygen rich" ARD at 220-m depth took nearly 9 years to go anoxic. The top layer is supersaturated with oxygen, mainly as a result of photosynthesis. Until only recently, anoxia and presence of hydrogen sulfide caused the conversion of soluble metals, such as zinc, copper, and cadmium, into insoluble sulfide precipitates primarily in the bottom layer (Kuyucak and St-Germain 1994). Now, sulfide precipitation mechanisms have been observed occurring rapidly within the middle layer as well after it became anoxic. In other words, even after intensive monitoring of this pit lake for more than 10 years, rapid changes are again taking place.

Island Copper provides a good example of the effectiveness of deep injection of ARD into a pit lake where chemically reducing conditions exist. Some of the considerations of a similar system for other pit lakes should include

- Physical structure and stability of the water column,
- Chemistry of ARD injection stream,
- Chemistry of pit lake receiving waters,
- Possibility of hydrogen sulfide generation as a treatment for dissolved metals,
- Ice cover conditions, and
- The pit lake outflow system (point source, diffuse discharge, evaporation, etc.).

Evolution of Biological and Geochemical Reactions Within the Island Copper Pit Lake

Figure 17.7 shows the levels of chlorophyll (indicative of the standing crop of phytoplankton) within the surface layer of the pit lake at Island Copper, together with the concentrations of dissolved zinc, copper, and cadmium. In the early years of this pit lake, rapid drops in the levels of dissolved metals coincided with each summertime rise in the population of phytoplankton. Research conducted using sediment traps within the pit lake water column indicated that the zinc, copper, and cadmium were being adsorbed onto the surfaces of the phytoplankton, which happen to live only 1 or 2 days and then sink to the bottom as organic detritus. After considerable research, a program of artificial fertilization was developed to enhance primary productivity in the surface layer. Plankton were caused to bloom on a continuous basis and perform as the primary metal removal mechanism for the top layer of pit lake water as well as to provide a source of energy for microbial decomposition and the production of sulfide in the bottom sediments. Today, approximately 1,700 L of liquid ammonium polyphosphate (10-34-0) and urea ammonium nitrate (28-0-0) are dispersed weekly into the pit lake surface behind the propeller of a motorboat. This amounts to 300 mg N/m^2 and 50 mg P/m^2 each week. The fertilization program is expected to continue until ARD quality improves to the point when the biological treatment

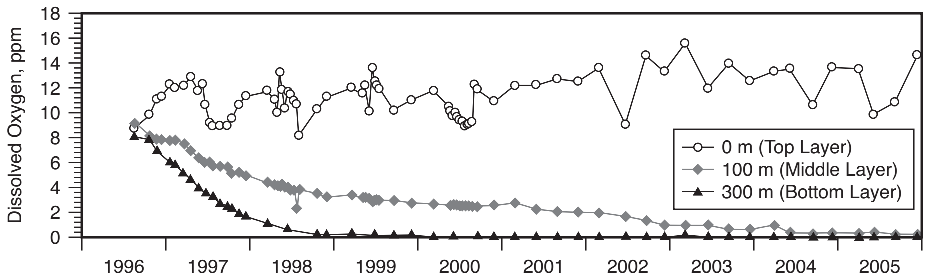

FIGURE 17.6 Evolution of dissolved oxygen concentrations in each layer of the pit lake

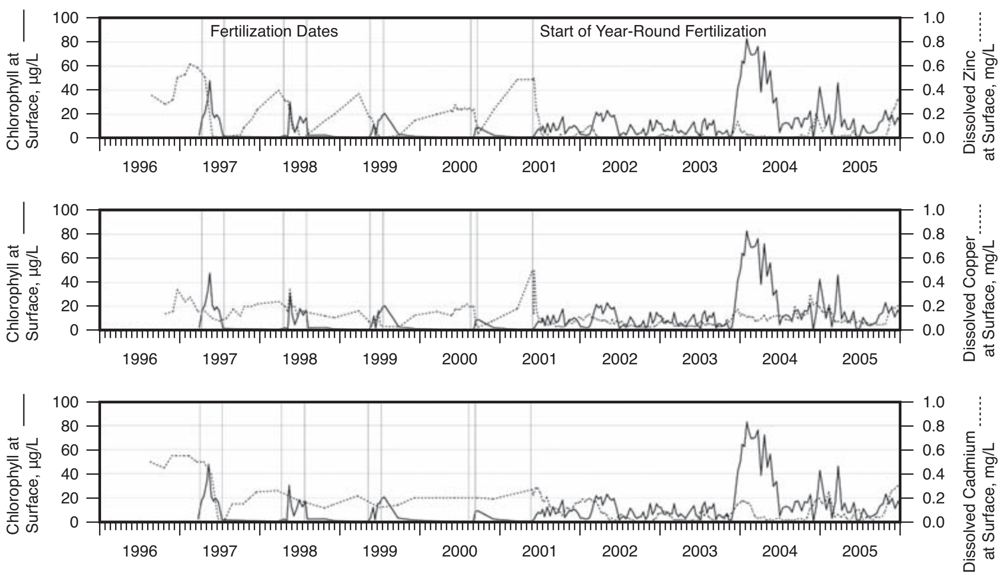

FIGURE 17.7 Evolution of chlorophyll (phytoplankton populations) and dissolved metal concentrations in the top layer (at a 1-m depth). Dissolved metal permit limits: 1 mg/L zinc, 0.05 mg/L copper, and 0.01 mg/L cadmium (see Table 17.2).

system is no longer needed. This is likely to take a very long time. The evaluation of mine drainage at the site has not been able to define the timeline for the exhaustion of ARD with any precision. The change in ARD rock is likely to occur slowly and will be monitored.

Figure 17.7 shows that since starting the artificial fertilization program in 2001, permit levels for dissolved zinc, copper, and cadmium have never been approached. Cadmium levels exceeded the 0.01-mg/L permit limit in the winter of 1996, but this was before discharge had begun. Utilizing the cautionary principle of not filling this open pit to overflow immediately upon flooding has turned out to be a wise choice. By far, the largest loading of dissolved metals occurred in this flooded pit almost immediately upon flooding and none of these waters came from either the seawater or the Marble River water. Almost all of this load of dissolved zinc, copper, and cadmium came from rapid dissolution of oxidation products contained within several million metric tons of waste rock that were stored at one end of the pit to minimize the mine footprint during the late stage operation of the open pit mine. Another lesson to be learned is that *if you don't want to start* *with a large load of dissolved metals in the pit lake water, then do not leave high-surface-area, highly oxidized, mineralized waste rock within the pit to be flooded.* One potential mitigation measure for this in-pit waste would be to control the pH of the water in the pit by the addition of lime (calcium oxide) to reduce the effect of dissolution of soluble acid salts.

Several significant changes had to be made in the Island Copper pit lake treatment system to maintain effective metal removal via the plankton adsorptive mechanism as time progressed. Alkalinity levels were falling too low to maintain healthy plankton blooms. Test work showed that the near-neutral NIS (mild-ARD) waters could be diverted to the surface layer and would provide the required alkalinity without adversely impacting the plankton blooms or their removal mechanisms. This addition began in June 2000 for just over a year and then again in July 2002 and continuing to the present. Eventually if the NIS surface flow becomes acidic or all alkalinity is depleted, some alkalinity may have to be added to the weekly fertilization cocktail added to promote phytoplankton blooms. Eventually, the NIS flow may be returned to the middle layer through the existing north diffuser for metal precipitation in the sulfide-enriched middle layer.

Solution to the Rising Pycnocline

One additional reason for diverting the NIS to the surface layer was to slow the rate of rise of the pycnocline, which might soon otherwise top the south wall weir and allow middle layer water with metal concentrations that might exceed permit limits to overflow into the receiving environment. Mathematical modeling had initially indicated that surface wind forces and thermal convection would cause sufficient currents within the surface brackish layer to erode waters from within the pycnocline into the surface layer and stabilize the top layer thickness to the order of 4–5 m. Such a limiting thickness would maintain the pycnocline to well below the level of the south wall weir and prevent middle-layer water from exiting the pit lake. Detailed monitoring and application of more sophisticated modeling predicted that even with the NIS directed to the surface, the pycnocline might still rise to overtop the south weir.

An idea emanated from within the Rescan engineering staff that if wind and convection were insufficient to erode middle-layer waters into the upper layer, then mechanical "middle-layer lifting" (MLL) could do the job. Major questions about potential detrimental impacts from mixing 1 to 2 million m^3 of middle-layer water (at a salinity of 25-30 psu) into the 6 million m^3 of surface water required answers. An adaptive management experiment was conducted within two 90-m-diameter limnological enclosures within the pit lake itself (Figure 17.8). One enclosure was used to test the concept of MLL, and the other tested the concept of diverting all SIS flows to the surface to eliminate the upward displacement of the pycnoline caused by the deep injection of SIS waters. Extensive test work showed that mechanical mixing and changes in salinity caused by MLL could be tolerated by the phytoplankton and achieves stability of the pycnocline elevation (Wen et al. 2004). The experiment also showed that placing the SIS on the surface at an accelerated rate could not be tolerated by the biological treatment system. On the strength of the experimental results, the plan for MLL was pre-approved by the British Columbia regulatory authorities. During 2005, a detailed design and construction of an innovative MLL system was implemented to take advantage of the 55-m elevation above the pit lake surface to energize the pumping required. Engineers designed a Venturi-eductor pump system on a floating barge tethered to the north slope of the pit walls, which was capable of pumping in excess of 2 million m^3 from a depth of 15 m and dispersing it along a manifold system along the north side of the pit lake. Figure 17.9 shows a schematic of the eductor system installed in 2005. By utilizing one to four intake pipes from the sump, flows can be varied from near zero to in excess of 250,000 m^3 per month. Monitoring shows the pycnocline is now stable. Mixing of this middle-layer water into the top layer has increased salinity

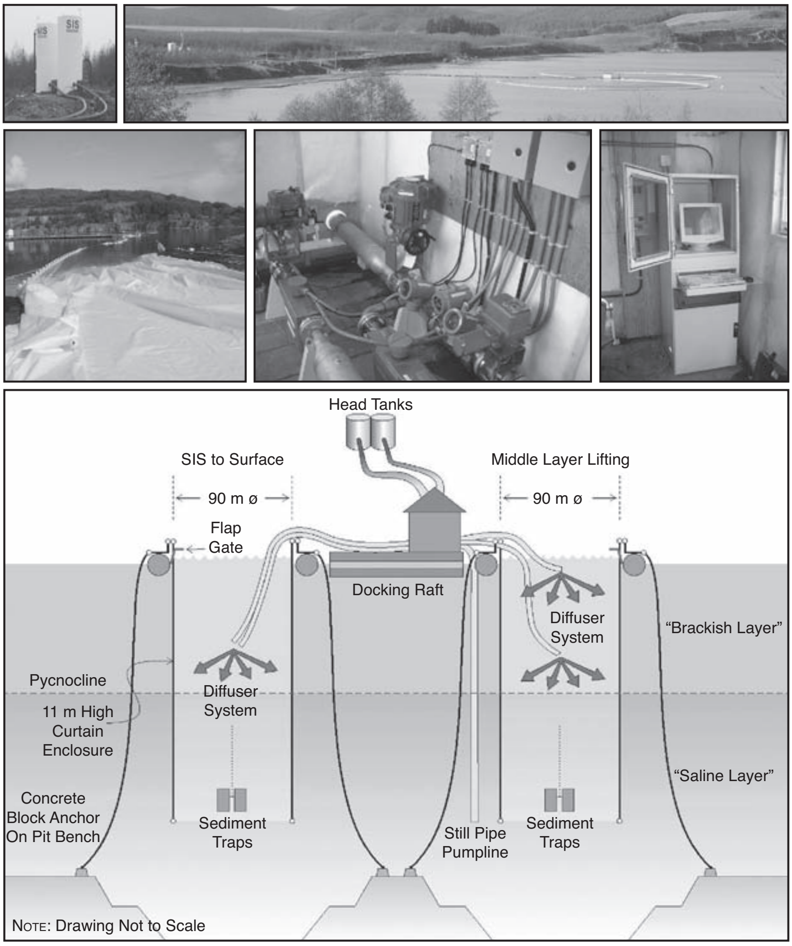

FIGURE 17.8 Experimental enclosures at Island Copper used to test management options for pit lake treatment

from approximately 3 to 6.5 psu and increased the levels of dissolved zinc, copper, and cadmium in the top layer. However, reference to Figure 17.7 shows that dissolved metals are still well within the permit limits. Figure 17.10 illustrates changes in phytoplankton populations from green algae to dominantly blue-green algae that occur every summer in the Island Copper pit lake surface.

One more evolving feature of the Island Copper pit lake treatment system is the dramatic removal of dissolved metals from within the middle layer. Figure 17.11 shows the recently observed rapid removal of zinc and cadmium, and less so for copper, from the middle layer. This is believed to be caused by developing anoxia within at least parts of the middle layer as a result of microbial respiration and diffusion from the pit bottom walls covered by organically rich reducing sediments producing sulfides. The onset of sulfide precipitation of these metals can explain the rapid reduction in concentrations and dissolved metal inventories. Precipitation as sulfides and subsequent sedimentation of these particulates onto the bottom lake bed provides a secure and permanent removal of the metal from the pit lake water column. The removal of metals in the water column as metal sulfides does not require permanent development of reducing conditions

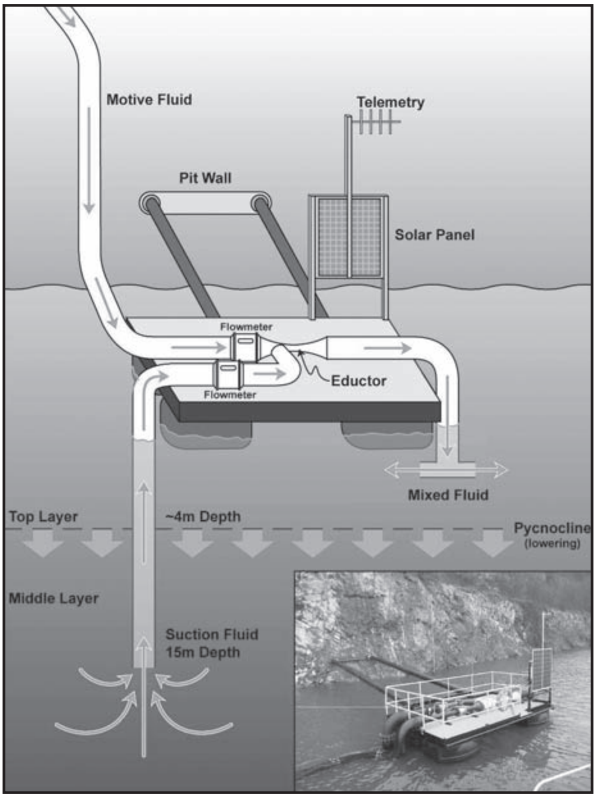

FIGURE 17.9 Venturi-eductor system installed for middle-layer lifting. The ARD flowing through ditches of the NIS is collected above the pit lake, where the head difference allows it to act as the motive flow for the MLL system.

in the water column. In fact, the dramatic removal of dissolved metals in the middle layer of the Island Copper pit lake occurred during a period when the presence of hydrogen sulfide was intermittent. This was sufficient to precipitate metals that sank to the organic-rich and anoxic lake bed.

CONCLUSIONS AND COMPARISONS

Application of surface flooding and biological metal removal from the top layer of the Island Copper pit lake has been continuously successful for the past 11 years. The alternative to this system would have been to install a conventional high-density sludge, lime neutralization plant and still employ the pit lake for deposition of the lime sludge product. With 1.8 m/yr of rainfall and run-in waters to eventually fill the Island Copper pit naturally, one would expect the pit lake to fill to overflowing within approximately 50 to 75 years. With the lime sludge in the bottom of the pit, a certain amount of preparation and monitoring for eventual overflow would be required in any event. The intensity of monitoring and research to ensure permit quality discharge waters would likely be less for installation of the lime neutralization plant.

When the Island Copper closure plan was submitted to the British Columbia government for approval, the estimated closure costs were Can$15 million (Pierce and Wen 2006). At least

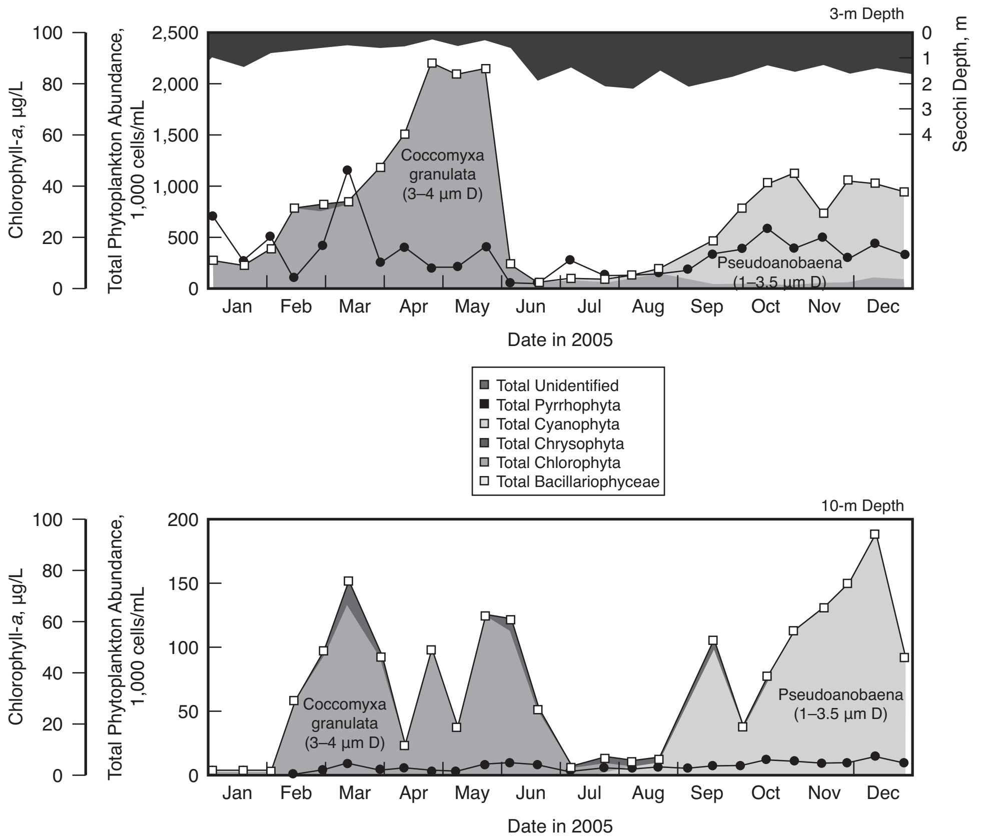

FIGURE 17.10 Changes to phytoplankton populations in the top layer from winter (green algae) to summer (blue-green algae)

two times this much has been spent on site reclamation to date, with most of the costs associated with re-sloping and top-covering waste rock dumps, improvement of collection ditch systems (more than Can$6 million spent), and removal of hydrocarbon-contaminated soils at the plant site. The 88,400 L of liquid fertilizer cost approximately Can$75,000 per year. Cost of fertilizer application, pit lake monitoring, and maintenance of the systems (e.g., boat, eductors, telemetry) in place costs approximately Can$200,000 per year. Including these and other minor costs, the total operating costs associated with the pit lake are approximately Can$300,000 per year. For comparison, a recently installed lime neutralization plant at Britannia, British Columbia, treating similar flow rates of ARD at similar acidities, cost more than Can$12 million in capital and more than Can$1 million per year to operate with no allowance for sludge disposal. Innovations during the evolution of the Island Copper pit lake have kept costs to a minimum by eliminating power for MLL, for example. The water in the middle layer has approximately 30,000 metric tons of residual alkalinity at present. Estimates are that this should be sufficient for the semi-passive treatment scheme to continue for the foreseeable future, after which time alkalinity might have to be supplied by chemical additions. The low cost of treatment used in the pit lake means that deferring active treatment into the future results in long-term savings for Island Copper closure (Pierce and Wen 2006).

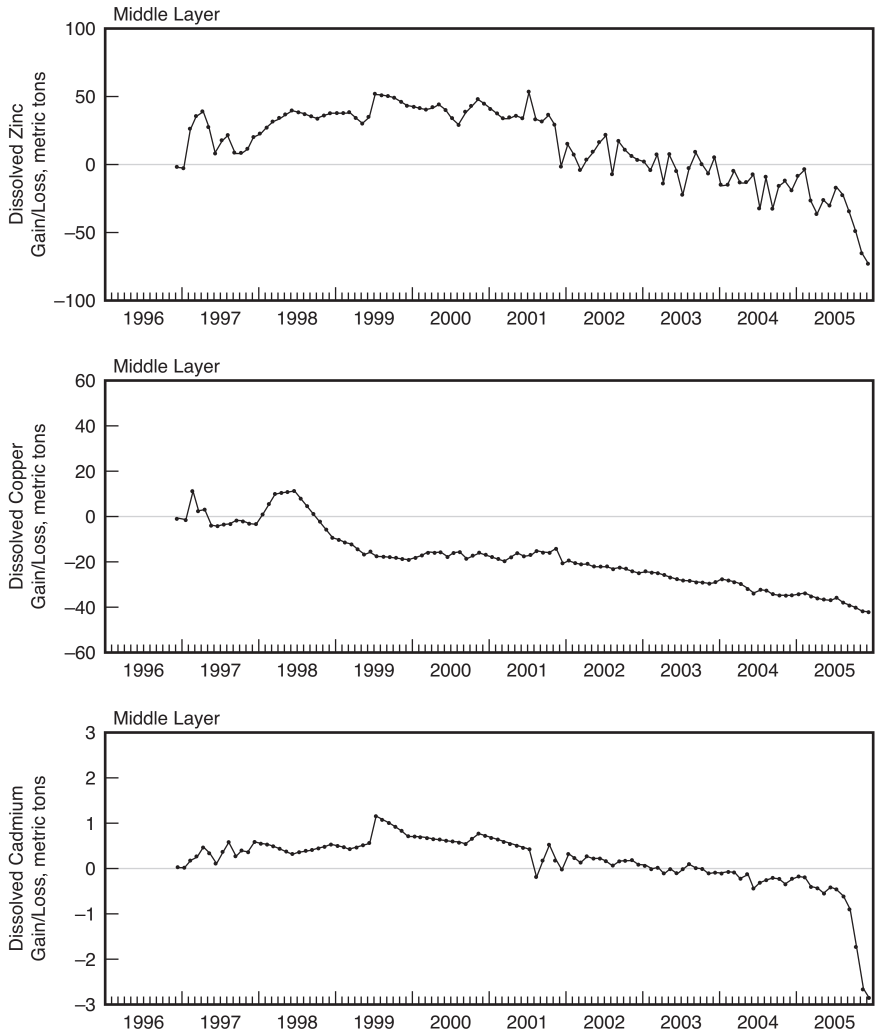

FIGURE 17.11 Middle-layer changes in dissolved metal concentrations. Primary metal loads occurred at the time of pit flooding in 1996. Metal removal in the middle layer is by adsorption to organic and mineral precipitates (hydrous ferric oxide, aluminum hydroxide). By the end of 2005, a total dissolved phase mass of approximately 75 metric tons Zn, 42 metric tons Cu, and 0.29 metric tons Cd had been removed. The bulk of the metals removed in the semi-passive treatment system at Island Copper have been in the middle layer.

The closure of Island Copper Mine has been successful in terms of minimizing long-term risks and costs despite the prospect of deteriorating ARD quality. Some unique aspects of the mine site were used to optimize closure:

- The proximity of the mine to a marine inlet, allowing the open pit to be flooded with seawater within a short period, minimizing long-term costs of pump flooding or natural flooding that could have taken 50 to 75 years.
- Engineering meromixis by capping the pit lake with readily available fresh water, ultimately creating a stable three-layer meromictic lake, as desired and predicted.

- The use of the well-buffered seawater as a neutralizing agent for acidic ARD injected deep into the lake.
- A temperate climate that allowed for successful year-round fertilization to stimulate bioremediation.
- The elevation differences between the pit lake's free water surface and a major ARD collection point, proving the opportunity to engineer a unique gravity-fed system to solve the issue of a rising pycnocline.

The detailed monitoring program at Island Copper has provided invaluable information on the performance of the pit lake and its likely evolution. Some useful lessons have been learned from the management of this pit lake. Rapid artificial flooding of the pit lake avoided long-term exposure of pit walls to oxidation in air, and the initial conditions of the fully flooded pit were known very quickly. Where possible, these authors believe that the benefits of artificial flooding generally outweigh the costs to do so. For some open pits, artificial flooding will require active pumping from a source lake at considerable costs; however, this must be weighed against the risks and financial liabilities of managing an open pit over the period of natural flooding. For some open pits, this could be as long as 1,000 years. Deep injection of ARD continues to be a successful practice at Island Copper and has benefited from the residual alkalinity of the seawater that makes up the lower layers of the pit lake. For other sites considering deep injection of ARD, an important aspect of the injection is the potential upward displacement of any permanent haloclines or chemoclines. Likely the most important lesson learned from the management of the Island Copper pit lake is that it is entirely possible, within limits, to engineer the physical structure of a pit lake and hence to artificially induce meromixis.

Some general recommendations to be drawn from the Island Copper case study regarding the prospects of flooding pit lakes and ending up with dischargeable waters are as follows:

- Prior to lake filling:
  - Perform sophisticated limnologic modeling to determine chemical and geochemical reactions or additions necessary to impart physical stability to the stratified structure that is almost sure to develop within the pit lake. Prevention of overturning should be a prime objective in order to secure dischargeable quality water in the upper layer.
  - Remove loose waste rock or possibly excavated low-grade-ore stockpiles that might be stored within the pit itself, particularly if this high surface area rock has undergone significant weathering-oxidation to generate rapidly soluble products. Alternatively, consider neutralizing the pit lake water with lime (calcium oxide) at a pH sufficient to precipitate metals as the pit is being flooded.
  - Determine potential effects from drawdown from source water bodies.
- During lake filling: Be conservative in initial filling of the pit lake. If possible, keep well below the level of discharge of surface layers to the receiving environment for a few years so that monitoring data and intimate knowledge of the physics and chemistry within the water column can be gained.
- After lake filling: Evaluate or confirm the prospects of biological metal removal mechanisms similar to that used successfully at the Island Copper pit lake. This can be tested in a partially filled pit lake.
- During and after lake filling: Collect detailed data on water quality and physical limnology using precision tools, such as conductivity-temperature-depth profiles.

- Be prepared to think outside of the box to seek innovative solutions to emerging problems that might develop as the pit lake system evolves with time. Place particular emphasis on collecting detailed monitoring data and on sophisticated modeling to attempt to predict future performance of the pit lake treatment system selected.

Keep regulators well informed of the state of evolution of the pit lake and the science to understand the processes involved. As conditions change, regulatory approval will usually be necessary before significant changes can be made to the operation of such a treatment system. Time delays for approval might cause problems in continuously meeting permit limits.

As these authors continue to apply the knowledge gained from Island Copper to other interesting pit lake problems, new challenges have been encountered that will require novel research, including

- The potential for pit lakes to act as nutrient sinks, affecting the productivity of water bodies within the watershed.
- For cold regions, ice formation and the role of salt exclusion is emerging as an important factor in determining the physical structure of the water column of pit lakes.
- For these cold regions, the ice cover for two-thirds of the year is important in the functionality of the pit lake system, as is the spring freshet.
- The effects of wind sheltering from high walls on pit lake mixing.
- The dynamics of multi-pit scenarios in series, and potentially with underground connections.
- In Arctic regions, the flooding of open pits may have an important effect on permafrost conditions surrounding the new lake.

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PART VI: REMEDIATION

CHAPTER 18

Biogeochemical Remediation of Pit Lakes

M. Kalin and W.N. Wheeler

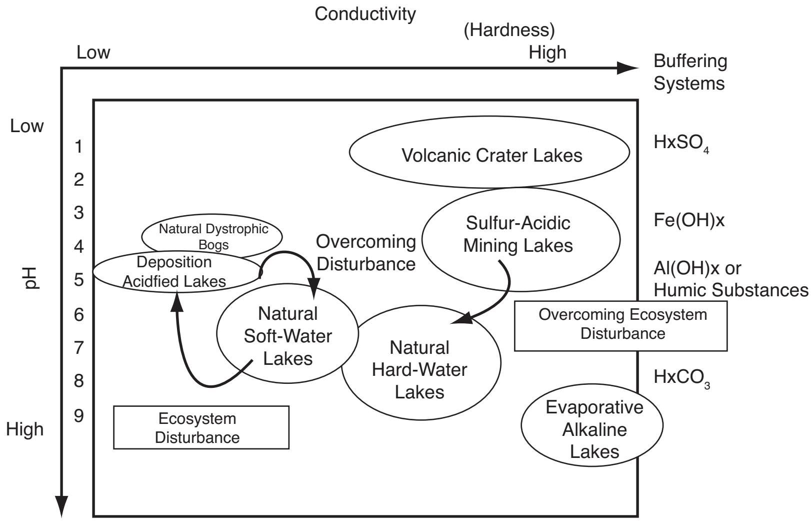
OVERALL APPROACH

Pit Lakes: The Knowledge to Assist Newly Forming Pit Lakes

A pit lake is simply a hole in the ground filled with water. As a surface water body, however, it lacks the limnological features that support diversity in a natural lake or pond (e.g., organic sediments, graduated shorelines, and a surrounding drainage basin). These features support diversity by providing particulates for water cleansing processes, (clay, alumina, silicates, etc.), and by providing organic carbon and nutrients, which in turn produce conditions for growth of adsorption surfaces. The primary focus of pit lake bioremediation is to compensate for these shortcomings. This requires a basic understanding of the hydrology, geochemistry, and limnology of these human-made ecosystems, which resemble natural lakes only to a degree.

Essential information leading to progress in predicting long-term geochemical conditions in pit lake chemistry may be found in the literature of extreme lake ecosystems, such as Mono Lake in California (United States). This extremely alkaline lake has a pH of 10, a salinity of 78 g/L (which is more than twice that of seawater), an alkalinity of 675 mEq/L, and an average sulfate concentration of 5.5 g/L (Domagalski et al. 1989). Abundant algae and cyanobacteria are reported from this lake. In contrast, the acidic crater lake, Ijen, in Indonesia has a pH of 0.02, total dissolved solids of 100 g/L, an aluminum concentration of 4.9 g/L, an iron concentration of 1.8 g/L, and a sulfate concentration of 60 g/L (Delmelle and Bernard 1994). Here, few records could be found of the life forms existing in the lake, but well-documented streamers (macroscopic biofilms) are reported in the acid mine drainage of Iron Mountain, California, with a similar low pH (Hallberg et al. 2006).

These extreme natural systems are well studied with respect to their geochemistry, although their ecology has received less attention. German pit lakes from open cast mining reflect chemistries that are remarkably close to such extreme natural systems. The ecology of these pit lakes, specifically their primary productivity, has been extensively studied (Fyson et al. 2006; and Nixdorf et al. 2001, 2003). Remediation efforts can therefore be viewed as assisting aquatic systems to make the shift from natural, extreme ecosystems to those which resemble oligotrophic lakes. Figure 18.1 describes the position of acid mining lakes in a pH/conductivity array. Changes in a number of limnological parameters, such as those depicted, are needed to move pit lake systems toward natural hard water lakes. Combining geochemical and ecological disciplines supports and expedites natural recovery processes. Rather than approaching pit lakes as being unique, with no parallels in nature, the objective of this contribution is to highlight the essential information needed to support the self-sustaining conversion of pit lakes to natural, hard water lake systems. This is achieved by broadly outlining the process by which remediation measures are selected. Three case studies are presented that illustrate these processes.



Source: Nixdorf et al. 2001.

FIGURE 18.1 Plot of electrical conductivity versus pH showing the position of pit lakes with respect to natural lake systems. The right axis indicates dissolved species that buffer pH within the specified pH range.

Hydrology

Sometimes the hydrology of pit lakes resembles those of natural kettle lakes, where input from springs is balanced by evaporation. During the mining phase, open pits are kept dry or nearly dry by pumping, and it is possible to determine recharge rates from the surrounding groundwater and atmospheric precipitation using the pumping records and the cone of depression, which was created in the groundwater table. The water quality collected from the pumping wells, which keep the pit dry during operation, can be used along with the hydrological information in hydrogeological models of pit development and dewatering. The output of these models is then used as input to predict the decommissioned pit lake hydrology.

When mining ends and pumping ceases, the original surrounding water table determines the final level of the pit lake. The combined clean and contaminated inflows and outflows determine the final water quality. Knowing the location of fault zones prior to flooding is essential for effective remediation measures. Older pit lakes may have origins from glory hole mining and hence display a complicated hydrology. Glory holes are pits resulting from a method of underground mining, where ore is mined from many tunnels with adits in the pit wall. The ore is collected at the bottom of the pit and transported through haulage tunnels to the mill. Water moves through the abandoned tunnels in the pit walls into the pit lake, providing upwelling and crosscurrent flows. It is important to know the location of incoming fresh and/or contaminated water plumes because those areas either support or hinder ecosystem functions. Thus, records of the mining methods and the physical development of the pit are essential. It may be necessary to address/alter the hydrological conditions through structural engineering measures, such as tunnel plugs or diversions of clean water through construction of a pervious surround (i.e., constructing an area with high hydraulic permeability).

A hydrological assessment is the key to any successful bioremediation scheme, as hydrology determines the theoretical residence time of the water in the pit lake. The residence time is simply the time available for in situ treatment. The annual volumes of incoming and departing water, evaporation/precipitation, and contaminant concentrations all provide the basis for calculating the contaminant loads. After loadings have been determined, an assessment can be made of the treatment options, either chemical or natural or both.

Geochemistry/Sediments

The geochemistry of the pit lake contaminants is controlled by physical factors such as oxidation state, temperature, metal concentration, pH, and the presence of buffers such as carbonates. The geochemical speciation of the contaminants, the surface charge of the suspended matter (e.g., positively or negatively charged particulates), and the concentrations of both particulate matter and the associated contaminant need to be determined using standard sedimentation traps. Force-flooding can provide an initial high load of inorganic suspended solids, serving to remove a large fraction of the metal loading released from the pit walls.

A water body without sediment cannot function as a natural lake ecosystem. The sediment provides a natural sink for water column contaminants (Walker et al. 1989; Mayers and Beveridge 1989). Organic matter needs to be introduced to construct sediments on the pit floor. Organic sediments initiate microbial activity that in turn generates reducing conditions which assist biomineralization. Construction of new sediments has been tested under the Canadian Mine Environment Neutral Drainage Program (MEND) and these authors' reports summarize the alkalinity-generating process (see Kalin 1993 and Kalin et al. 2006a). This alkalinity-generating process is referred to as acid reduction using microbiology. Studies of pit lakes from the coal mining region of Germany have shown that the reducing capacity of the sediments can be extended into the water column, leading to metal precipitation and meromictic stratification (Fyson et al. 2006).

Limnology/Biology

Vertical (depth) profiles of physical and chemical parameters (i.e., oxygen, temperature, conductivity, pH, light penetration) need to be quantified over an entire year. Picoplankton, phytoplankton, periphyton, and other aquatic biota such as zooplankton need to be identified and quantified. Early pit lake ecology is often dominated by single species with extreme densities that produce extracellular polysaccharides serving as flocculants for particulates with contaminants. The nutrient status of the lake must be determined. Growth rates of potential bio-polishers need to be compared to optimal growth rates in the literature. Climatic conditions determine the physical dynamics of the water body and are reflected in the temperature regime, light availability, and nutrient status of the water. These factors are the most relevant drivers of the sustainability of natural cleansing processes, which include phytoplankton, periphyton, and aquatic plants, the primary production of which provides new adsorption sites for contaminants and flocculants.

When the natural cleansing process drivers are estimated for a specific pit lake, a grounded sense of the contaminant removal potential is obtained. The hydrological parameters, along with the limnological status of the lake, provide the basis for estimating the biological and geochemical polishing capacities, which in turn determine the cleansing potential. With this information, the processes can be optimized by providing nutrient inputs that might be limiting the growth of the bio-polishers.

Scale-up

Pilot-scale testing is best carried out in the pit lake through gradual additions of the limiting nutrients, followed by monitoring the changes in lake chemistry brought about by the metabolic activity of the bio-polishers. These changes occur gradually, reflecting seasonal growth cycles over several years. In spite of all efforts to understand the dynamics of the new ecosystem, it is very likely that scale-up will bring surprises as elements of geochemistry, hydrology, and biology interact. It is difficult, if not futile, to predict the long-term cycling of elements without collecting data for at least the first 5 years after the pit lake has been established. The development of chemoclines, thermoclines, and/or meromictic conditions interact in unpredictable ways. Any one of these conditions may require further changes in the remediation strategy. For example, if a thermocline or chemocline develops, particulate matter (total suspended solids [TSS] with associated contaminants) in the epilimnion will never pass through the established "cline" to the sediments below. They will be cycled seasonally or, under meromictic conditions, may degrade before they reach the sediment, re-releasing contaminants into the water column. Whether or not stratification should be encouraged or prevented in the lake will depend on the geochemistry of the specific contaminant to be removed.

PIT LAKE BIOREMEDIATION PROCESSES

Generally, treatment consists of three steps: (1) Contaminants are sequestered with biomass and suspended solids, (2) then transported with the biomass and inorganic, agglomerated particulates (large enough to settle to the sediments), (3) where through microbial activity, the contaminants are biomineralized.

The way in which particulates are formed is specific to the contaminant. The cell walls of most algae or microbes are composed primarily of polysaccharides. These molecules contain negatively charged ligand groups that adsorb dissolved metal contaminants, which often have positively charged surfaces. Polysaccharides can also be excreted by the cells as a by-product of growth. For example, Cameron et al. (2006) found that iron is adsorbed to dissolved organic carbon in the Berkeley pit lake, Montana, United States. In the B-Zone pit lake, described in text that follows, the alga *Dictyosphaerium pulchellu* agglomerated with its polysaccharide sheath to form particles, as schematically depicted in Figure 18.2. Without a flocculating agent, the particulates remain suspended and do not become large enough (>1 μm) to settle out of the water column. Much work remains to be done to facilitate this process effectively.

To date, attempts have been made to describe some relevant aspects of the process for uranium (Kalin et al. 2004) and for arsenic and nickel in highly dilute pit lake water (Kalin et al. 2001). The chemical, physical, and biological characteristics of the particulates collected in sedimentation traps from all three of the case studies indicated that microbial cells were precipitating iron and zinc on their cell walls and promoting the nucleation or aggregation of particulates (Lowson 1998). It is well documented that surface currents and Eh/pH conditions on or near cell surfaces encourage precipitation on the outside or inside of cells. Kalin and Wheeler (1992) described algal mats that promoted zinc carbonate precipitation. In the Berkeley pit lake in Montana, Mitman and associates have been documenting the effects of algal bio-adsorption of metallic contaminants (Mitman 2000, 2001; Bartkowiak and Mitman 2002). The addition of nutrients to acidic pit lakes enhances algal and microbial production, which in turn increases adsorption and sequestration of contaminants (Crusius et al. 2003).

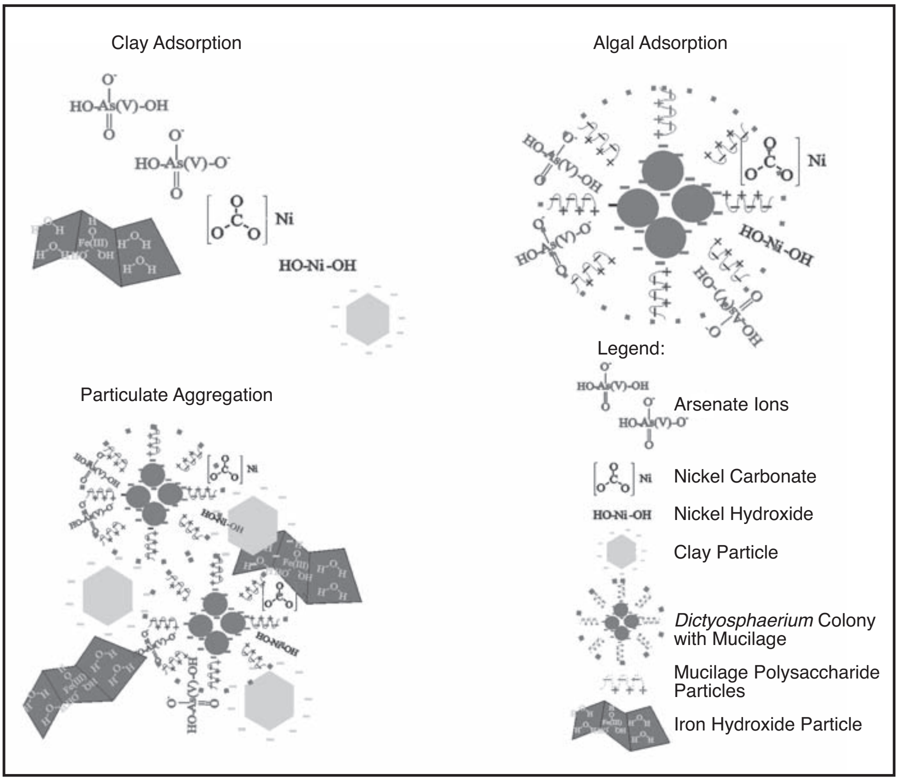


FIGURE 18.2 Schematic associations of arsenic and nickel particulates

PIT LAKE BIOREMEDIATION EXPERIENCE

Biogeochemical polishing processes were encouraged in the three case studies presented in the following sections. These Canadian lakes all represent differing hydrological and chemical challenges. Two of the projects were initiated as alternatives to perpetual treatment, and one averted pumping and treatment costs. The first case is an open pit in northern Saskatchewan where the major input is atmospheric precipitation. The second system is an acid lake in northern Ontario, and the third, in central Newfoundland, treats the combined effluents from two glory holes or pits and a drainage tunnel. Both of these latter water bodies have circumneutral pH values.

The contaminants to be removed are zinc, copper, arsenic, and nickel. The projects were initially intended to test ecological engineering concepts. Encouraging results from the field projects led to full-scale implementation in the acid lake and to the effluents from the glory holes. The pit lake in Saskatchewan reverted to a natural lake and provided an excellent data set on contaminant removal processes. For details, see work by Kelly et al. (2007).

Uranium Mine and Pit Lake, Saskatchewan, Canada

In northern Saskatchewan (58°11'N, 103°41'W), a mined-out pit was force-flooded with water from an adjacent lake, forming a 5×10^3 m^3 pit lake with a maximum depth of 45 m. The pit was essentially dry with minor seeps from fault zones. Arsenic and nickel dissolved from the pit walls after force flooding and, as later determined, from the run-off of evaporates formed on a road on the pit lake rim that had been constructed with waste rock. One year after flooding, the concentrations of the contaminants were low (<1 mg/L) and TSS were still high. The organic matter needed for the bottom sediments originated from eroding peat and muskeg at the rim of the pit. Water samples were collected from the lake two to three times per year, starting in the winter of 1991 and continuing until 1998. The contaminant removal process was quantified by following the fate of the particulates from the water column to the bottom sediments and by analyzing their elemental composition. After three summer seasons, it was evident that arsenic and nickel concentrations in the pit had decreased by 22% to 35%, and that iron had decreased by 39%. The transport from the surface water to the thermocline was facilitated by a dense population of the polysaccharide-forming alga, *Dictyosphaerium pulchellum*. Based on the quantities of algae collected in sedimentation traps and the growth rates of these algae, it was estimated that 31 metric tons of dry mass were generated within one season. The nickel increase in the algal biomass and in the inorganic component of TSS (collected in sedimentation traps placed at 2-, 12-, and 30-m depths) accounted reasonably well with the summer decline in nickel concentrations in the water above the thermocline. However, each spring after the snow and ice melted, the nickel concentration rebounded to a level just slightly below that of the previous year. In contrast, the arsenic concentrations declined gradually year by year.

The contaminant concentrations and their ratios changed in the material collected from the sedimentation traps at different depths and over the years. The composition of the particulates and the location of the associated contaminants were investigated with secondary ion mass spectroscopy. Geochemical modeling identified the probable chemical forms of the contaminants in the water. The arsenate ion was associated with the iron hydroxide particles, whereas the two nickel species, nickel-carbonate and nickel-hydroxide, were preferentially associated with the algal biomass. This was initially interpreted as the cause of the seasonal nickel concentration rebound. The biomass was probably decaying at the thermocline, which broke down under the ice, releasing the nickel. Only the arsenate that was associated with the iron-hydroxide (which in turn agglomerated to clay particles) formed large enough particles to settle to the sediment. This also explained the changing ratios of the arsenic and nickel in the particulate matter. Over time, however, the particle composition changed as algal species diversity increased, leading to different adsorption patterns. These changes were documented by adding bentonite (with differing surface charges) to buckets of pit lake water.

If the biomass decay at the thermocline contributed to the annual rebound of the nickel concentrations, differences should have been seen in the elemental composition of the bottom sediments. But through a mass balance of arsenic and nickel, their concentrations in the sediment traps (water column) and the top 5 cm of the sediment were compared and it was found that, rather than a recycling, an external source of nickel had to exist. An external source of nickel was found in the gravel-covered portion of the perimeter road. The road material was removed and the nickel concentrations in the pit lake continued to decline.

Boomerang Lake, Northern Ontario, Canada

In northern Ontario (longitude 92°40', latitude 51°08'), a 3×10^5 m^3 contaminated, acidified natural lake was utilized as a biological polishing lake. The mine is located on a peninsula of a large trophy sports fishery lake. Although not a pit lake, Boomerang Lake had many shared characteristics with other, more classical pit lakes. During operations, copper, zinc, and iron were discharged from the mill and mine site, and groundwater started to seep from the adjacent tailings pond directly into the lake. On shutdown of the operation, the underground workings started to overflow, and this, too, was diverted into the lake. In addition to the contaminated surface runoff, the 5-m-deep lake receives clean surface and groundwater from opposite shores of the mine mill site. The water column turns over completely in spring and fall and has a residence time of 3 years.

After 10 years of mining, the sediments contained up to 23% iron along with zinc and copper. Seasonal iron cycling resulting from spring turnover of the lake gradually decreased the pH from 4.5 in 1986 to 3.2 by 1999. The project started in 1982 with an inventory of indigenous flora and water quality analyses. Periphytic algae found on submerged vegetation contained 0.6% copper, 6.7% iron, and 0.5% zinc on a dry weight basis. Based on the high concentrations found in the local periphyton, a decision was made to increase the surface area in the lake on which these algae would grow. After considerable testing of attachment surfaces, it was found that local fir and alder brush cuttings provided the most suitable surfaces along with introduction of an acid-tolerant moss. This moss was found in acidic, abandoned tailings ponds in Elliot Lake, Ontario. Radioactive carbon studies documented the oxygen consumption of these moss carpets in tailings ponds in northern Saskatchewan (Kalin 1982). The moss carpet provided reducing conditions at the sediment surface, which slowed further acid generation in the lake. The reducing zone also provided surfaces for iron hydroxide precipitation and sequestration.

In 1987 and 1988, 600 brush cuttings were deposited in the shallow areas around the lake perimeter. A further 3,400 brush cuttings were added in 1990 and, in 1995, 40 truck loads of brush (~14,000 cuttings) were added. Sediment traps were installed to quantify the organic and inorganic suspended solids, and algal traps were installed to document the growth and sloughing of the algae (Kalin 2000). Extensive documentation of this project can be found on the Internet (Kelly et al. 2007).

The year after the first brush addition, it was estimated that each brush cutting supported 8 kg of algal growth. This increased to 20 kg per cutting in the second year and remained at this level thereafter. The attached algae contained 4.2% iron, 0.02% copper, and 0.05% zinc by dry weight, which, except for the iron, was an order of magnitude lower than earlier measurements. Based on the measured growth rates, each brush cutting removed between 4 to 10 g of zinc within the first 2 years of growth, clearly insufficient for the contaminant load the lake received annually. These authors neglected to factor in that after a critical biomass is reached, the algae and accompanying precipitates slough off, and growth starts anew. If two or more growth cycles per annum (fall and spring) could be attained, biological polishing would be a viable technology. Ways were sought to enhance the growth of the bio-polishers and simultaneously reduce the contaminant loading, focusing on the mill and mine site.

The addition of phosphate rock was considered as a means to solve both problems. Earlier studies had shown that algal growth increased significantly if phosphate was added. Phosphate is also a buffer and reacts with iron and other contaminants to form insoluble precipitates. After extensive laboratory and field testing, 162 metric tons of finely ground phosphate rock were added to the lake over a 3-year period. The distribution of the finely ground rock over the sediment was expected to counteract the annual decrease in pH caused by the reoxidation of iron hydroxide on the sediment, provide phosphate nutrients to the bio-polishing flora, and cause oxidized iron to precipitate as relatively stable iron-phosphate.

One metric ton of calcium nitrate was also added to the lake with the last phosphate additions to correct a shortage of nitrogen compounds in the water column. The effects of the sediment fertilization were dramatic. Acid-tolerant moss produced a dense carpet over the sediment and lowered the redox (oxidation–reduction) above the sediment, thereby slowing the annual reoxidation of iron-hydroxide. This stabilized the water column pH. The periphyton flora grew faster and removed more contaminants.

An elemental mass balance for the system was needed to quantify contaminant input to the lake, the retention capacity of the lake, biological polishing measures, and ultimately to assess the effectiveness of all the measures taken. Monitoring data (water and flow) were used to estimate the contaminant loads and sinks. Between 1987 and l994, no remedial actions were taken; between 1994 and 1999, brush was added and the sediment fertilized; between 1999 and 2003, metallic magnesium scrap was added to the lake, the effects of which were monitored until 2003, the end of the project.

The contaminant load to the lake from the mine/mill site and the groundwater seeps was relatively well documented, with flows and contaminant concentrations measured several times per year. Water quality was also measured at several locations in the lake, and the lake load was calculated by multiplying the volume of the lake by the contaminant concentrations. From the lake load, the elemental mass in 100,000 m^3 was subtracted as it was leaving the lake. The difference between input and output was considered the load retained in the lake.

To verify whether this approach to a mass balance was realistic, sediment cores were obtained and the elemental concentrations assessed to a depth of 20 cm, which, based on the sedimentation rate, should include sedimentation during the previous 3 or 4 years. Because the sediments were collected in 1998, the cores sampled to a depth of 20 cm should have included sediment deposited during the period from 1994–1998 (Table 18.1). The lake sediments, when extrapolated from the cores, included 2 metric tons of copper, 468 metric tons of iron, and 51 metric tons of zinc. These amounts of copper, iron, and zinc are roughly the same as the total contaminant load estimated for each of the metals, with the possible exception of copper. Copper loading to the lake was significantly reduced after milling ceased, as the copper originated from run-off of the concentrate storage pad and was carried via rainwater and the wind into the lake. Because the mass balances "balanced" to a reasonable degree, there was confidence that this approach could be used to evaluate the ecological approach to contaminant removal.

During the second period from 1995 to 1999, the moss carpet developed and the copper input was lower. With no remedial measures in the first period, 73% of the copper was retained, whereas during the period with the moss, phosphate, brush, and algae, only 54% was retained. Even less of the copper was retained in the last period (25%). However, copper and zinc concentrations in the lake increased from 0.05 mg/L in 1986 to 0.7 mg/L in 1999, and 2 mg/L to 35 mg/L, respectively (Kalin et al. 2006b). Iron retention remained between 97% to 96% through all three periods.

TABLE 18.1 Boomerang Lake total contaminant loads and sediment sinks

	Cu			Fe			S			Zn		
	Boomerang Lake Load (total metric tons)											
	In	Out	Retain	In	Out	Retain	In	Out	Retain	In	Out	Retain
No treatment (1987–1994)	2.6	0.7	1.9	355	9	345	461	239	221	101	22	79
Phosphate and brush (1995–1999)	1.1	0.5	0.6	416	9	407	466	228	238	98	41	57
Magnesium (2000–2003)	0.8	0.6	0.2	314	11	303	339	244	95	88	47	41
	Sediment Sink (total metric tons)											
Sediment (1998)			2			468			Not Appli-cable			51

Sulfur, zinc, and iron inputs to the lake only dropped during the third period. These reductions were due to covering the mill site with clay, and enlarging the ditch directing underground effluent to the lake, thus increasing the iron hydroxide retention. The mass balance suggested that the sediments were releasing the metals on account of the slow but steady decline in water pH, and the moss cover had likely not reached its full effectiveness. A gradual but steady increase to a pH of 4.0 would be needed to provide carbon for the bio-polishers to grow fast enough to sequester most of the remaining contaminants.

In 2000, 16 metric tons of metallic magnesium were suspended from barges in the lake. The magnesium metal corroded and consumed hydrogen ions, which counteracted the pH decline. This measure along with the ever-increasing moss cover led to a 10-mg/L reduction in zinc and a 0.3-mg/L reduction in copper. By 2003, the magnesium corrosion had stabilized the pH at or above 3.0 (Kalin et al. 2006b). Finally, in 2006, a site visit revealed that the moss had continued its prolific growth, and the lake water pH had remained at 3.0. Copper and zinc concentrations had also remained at 2003 levels. This case study of an acid lake highlights the need to produce elemental mass balances and the importance of pH as a controlling factor in iron-rich systems.

Buchans Pit Lakes and Polishing Ponds, Newfoundland, Canada

Two flooded pits, both former glory holes, are located in Central Newfoundland (48°N, 57°E). In these types of pit lakes, upwardly welling groundwater from the bottom of the lake along with strong currents along the pit walls (from the former mine workings) do not allow the deposition of sediment. Without sediment, the glory holes cannot be nudged on the road to natural recovery. The lake water had to be treated after it left the pit lake. This was done by developing a biological polishing-pond system in a meadow just below the pit lake where muskeg vegetation and peat provided a great sediment. This project highlights the importance of understanding pit lake hydrology.

One of the pit lakes was used to oxidize iron as it upwelled in a reduced state from the groundwater. A living-particulate capture system was installed on the lake surface in the form of floating cattail rafts. These rafts used the cattail's extensive root system to sequester the iron hydroxide. The polishing system of the ponds was designed to treat a flow of 20 L/s. However, the site operators added an additional flow of 20 L/s from a drainage tunnel, doubling the flow and zinc load through the pond system. This reduced the retention time in the ponds, producing less iron oxidation and more erratic zinc removal in the summer.

Work began by collecting water samples throughout the year. These samples were analyzed for metals, nutrients, pH, Eh, and temperature. Local algae were identified and their growth rates quantified. Sediment traps were installed to collect particulate matter in the pit lakes and to determine the composition of precipitates.

A field pilot-scale system was constructed in 1989 to develop design criteria for the full-scale system. Six serial ponds were constructed, each with a surface area of 70 m^2 and containing a volume of 40 m^3. Alder branches were placed in each pond to increase the surface area (~3.8 m^2 branch surface area per cubic meter of pond volume) for algal growth. Flow through the pond was controlled to provide residence times between 16 and 79 days.

The zinc concentration in the pit lake effluent declined as the effluent passed through the pond system from an initial level of nearly 16 mg/L to 1.2 mg/L (88% removal). A linear regression analysis of the reduction in zinc concentration versus residence time was performed using data collected between 1989 and 1993. A zinc concentration decrease of 2.2% per day (whole year) or 5.2% (summer only) for the system treating the 20 L/sec leaving the larger pit was projected (Kalin and Wheeler 1992; Geller et al. 1998). The addition of fertilizer (Plant Products

10-52-10) was tested in the pilot system and found to initially double and, in the second year, triple the growth rates of the algae, leading to a substantial increase in the removal of zinc.

Two full-scale systems operating in parallel were constructed in 1997. Each system consisted of four ponds in series, each with a water surface area of 13,100 m^2 and a volume of 6,000 m^3. The flow rate through the ponds was 4.7 L/s, resulting in a residence time of 15 days. Using the full-year projection of 2.2% reduction per day of residence, a 32% reduction in zinc was expected. Using the summer reduction rate of 5.2% per residence day, it was estimated that 78% of the zinc should be removed during the summer. In the first 2 years of full-scale operation, the annual average zinc removal was 31% and the summer average removal was 66%. This suggested that residence time was not the only factor that controlled zinc removal. In later years when the second half of the parallel pond system was installed, summer removal was 80% or higher, but nothing was removed during the winter. Iron oxidation cannot take place under the ice, so there were no particulates on which zinc was co-precipitated. The algal growth was very low in the ponds during the winter, so surface areas remained saturated and likely contributed to the good summer performance. The combination of air, rising water temperatures, and light enhanced the precipitation of iron hydroxide and zinc carbonate, and the co-precipitation of metals. These factors represent the geochemical conditions needed for the natural cleansing processes. Light, with nutrients, provided the conditions for the growth of algal surfaces onto which the metals and precipitates could be sequestered. The absence of these conditions during the winter slowed the removal process to near zero. Although the effluent was not treated effectively year-round, the processes by which the zinc and iron removal took place were proven.

DISCUSSION

The projects described demonstrate that natural cleansing processes, once set in motion, will become self-sustaining. The natural recovery process will proceed once assisted through ecological engineering measures, but much work remains to define measures more broadly than outlined in the three case studies presented. From the management of reservoirs and human-made lakes (Cooke et al. 2005), much can be learned for the management of pit lakes. From the work of Buffle and Vanleeuen (1992), an excellent understanding of environmental particles can be derived. The case studies discussed in this chapter and the associated research have shown that the initiation of biologically mediated removal requires extensive monitoring and understanding of the ecological control factors. This was evident in the acid lake restoration and the work in the glory holes. As there was no control over the field site, not all the measures these authors would have liked to implement were possible. However, the studies document that natural water cleansing processes can be supported and/or facilitated.

This chapter has summarized the salient components needed to define pit lake ecosystems and the relevant processes that take place in pit lakes. Reductions in metal concentrations take place slowly, and yet, there is a tendency among the sponsors of remediation work to conclude that a project is complete when improvements have just started to be observed and documented. Because such decisions curtail data collection, more long-term data observations are needed to confirm long-term predictions. All three of the described sites would provide a wealth of information if follow-up work could be carried out. Pit lakes have the potential to gradually clean themselves up, but, using ecological engineering measures tailored to pit lake limnology, these natural cleansing processes can be assisted greatly.

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PART VI: REMEDIATION

CHAPTER 19

In Situ Bioremediation of Pit Lakes

B. Wielinga

INTRODUCTION

Pit lakes that form at the suspension of open pit mining often contain elevated concentrations of heavy metals, metalloids, and/or radionuclides that may be above regional background levels and/or regulatory standards. Traditional water treatment processes for removing these constituents are expensive, may increase the salinity of receiving water as a result of chemicals used in the treatment process, and typically create process solids that require on-site or off-site disposal.

Bioremediation is a technology that relies on microorganisms to reduce, eliminate, or contain contaminants. Approaches to the bioremediation of pit lakes containing elevated concentrations of trace metals and metalloids will typically involve

- Nutrient addition to enhance phytoplankton growth, metal uptake, and removal by sedimentation;
- Organic/nutrient addition to support direct microbial metal reduction/precipitation; and/or
- Organic/nutrient addition to enhance microbial sulfate reduction and facilitate metal sulfide precipitation.

This chapter discusses the scientific basis for bioremediation technology and important aspects to think about when considering the bioremediation processes, and provides case studies where this technology has been implemented.

SCIENTIFIC BASIS FOR TECHNOLOGY

The bioremediation approaches described above rely on either adsorptive concentration of contaminants to cellular biomass and removal from the water column by sedimentation, or the enhancement of microbial respiration to produce a chemically reducing environment, which is a process sometimes referred to as in situ redox manipulation (ISRM). The primary goal of ISRM is to create reducing conditions in the water column, which will induce a change in the valence state of the target elements, and thereby cause the precipitation and/or detoxification of the contaminants. In essence, the technology is based on significantly speeding up the natural process of lake eutrophication by adding a significant mass of nutrients over a short time span. In this chapter, the scientific basis for these approaches is discussed and case studies where the approaches have been applied are presented.

Adsorption and Sedimentation

Algae, fungi, and bacteria can be effective for adsorption and concentration of heavy metals and radionuclides (Costerton et al. 1995; He and Tebo 1998; Konhauser et al. 1993, 1994; Mullen et al. 1989, 1992; Warren and Ferris 1998). Remedial approaches that exploit this phenomenon are focused on enhancing microbial growth via the addition of limiting nutrients, typically nitrogen and/or phosphate, which in turn increases the mass of metals accumulated by the biota (Wang and Dei 2001). As microbial cells die off, the biomass and adsorbed metals are removed from the water column via sedimentation and sequestered in the lake sediment. Deep meromictic lakes can favor this approach as contaminants once entrained in the sediments are less likely to be recirculated to the water column under permanently stratified conditions.

Case study: DJX uranium pit lake, Saskatchewan, Canada. Large mesocosm studies to evaluate this approach have been conducted at the DJX uranium pit lake in Northern Saskatchewan (Dessouki et al. 2005). In these studies, eight cylindrically shaped mesocosms measuring 4 m in diameter and 8 m in depth, with a closed, tapered bottom, were suspended in the DJX uranium pit at Cluff Lake (Saskatchewan) and filled with contaminated water. The surface water contained elevated concentrations of cobalt, copper, molybdenum, nickel, uranium, radium-226, and zinc at or above the Saskatchewan Surface Water Quality Objectives (SSWQO). Initial surface water measurements indicated that phosphorus was likely the limiting nutrient for algal growth and the mesocosms were amended with different amounts of potassium phosphate. Over the 35-day experiment, algal growth was rapid in fertilized mesocosms, and a statistically significant decline in the surface water concentrations of As, Co, Cu, Mn, Ni, and Zn was reported. Surface water concentrations of ^{226}Ra, Mo, and Se showed no relationship to phosphate load. Based on the results of the studies, the authors estimated that SSWQO could be met in approximately 45, 65, 15, and 5 weeks for Co, Ni, U, and Zn, respectively. These studies provided proof-of-principle that the simple addition of limiting nutrients could be effective for pit lake treatment.

In Situ Redox Manipulation

The primary goal of this treatment method is to add sufficient organic carbon and macronutrients (typically N and/or P) to the pit lake to enhance microbial growth and facilitate a change from aerobic, oxygen-dependent respiration to anaerobic respiration supported by the reduction of metals and/or sulfate. All life on earth is maintained by a complex series of oxidation–reduction (redox) reactions collectively called respiration. In higher forms of life, such as mammals, respiration involves the oxidation of an organic carbon compound coupled exclusively to the reduction of molecular oxygen as the terminal electron acceptor. In contrast to higher organisms, many bacteria inhabit environments where molecular oxygen is unavailable and have adapted by coupling the oxidation of an organic substrate to the reduction of numerous compounds, including nitrate, numerous metals and metalloids, sulfate, and carbonate. Heavy metals and radionuclides known to be directly or indirectly reduced by bacteria include Mn(IV), Fe(III), Se(VI) and Se(IV), As(V), Cr(VI), Hg(II), Tc(VII), V(V), Mo(V), Np(V), Au(III) and Au(I), and Ag(I).

The coupling of carbon oxidation with the various potential electron acceptors follows a predictable order, which is based on the energetics of the reaction (i.e., how much energy is available from the redox reaction). For example, molecular oxygen is a much stronger oxidant than sulfate and there is considerably greater energy available from carbon oxidation coupled to the reduction of O_2 than there is when sulfate is used as the terminal electron acceptor. Some of the important biogeochemical reactions have been previously summarized in Table 13.2 in Chapter 13, where CH_2O is the empirical formula used to represent the carbon form utilized by microbes, such as a carbohydrate (sugar).

When organic carbon is added to a system, the bacteria that use oxygen in respiration will generally out-compete those that use less energetically favorable terminal electron acceptors and therefore dominate the system. Once oxygen has been consumed, if nitrate is available, denitrification will be the dominant process. After nitrate is depleted, then manganese and iron-reducing bacteria will become dominant, and so on down the chain of potential terminal electron acceptors until sulfate reduction becomes the primary terminal electron accepting process.

ISRM as a remediation technology takes advantage of these respiratory pathways to promote direct and/or indirect metal reduction and precipitation, and the removal of nitrate and sulfate.

Direct metal reduction and precipitation. Several metals and metalloids, including iron, manganese, chromium, arsenic, selenium, and uranium, are reduced directly by bacteria that use them as terminal electron acceptors in anaerobic respiration (Fendorf et al. 2002; Lovley 1993; Nealson and Myers 1992; Newman et al. 1998; Oremland et al. 1999; Tebo and Obraztsova 1998; Wielinga et al. 2001). The bacteria involved in these reactions appear to be ubiquitous in nature. Direct reduction of chromium, selenium, and uranium can be exploited for pit lake remediation as the conversion of these elements from oxidized valence states to reduced valence states can result in formation and precipitation of solid phases as described by the following reactions:

$$3CH_2O + 4CrO_4^{2-} + 8H^+ \rightarrow 3CO_2 + 4Cr(OH)_{3(s)} + H_2O \qquad \text{(EQ 19.1)}$$

$$2CH_2O + 2HSeO_3^- + 2H^+ \rightarrow 2Se^0_{(s)} + 2CO_2 + 4H_2O \qquad \text{(EQ 19.2)}$$

$$2CH_2O + 2UO_2(CO_3)_2^{2-} + 2H_2O \rightarrow 2UO_{(s)} + 5HCO_3^- + H^+ \qquad \text{(EQ 19.3)}$$

Chromium and selenium have a relatively high redox potential and are predicted to be reduced along with or just following denitrification. Uranium reduction is predicted to occur under iron-reducing conditions and under conditions less reducing than sulfidogenic. When precipitated out of the water column and deposited to the lake sediments, the solid phase species can be protected from reoxidation by the accumulation of organic material and burial in the sediments or by establishment of a permanently stratified lake with anoxic bottom waters.

Case study: Sweetwater pit lake, Wyoming, United States. This approach was successfully implemented at full scale at the Sweetwater pit lake in Wyoming (Paulson et al. 2002). The Sweetwater pit lake was formed by flooding the Sweetwater open pit uranium mine following the cessation of dewatering in 1983. Prior to treatment, it was oligotrophic and contained approximately 1.2 million m^3 of water with concentrations of selenium at 0.46 mg/L and uranium at 8.1 mg/L, which were above the applicable standards set at 0.05 mg/L and 5.0 mg/L, respectively. Kennecott Uranium Company treated the water by adding approximately 498 metric tons of sugars, fats, proteins, alcohols, and nutrients from October 19 to December 22, 1999 (Paulson et al. 2002). The average concentrations of dissolved Se and U in the water column decreased to 0.01 mg/L and 4.33 mg/L, respectively, by October 24, 2001 (Paulson et al. 2002). The appearance of an orange Se^0 precipitate along the lakeshore provided dramatic evidence for the reduction of selenate to elemental selenium.

Sulfate reduction and precipitation of metal sulfides. Treatment of mining influenced waters (MIWs) by promoting the activity of sulfate-reducing bacteria to add alkalinity to the system and precipitate metal sulfides has been employed at numerous sites. Some of the earliest work on using sulfate-reducing bacteria (SRB) for treating heavy metal contaminated waters was done by J.H. Tuttle and others focusing on the treatment of acid rock drainage (ARD) resulting from coal mining (Tuttle et al. 1969). The process for metal removal through sulfate reduction is described by the following general reactions:

$$2CH_2O + SO_4^{2-} \rightarrow 3HCO_3^- + H_2S \qquad \text{(EQ 19.4)}$$

$$H_2S + Me^{2+} \rightarrow MeS_{(s)} + 2H^+ \qquad \text{(EQ 19.5)}$$

where Me^{2+} represents a divalent metal ion, such as Cd^{2+}, Cu^{2+}, Fe^{2+}, Ni^{2+}, or Zn^{2+}. This process has been utilized in various technological applications, such as permeable reactive barriers (Benner et al. 2002), wetlands (Wildeman et al. 1994), sequential alkalinity reactor systems (Skousen et al. 1998), and bioreactors (Tsukamoto and Miller 1999) to successfully treat ARD by adding alkalinity and removing metals and metalloids.

Case study: Anchor Hill pit lake, South Dakota, United States. A two-stage, in situ treatment approach consisting of lime addition to neutralized pit lake water followed by the addition of organics and nutrients was implemented at the Anchor Hill pit lake at the Gilt Edge mine in South Dakota (Doshi 2006; Lewis et al. 2003; Park et al. 2006). This project was developed collaboratively by the U.S. Environmental Protection Agency (EPA) National Risk Management Research Laboratory, the Mine Waste Technology Program, and EPA Region VIII Superfund office to evaluate the potential for the passive remediation of pit lakes.

The Anchor Hill pit encompasses approximately 9.6 ha (water surface area of 1.8 ha), a maximum depth of about 26 m, and a relative depth of about 18% (Lewis et al. 2003). Prior to beginning the bioremediation project, the pit was filled with approximately 270,000 m^3 of MIW with a pH of 3.1. It is important to keep in mind that the Anchor Hill pit lake was essentially created for this experiment and therefore may not behave the same as a pit lake that has filled slowly and existed for many years. Pit treatment was conducted in two phases:

- Phase 1—Lake neutralization using NeutraMill lime treatment;
- Phase 2—Addition of organics to establish reducing conditions and facilitate biological nitrate, sulfate, and metals removal.

SRB have been isolated from environments that range in pH from 4 to 9.5 (Barton and Tomei 1995), and sulfate reduction has been observed in acidic (pH ≤3) sediments (Koschorreck et al. 2003). However, increasing the pH of the lake/sediment to pH ≥5 is expected to significantly enhance the rate of sulfate reduction and speed the bioremediation process.

Lewis et al. (2003) reported that the initial plan was to have a single organic addition to commence shortly after the neutralization step. The initial treatment conducted in the spring of 2001 consisted of the addition of 240 metric tons of lime, followed by a short equilibration period and the addition of the organic carbon material (130,900 kg molasses and 31,640 kg methanol). However, neutralization of the pit lake water column proved to be more complicated than anticipated and expected redox changes also progressed slowly. Consequently, from October 2001 to September 2002, the water column was supplemented with additional organic carbon material (both liquid and solid phase) and sodium hydroxide. The lake became chemically stratified by 2001, consisting of three zones: a well-mixed surface layer of ~3 m, a chemocline with gradually increasing conductivity from 3 to 12 m, and a relatively uniform layer below 12 m that is isolated from the surface (Doshi 2006). By April 2003, there was evidence of sulfate reduction as indicated by the presence of a black plume and the smell of hydrogen sulfide, and metals concentrations began to drop dramatically. In 2005, surface water from the lake was discharged from the pit lake to Strawberry Creek, meeting all applicable South Dakota state water quality criteria (Doshi 2006).

Lessons learned from the bioremediation treatability study at the Anchor Hill pit lake included the following (Doshi 2006; Lewis et al. 2003):

- Passive treatment of pit lakes using lime neutralization and ISRM shows promise for treating MIW.
- Effective mixing of the lime and/or other alkaline amendments into large pit lakes can present a significant challenge.
- Significant time (a year or more) may be required to achieve lake neutralization and for reducing conditions to develop.
- The onset of sulfate reduction may be further delayed by the presence of other electron acceptors, especially nitrate.
- The addition of excess organic carbon can lead to the accumulation of hydrogen sulfide in the water column, which may present health and safety concerns.

PIT LAKE BIOREMEDIATION: APPLICATION AND ISSUES

Bioremediation of mine pit lakes that contain water with elevated concentrations of metals, metalloids, nitrate, and sulfate is likely an appropriate treatment option in many if not most instances. In previous sections the scientific basis for the technology was discussed and case studies where the processes have been applied were provided. This section provides a discussion of the factors that can assist in assessing the applicability of bioremediation to a specific site.

The implementation of any remedial technology often follows a selection process where several technologies are evaluated and compared and will typically entail evaluation of numerous site-specific factors in the selection of a preferred remedial approach. The case studies provided previously in this chapter offer insight into factors that may need to be considered when assessing in situ bioremediation as a strategy. Some of the questions typically addressed in the selection process are posed here.

What Are the Advantages, Disadvantages, and Limitations of In Situ Bioremediation?

Potential advantages include

- Simplicity: There is little required infrastructure; therefore, engineering costs can be minimized and time between initial planning and implementation can often be reduced. Application of organic and inorganic nutrients can be as simple as broadcasting a solution containing these compounds over the pit lake via a hose or water cannon and allowing natural physical and chemical processes to mix and distribute the nutrients throughout the water column.
- Elimination of solids/sludge handling and disposal: Contaminants are precipitated to and sequestered in the pit lake sediments, which eliminates the need to handle and dispose of process solids.
- Cost-effectiveness: A significant advantage of an in situ treatment approach is the potential for large cost savings. Unfortunately, the case studies presented previously did not provide details for project costs. Bioremediation is an emerging technology with limited full-scale applications, and thus a discussion of average costs for this remedial approach is not possible. However, estimates suggest that 50% or greater cost savings over traditional water treatment technologies can be obtained. These cost savings can be realized as a result of the limited engineering and elimination of solids handling and disposal.

- Broad applicability: With one or a combination of the above bioremediation approaches, many of the contaminants typically associated with mine operations and pit lakes can be effectively removed from the water column.

Potential disadvantages include

- Metals not removed from the system: Because the metals are not removed from the system and are not degraded or destroyed, they will remain within the system for perpetuity. Thus, the potential exists for the metals and metal-sulfides that have been precipitated to be reoxidized and again released to the water column. Case studies provided previously suggest that this may not present a significant problem, and measures such as addition of solid phase organics to the sediment can eliminate this potential disadvantage by maintaining anoxic conditions within the sediment. In addition, mixing and oxidation is unlikely in stratified lakes. However, because the case studies presented have not been monitored over long time periods (e.g., decades), the long-term stability of such precipitates should be an avenue for future research.
- Production of high levels of hydrogen sulfide: Addition of excess organic carbon to stimulate sulfate reduction can produce elevated levels of hydrogen sulfide. Because of health and safety concerns with high concentrations of sulfide gas, additional measures may be required, such as the addition of ferrous iron to bind the sulfide or an oxidant to oxidize the sulfide back to sulfate, to remove the H_2S from the system.
- Level of predictability: The physical and chemical processes involved with many traditional water treatment technologies (e.g., coagulation/precipitation, ion exchange) are well understood. Bioremediation relies on the stimulation of microbial growth and reactions to remove contaminants. There will likely be diverse populations of microorganisms present in the pit lake living in a dynamic environment. Current understanding and ability to control microbial population dynamics is limited, and thus the ability to predict which populations and reactions will predominate in the system is also, to some extent, limited. In other words, in most cases, microbial growth can be stimulated but engineering the correct species distributions to achieve remedial goals may be problematic.

Limitations

As with any remediation or treatment technology, there are potential limitations and concerns that need to be considered during an evaluation process. It is difficult to anticipate all such limitations and concerns and it is unlikely that they could all be addressed in this chapter. However, some of the most common concerns are discussed briefly here.

- Presence of requisite microbial populations: There is often concern whether the necessary microorganisms will be present at a site or whether they will need to be added. It has been this author's experience that these organisms are ubiquitous in nature and he has yet to encounter a site where exogenous organisms needed to be added.
- Applicability of process in cold climates: Because this process relies on biological reactions, it is often believed that they are limited in cold climates. This is typically not the case as many bacteria have adapted to function over a wide range of temperatures. The Sweetwater pit was treated in late fall when lake temperatures were low and just before ice over. It was thought that the formation of the ice actually enhances the process by limiting oxygen diffusion into the water column.
- Factors that affect cost: Potentially the most important factor that will impact cost is the presence of nontarget electron acceptors, oxygen and nitrate. If the target is removal of

metals, either directly or via sulfate reduction, in lakes that are oxygen saturated and/or contain high levels of nitrate, greater than 90% of the organic material added will be consumed to remove the oxygen and nitrate. Pit lakes that are anoxic at depth and low in nitrate will require considerably less organic addition than those that are oxygen saturated and contain high levels of nitrate.

- Requirement for stratified and anoxic lakes: There is not a fundamental requirement that a lake be permanently stratified and/or anoxic for bioremediation to be successful. Provided sufficient organic carbon is added to consume the oxygen and/or nitrate, the system can be shifted to a highly reducing environment that will support the previously described microbial processes. Treatment of well-mixed and oxic lakes will require the addition of greater amounts of organic substrate and therefore bioremediation would be achieved at a greater cost than the treatment of lakes that are initially anoxic. In addition, if mixed water columns return to oxidizing conditions, the precipitated metals/sulfides could be reoxidized, and therefore the long-term effectiveness of the treatment approach would be less predictable. The stability of the precipitated solids will depend on how rapidly the lake returns to oxidizing conditions and to what extent the precipitates are buried within the sediments and thus potentially protected from reoxidation.

CONCLUSIONS

In situ bioremediation has the potential to provide a simple, highly efficient, and cost-effective alternative for the treatment of mine pit lakes containing concentrations of metals, metalloids, and other constituents above regulatory standards. The bioremediation approaches discussed previously can be used alone or in combination to treat a wide variety of contaminants. Although this treatment approach is still in the early stages of development, the scientific basis underlying technology is now understood to a level that will permit assessment of applicability to diverse sites. The remoteness of many mining operations and the associated pit lakes and the large volumes of water often contained in the lakes that may require treatment as the open pits fill makes this passive, in situ approach to treatment an attractive alternative.

Careful planning and implementation of measures to prevent the formation of pit lakes with poor water quality may be the most cost-effective strategy for mine closure as preventive measures can be much less expensive than remediation (Castro and Moore 2000). However, when implementation of preventive strategies at new operations is not practical or when faced with the challenge of treating an existing pit lake, in situ bioremediation is an approach that may provide the most practical treatment alternative.

The availability of models that could help predict the effects of pit lake manipulations, such as addition of organic carbon, would be helpful, and there is currently the need for geochemical models that integrate microbial reactions. Current models (e.g., MINTEQA2, PHREEQC) are based on the principles of thermodynamic equilibrium, which typically does not occur in natural waters between redox pairs, meaning our current modes of geochemical prediction do not accurately address these reactions. The limnological model CAEDYM may be the closest model currently available, but better models would improve our predictive ability. There is still a large need for more basic information about these reactions and these organisms before such models can be developed; hopefully this will remain an area of active research.

USEFUL CASE STUDIES

Useful studies on pit lake bioremediation are listed in Table 19.1.

TABLE 19.1 Case studies on pit lake bioremediation

Pit Lake Name and Location	Reference
DJX uranium pit lake, Saskatchewan, Canada	Dessouki et al. 2005
Sweetwater pit lake, Wyoming, United States	Paulson et al. 2002
Anchor Hill pit lake, South Dakota, United States	Lewis et al. 2003; Doshi 2006; Park et al. 2006

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PART VI: REMEDIATION

CHAPTER 20

Backfilling

R.D. Williams

INTRODUCTION

Backfilling of open cut and open pit mines is a means of avoiding pit lake development altogether, and has often been thought of as a solution to many of the environmental issues associated with the closure of large-scale open pit mining operations. Some of the environmental issues related to open cut and open pit mines include the formation of pit lakes that may have limited utility on account of poor water quality, and visual impacts and physical hazards due both to the remaining open cut or open pit highwalls and benches and associated waste dump complexes outside the pit. The objectives for pit backfill may be to eliminate or reduce some of these issues (Throop 1991; California State Mining and Geology Board 2007). Figure 20.1 may be a useful way to envision possible pit backfill objectives.

The pit highwalls and associated waste dump complexes are often terraced and have a distinctly artificial appearance due to the flat tops, long even side slopes, and low angle diversion ditches that often characterize waste rock dump complexes. These features generally do not blend in well with the natural surrounding landscapes. Figure 2.1 in Chapter 2, and Figures 20.2, and 20.3 display some of the features that are typically associated with open pits. These may include large pit lakes potentially with low-quality water, exposed pit highwalls, and related waste rock dumps.

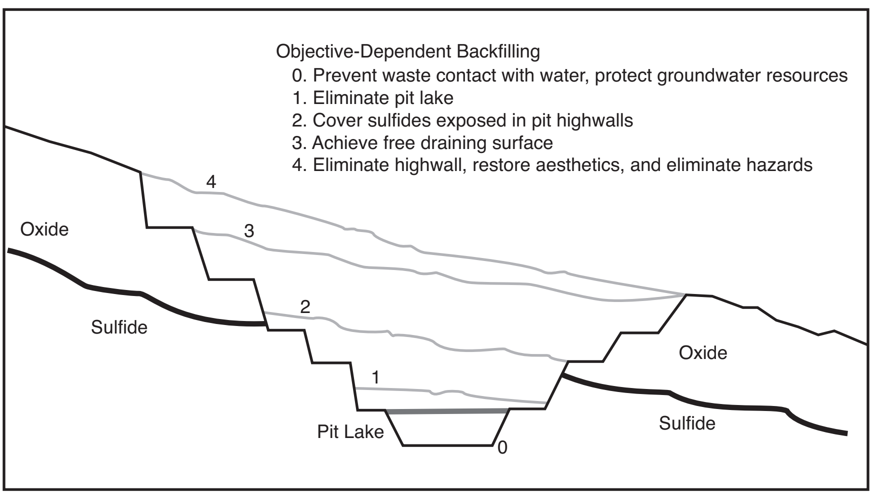

FIGURE 20.1 Possible pit backfill objectives

FIGURE 20.2 Golden Sunlight mine, Whitehall, Montana, showing pit highwalls and typical waste rock terrace features

FIGURE 20.3 An example of the structural and lithologic complexity at the Golden Sunlight mine, Whitehall, Montana. Each of the structural elements must be evaluated and characterized for groundwater flow potential as one evaluates potential backfilling issues.

The perception that backfilling is an easy way to resolve these problems has an appeal as a way to "put it back the way it was," but the reality of pit backfilling is often far more complex than that, as this chapter will explain.

CHARACTERIZATION REQUIREMENTS PRIOR TO BACKFILL CONSIDERATION

Characterization requirements related to pit backfill are at least as complex as those needed for the routine characterization of waste rock associated with the actual mining operations at open pit mines. Characterization work can help in determining whether the material is suitable for use as backfill or has the potential to cause future environmental issues. Where impact to groundwater is a potential issue, proposed backfilling can require a more detailed analysis of groundwater characteristics and flow paths.

Pit Floor and Wall Rock

The characterization work in support of any proposed backfill of an open cut or open pit should start with detailed geologic mapping and sampling in order to accurately evaluate any potential interaction between saturated or unsaturated backfilled materials and the open cut after backfilling. Geologic mapping and sampling should include detailed pit floor and wall rock lithology, including detailed mineralogy, detailed structural information including all faults and joints, their orientation and characteristics, and, as noted later in this chapter, their hydraulic importance. Another critical element to consider is the presence of any existing underground workings in the vicinity of the pit floor and highwalls. These open workings act as direct hydraulic conduits, thereby affecting subsurface flow patterns as discussed in the "Groundwater" section later in this chapter.

Fill Material

Fill material could include waste rock, spent ore from acid or cyanide leaching, or mill tailings. Backfill characterization data should be available from waste rock evaluation work performed before and during mining. In the event spent ore from leaching or mill tailings are used as backfill materials, this same type of characterization data might be available. This routine characterization data could include either static or kinetic testing to determine acid-generating potential, salt loading, and metals release rates. In the event this work has not been done, the baseline data collection will need to be conducted to develop a scientifically sound waste rock characterization program. The characteristics of the potential backfill material may have changed after it was placed in waste dumps through continued oxidation, settling, and physical weathering. Characterization work for backfill material must consider not only the geochemical characteristics of the material as it exists, but also the physical, engineering, and hydraulic characteristics of the material. This characterization work should be performed on the material prior to use as backfill and predictions made as to the material's future properties after placement as backfill, both physically and geochemically. This prediction of future performance is a critical element for developing a scientifically sound backfill program, but there may be a significant element of uncertainty in any such prediction. Regardless, whether the material to be used as backfill is potentially acid generating or not, a detailed understanding of these issues is critical.

Additional backfilling considerations. Most backfilling is done by simply placing stockpiled waste material back into the pit without complex engineering to refine objectives or performance. More complex engineering solutions are possible in order to meet specific pit backfill performance objectives. These can include using more permeable surrounding backfill to avoid a driving head through the backfilled waste, zoned layering, or a "hydraulic cage" encouraging groundwater inflow into the backfill that is collected and pumped (Robertson and Shaw 2007). Amending or blending backfill with neutralizing material is also possible. The use of amended backfill does not necessarily resolve issues related to trace metal mobility, so continued testing is likely to be required to ensure that performance requirements can be met (S. Shaw, personal communications, 2008).

Groundwater

Groundwater quality and quantity are two of the most critical issues that must be evaluated in determining whether a backfill program will solve or create environmental issues. A detailed understanding of groundwater flow paths, quality, and flow rates is necessary to determine whether the backfill will maintain a saturated or unsaturated state. This information is important in order to determine the effects of interactions between the backfill and groundwater. If the material to be used as backfill has been characterized as acid generating, or has a potential to leach constituents, groundwater interactions with the backfilled material can cause negative environmental impacts, especially in the short term. If the groundwater quality is good upgradient from a backfilled pit, then contact with backfill material can cause adverse impacts to the downgradient groundwater quality, and the groundwater may need to be pumped and treated in order to avoid groundwater quality violations. If the upgradient groundwater quality is poor, then contact with backfill material is unlikely to improve the groundwater quality, and it may still be necessary to pump and treat groundwater to avoid groundwater quality violations. In some instances, however, backfilling that is susceptible to oxidation and acid generation under unsaturated conditions may gradually show a reduction in solute release as groundwater inundates the backfill and reduces the availability of oxygen. Any potential for attenuation of impacted water downgradient of a backfilled pit should also be evaluated, although downgradient attenuation possibilities are often exhausted by previous natural acid rock drainage.

In summary, because of the importance of groundwater as an environmental issue, it is critical to have a detailed understanding of groundwater quality, quantity, and flowpaths, both upgradient and downgradient from a backfilled pit.

Surface Water

Surface water management in a backfilled pit is an important consideration in order to avoid excess infiltration into backfilled material. Often the infiltration calculations for adjacent waste rock dumps will be applicable for a backfilled pit, assuming similar reclamation practices are used. Excess infiltration into backfilled material could have adverse impacts, either in terms of adding to groundwater that might ultimately need to be pumped and treated, or impacting the physical characteristics of the backfilled material causing settling or piping in the backfill. Another important aspect of surface water management is keeping any upgradient surface flow away from backfilled areas. This is particularly important for areas where the backfill abuts rock faces. If settling or piping (development of preferential flow paths) occurs, either adjacent to rock faces or elsewhere on the backfill, it will generally require repair in order to avoid continued erosion or excessive infiltration into the backfill. This could require construction of haul roads and diversions onto previously reclaimed surfaces with all the concomitant expenses that such an endeavor would entail.

A detailed discussion of surface water management practices is outside the scope of this handbook, but the two principal alternatives for managing surface water are

1. Engineered structures, typically including constructed diversion ditches and drainageways; and
2. Artificial topographies using geomorphic principles.

Engineered structures will typically require routine monitoring and repair. Artificial topographies are a more recent development but may offer reduced long-term monitoring and maintenance costs.

BACKFILLING METHODOLOGIES

Two primary techniques for backfilling are

1. End dumping material into the open cut or pit (Figure 20.4); and

2. Hauling material into the pit and placing it (i.e., backhauling), essentially the reverse of the actual mining (Figure 20.5 and 20.6), perhaps even using the same mining equipment and haul roads.

Backhauling material may have the advantage that the placed material is compacted as trucks and earthmoving equipment traverse over it. It may also be possible to use cast blasting in upper portions of the highwall to reduce the highwall slopes to an artificial talus, helping to give the backfilled pit more of a natural appearance.

FIGURE 20.4 End dumping material to backfill a portion of the Suprise pit at the Zortman–Landusky mine, Montana

FIGURE 20.5 Equipment sloping backfill material in the Gold Bug pit, Zortman–Landusky mine, Montana

FIGURE 20.6 OK-Ruby pit backfilling to cover sulfide-bearing rocks in pit floor, Zortman–Landusky mine, Montana

Each of these methods may have problems depending on the ultimate backfilling objectives and material. Hauling material back into a deep pit can present equipment problems with haul trucks. Brakes on fully loaded haul trucks traveling downhill may not meet operational safety requirements. It may be necessary to check with the equipment manufacturer before commencing any backhauling under those conditions. In the event it is not possible to haul fully loaded trucks, hauling expenses will increase. End dumping may have problems with settling, which may then require extensive and expensive repairs of engineered diversions to ensure that surface water runoff can be effectively managed. Sorting problems commonly associated with end dumping (a tendency for coarser material to segregate at the bottom of the dump) may also result in the creation of an unintended French drain, altering the hydraulic characteristics of the backfill. Preferential flow paths can also develop as a normal by-product of dump construction. Detailed evaluations of the construction methods and resulting hydrologic properties of the backfill material must be assessed prior to backfilling in order to avoid adverse conditions that may promote undesirable hydrologic characteristics.

ENVIRONMENTAL IMPACTS

Favorable Impacts

The most obvious favorable environmental impact from pit backfilling is an improvement in the visual resources as compared with non-backfilled pits (see Figures 20.7 and 20.8). As these photos illustrate, an improvement in the visual resource at a site often does not mean that the mine features are not obvious, just that the overall visual impact has been reduced. Where pit floors and wall rock are acid producing, it may be possible to limit infiltration and contact with meteoric waters by covering highwalls and pit floors and sloping them so as to drain water away from problematic highwalls or pit floors. The use of clay liners can enhance this effect. In some cases, these actions can limit acid rock drainage, but the extent of their effectiveness depends on the site-specific groundwater flow paths. Backfill incorporating geomorphic principles, as noted previously, will more closely resemble original topography. In the more mountainous areas of the

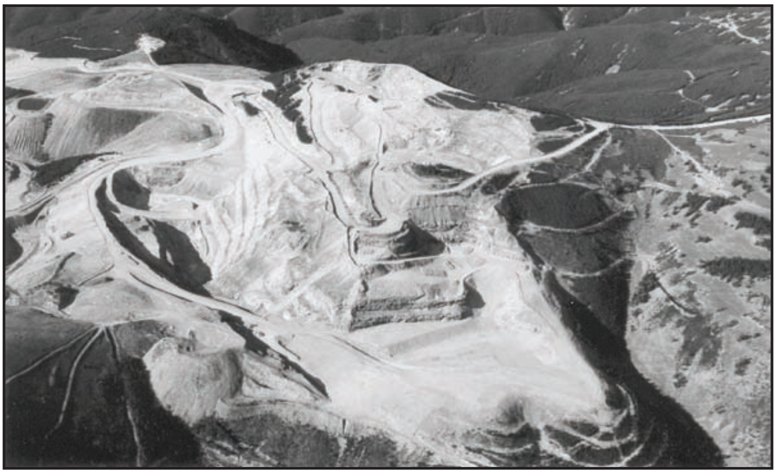

FIGURE 20.7 OK-Ruby pit at Zortman–Landusky mine, Montana, prior to backfilling, July 2000

FIGURE 20.8 OK-Ruby pit at Zortman–Landusky mine, Montana, after backfilling, July 2005

western United States, there are no examples of large backfilled pits where original topography has been restored.

Given the generally negative connotation associated with pit lakes resulting from open pit mines, pit lake avoidance can be considered a favorable impact. Water loss through evaporation of a pit lake in an arid region constitutes an irreversible loss of groundwater, which is unquestionably an adverse impact in an arid region. Evapoconcentration of pit waters over time exacerbates water quality issues and is also an adverse impact. Backfilling would be a favorable impact by virtue of avoiding both of these issues.

Another potential advantage of backfilling is the elimination of highwalls. This may not be an important consideration at mine sites where public access may be controlled or prevented, but at other sites the elimination of highwalls will reduce concerns about public safety.

Unfavorable Impacts

Unavoidable, unfavorable impacts related to pit backfilling include increased noise, dust, air quality impacts, and energy consumption during backfilling, which are similar to impacts associated with the actual mining operations. Other possible unfavorable impacts related to pit backfill include unanticipated degradation of ground- and surface waters because of incomplete characterization work, which could include unanticipated flushing of prior oxidation products in the backfill. These can result in long-term, costly, and complicated water collection and treatment requirements. Backfilling can also delay final reclamation alternatives until near the end of mine life, reducing opportunities for concurrent reclamation.

It might be possible that sulfides submerged in a saturated zone would result in reduced oxidation, but unless the sulfides were fresh, which would be unlikely in a backfill setting, prior oxidation products would flush from the backfill for an extended period.

If backfilling involves the use of blending neutralizing material—limestone ($CaCO_3$), lime (CaO), or calcium hydroxide ($Ca(OH)_2$)—with backfilled material, there may be associated environmental impacts with mining and transporting the neutral material to the backfill site, particularly if limestone is quarried for this purpose.

Other considerations. Another potential consideration for pit backfill, particularly where studies suggest that the backfill may pose an environmental risk, may be climate impacts. Although there have been no detailed studies or evaluations of potential climate impacts specifically related to pit backfilling, calculations based on engine manufacturers data can be used to estimate CO_2 production on a metric-tons-per-kilometer basis. Although a detailed evaluation of these parameters is outside the scope of this chapter, relatively simple calculations suggest that large diesel-powered haul trucks ($\approx$140 metric tons rated) produce on the order of 21.6 kg CO_2/km (76.7 lb/mi). Obviously for an extensive backfill project, CO_2 and related greenhouse gas production could have substantial climatic impacts.

ECONOMICS

The economics of backfilling can depend extensively on mine sequencing and mine pit development. Mines where ore is extracted from a single pit that deepens throughout the mine life are poor candidates for backfilling from a strictly economic standpoint for the following reasons. Waste dumps will likely have some sequential reclamation throughout the mine life, so backfilling at the completion of open cut or open pit operations would mean deconstructing reclaimed waste piles and backhauling into the pit. As discussed previously, there may be safety issues associated with hauling fully loaded trucks down into a pit. If fully loaded trucks cannot be used, hauling costs would rise commensurately. End-dumping or cast-blasting may reduce haul distances and avoid downhauling of fully loaded trucks. In either case, haulage expenses would be approximately what expenses were as the waste was moved the first time. Obviously, this extensive double or triple handling of material near the end of mine life would dramatically adversely impact mine economics.

In other cases where mines consist of a series of open pits, it may be possible to backfill a closed pit with waste rock generated from an active pit to accomplish reclamation goals and reduce reclamation and other operational expenses. In this case, backfilling economics may be quite favorable as backfilling could actually lower operating and reclamation costs through limiting overall surface disturbance and simpler reclamation. This technique also minimizes the need for additional free-standing waste rock piles.

EXISTING EXAMPLES

Because of the generally unfavorable economics, pit backfill is not a common reclamation feature with open cut or open pit mines. However, there are several mines that have used backfill as a reclamation practice at least in part. Studies evaluating backfill performance are not common, but some have been done as noted in the following lists. Mine sites that have used backfilling include

- Zortman–Landusky mine in the Little Rockies, Montana (Figures 20.4 to 20.8)
- Beal Mountain mine, Montana (Figure 20.9)
- Flambeau mine, Wisconsin (Figure 20.10 and 20.11)
- Golden Cross mine, New Zealand (Figure 20.12 and 20.13)
- Summitville mine, Colorado (Figure 20.14 and 20.15)

Photo courtesy of the U.S. Forest Service.

FIGURE 20.9 The backfilled South pit at Beal Mountain, Anaconda, Montana

Source: Reproduced with permission of the Wisconsin Department of Natural Resources.

FIGURE 20.10 Flambeau mine, Wisconsin, United States, during operations

Source: Reproduced with permission of the Wisconsin Department of Natural Resources.

FIGURE 20.11 Flambeau mine, Wisconsin, following backfill and reclamation

Other mines in the United States that have reportedly used pit backfill include

- Basin Creek mine, Montana
- CR Kendall mine, Montana
- Treasure mine, Montana
- Yellowstone mine, Montana
- San Luis mine, Colorado
- Richmond Hill mine, South Dakota

These mines used pit backfill to accomplish a variety of different objectives (MDEQ/BLM 2004, 2007) with varying degrees of success. Perhaps the only constant from all the examples was the difficulty of predicting the postmining water quality. A review of pit backfill conducted for an environmental impact statement in Montana (MDEQ/BLM 2004, 2007; Kuzel 2003; Gallagher 2003, SENES Consultants Ltd. 1995) found relatively few examples of pit backfill. When the search was refined to approximate large open cut or open pit mines with detailed water quality information in the western United States, even fewer examples were found (Canonie Environmental Services 1993, 1994; Duaime et al. 2004; Maest 2003). The studies by Gallagher (2003) and Kuzel (2003) also determined that predicting postmining water quality was a persistent problem, highlighting the importance of detailed and accurate characterization work to determine whether or not pit backfill will result in long-term water quality issues.

SUMMARY

Pit backfill can have utility as a reclamation alternative where the backfill material will be deposited well above regional groundwater, where the backfill material is geochemically inert or neutralizing, or where visual resource considerations are important factors. Pit backfill can be questionable where the backfill material is potentially acid generating, where groundwater flow paths and interactions with backfill cannot be determined with certainty, or where the backfill material may be saturated. The decision of whether or not to backfill a pit that will become saturated will determine whether a pit lake will develop after mine closure, and the costs and benefits

Photo courtesy of Jeff Mauk, University of Auckland, New Zealand.

FIGURE 20.12 The active Golden Cross mine, New Zealand, in 1997

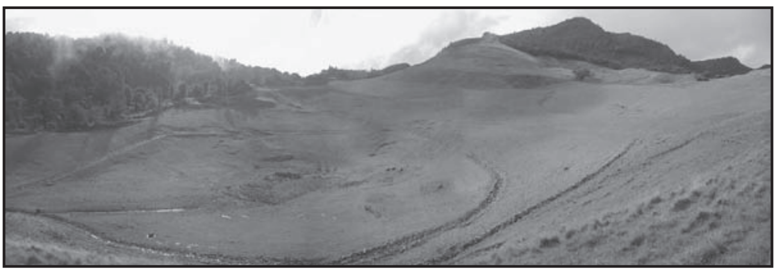

Photo courtesy of Devin Castendyk, State University of New York, College at Oneonta.

FIGURE 20.13 The backfilled and rehabilitated Golden Cross mine, New Zealand, 2002. Notice the extensive surface drainage network.

of backfilling should be carefully compared against costs and benefits of pit lake development when making this decision. For saturated backfill, it may be necessary to either plan for pumping and treatment of impacted waters or include this possibility as a contingency. Saturating backfill may be a positive move to limit oxidation, but it must be carefully evaluated to ensure that existing oxidation products are not flushed downgradient into unimpacted water resources.

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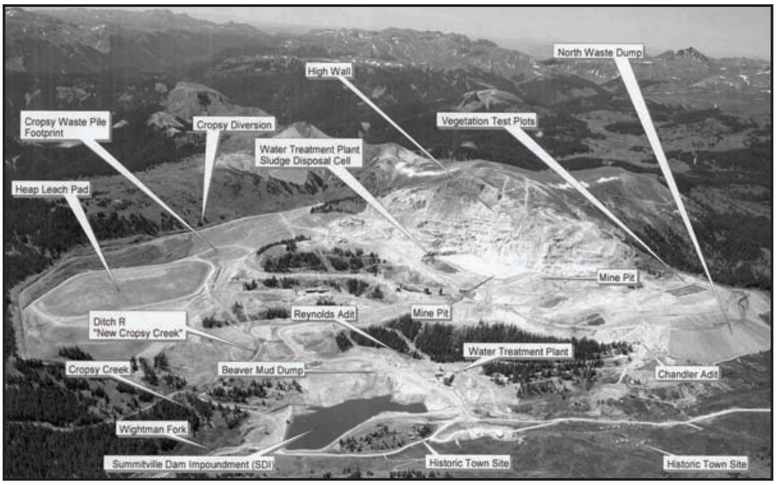

Photo courtesy of Colorado Department of Public Health and Environment and EPA Region 8.

FIGURE 20.14 Summitville mine, Colorado, United States, prior to reclamation

Photo courtesy of Colorado Department of Public Health and Environment and EPA Region 8.

FIGURE 20.15 Summitville mine, Colorado, after reclamation. Notice the minor use of pit backfill against the open cut face.

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CHAPTER 21

Induced Meromixis

M. Schultze and B. Boehrer

INTRODUCTION

Vertical circulation of lakes is responsible for the distribution of dissolved substances in the water column. In some cases, a full overturn and subsequent homogenization of the water masses on a regular time scale may be an advantage, whereas in other cases, a restriction of vertical circulation to the upper water body may be preferred. As a consequence of the limited matter exchange through the water column, a chemically different bottom layer is formed, referred to as the monimolimnion, in contrast to the regularly circulated water above, referred to as the mixolimnion. Lakes possessing a monimolimnion are called meromictic. Circulation features of lakes, mechanisms creating stratification, and the limnological terminology have been introduced in Chapters 5 and 6.

In general, two goals can be aimed for when considering the artificial implementation of a permanent stratification in a pit lake: first, confining undesired substances in deep waters, and second, the creation of special chemical conditions for water treatment.

DEPOSITION OF UNDESIRED SUBSTANCES IN DEEP WATERS AND SEDIMENT

Subaqueous deposition of undesirable substances in a monimolimnion appeals as a simple and straightforward approach. Water quality problems are confined to a deep layer, which is not accessible to the public and does not interfere with the visible flora or fauna. Still, the affected water body is accessible for survey and further treatment. If transport of matter out of this affected zone is small enough, the rest of the water body may form a healthy environment, where the organisms can dwell without being greatly affected by material deposited in the monimolimnion.

After a nondegradable substance has entered the monimolimnion, it is released only by upward diffusion across the chemocline, the erosion of the monimolimnion, seepage into groundwater, or sedimentation. The risk involved for the mixolimnion and groundwater must be assessed by quantifying these transport pathways. Monimolimnia tend to accumulate substances and hence increase their water density, which can actuate convective transport into the groundwater domain. Sources and sinks define the temporal evolution of hazardous species concentration in the monimolimnion.

In most cases where undesired substances, such as heavy metals, cannot be degraded entirely by microbes, the removal from the water body and the permanent burial in the sediment may be the only viable option. If a special chemical climate that prevents dissolution of sequestered metals will not be sustained for an indefinite time, then any potentially hazardous material must be buried by more recent precipitates or sediment capping. Special cases of metal-rich layers may even be considered for mining under changed market conditions.

WATER TREATMENT UNDER CHEMICAL CONDITIONS OF A MONIMOLIMNION

The treatment of water under the special conditions of a monimolimnion utilizes three aspects of meromixis

- Relative isolation of the monimolimnion from neighboring water layers, especially the mixolimnion;
- Isolation of the monimolimnion from the atmosphere, which favors anoxic conditions; and
- Tendency of substances to accumulate in monimolimnia, which may make a treatment in the monimolimnion more efficient than in the mixolimnion (see also Figure 5.10 in Chapter 5).

Sulfate removal, metal removal, and neutralization can be achieved by microbial sulfate reduction under anoxic conditions. Addition of organic substances may be necessary to stimulate microbial reduction. Metals may be added for the formation of sulfides if concentrations of sulfur and metals are not balanced. Iron compounds are usually the cheapest potential additives. The following equation is a gross summary for the initial and end reagents of several partial microbially mediated intermittent steps. More details should be read from the appropriate literature (e.g., Schlesinger 2005). In Equation 21.1, CH_2O is used to represent the stoichiometric form of organic substance involved in reduction reactions for simplicity, and Me^{2+} indicates a bivalent metal ion (e.g., Fe^{2+}, Zn^{2+}, Cu^{2+}).

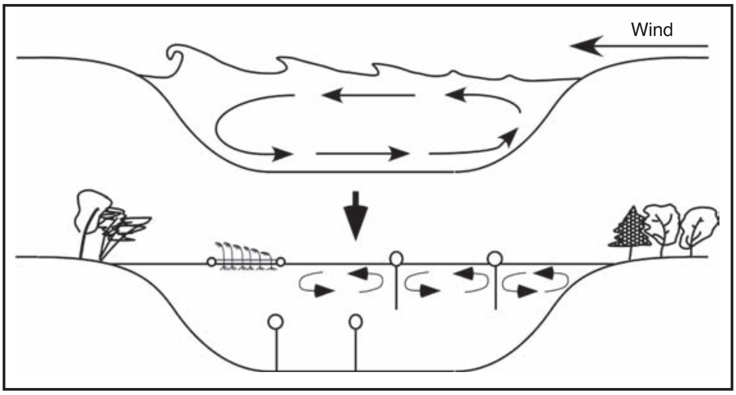
$$SO_4^{2-} + 2H^+ + Me^{2+} + 2CH_2O \rightarrow MeS\downarrow + 2H_2O + 2CO_2\downarrow \qquad \text{(EQ 21.1)}$$

Ferrous iron is a preferred additive because ferric iron oxidizes pyrite in the anoxic monimolimnion, as reported in the Berkeley pit lake in Butte, Montana, United States (Madison et al. 2003). Ferric iron precipitates settling from the mixolimnion are dissolved without microbial reduction in the acidic monimolimnion of the Berkeley pit lake, although the monimolimnion is anoxic. Obviously, the availability of reducing substances (e.g., organic substances) is not high enough in the monimolimnetic water to ensure a complete microbial reduction of settling ferric iron. The ferric iron oxidizes the sulfur of the pyrite when coming into contact with remaining pyrite of the side walls. Additional acidity is liberated in this way in the Berkeley pit lake. Peine et al. (2000) and Koschorreck et al. (2007) observed similar effects of oxidation by ferric iron in the sediment of German pit lakes counteracting neutralization. To remove accumulated metals from the monimolimnion, the addition of H_2S or alkaline sulfides may be required, if microbial sulfate reduction does not provide enough sulfide for metal precipitation.

Under certain conditions, the application of flocculation agents, such as those used for water and wastewater treatment, can help remove trace contaminants that cannot easily be precipitated. The use of aluminium-based flocculants requires addition of neutralizing substances to compensate for the liberation of acidity by aluminum hydrolysis. Iron-based flocculants are generally less suitable because precipitates of ferric iron may be redissolved because of microbial iron reduction or may act as undesired oxidants as mentioned previously.

OPTIONS FOR PRODUCING AND FOSTERING MEROMIXIS

The most obvious way to induce meromixis is the introduction of water masses of different salinities. As an alternative, salt can also be added directly to the deep water. The best time for



Source: Klapper et al. 1996.

FIGURE 21.1 Inducing density stratification in a previously holomictic pit lake with floating and submerged barriers and by shore afforestation

introducing dense water is during the stratification period, when vertical transport is restricted and salt can be placed at the desired depth. Further options to create or sustain meromixis are known from natural lakes and discussed in Chapter 5 and in works by Boehrer and Schultze (2008). However, no intentional implementation has been reported in the scientific literature.

Once created, the density gradient needs to be sustained against the continuous erosive action of internal waves, currents, and turbulence on the stratification. A process is required that produces stability continuously or regularly. If the source for the stability production ceases, the permanent stratification will disappear within a limited period of time. The rate of change of chemocline erosion and exchange with mixolimnion and groundwater are controlled by local conditions.

Wind stress at the water surface introduces turbulent kinetic energy into the water body (Figure 21.1). In addition, it can force upwelling and create internal waves, which cause current shear and mixing. Stands of forest around a lake reduce wind impact and mixing (Figure 21.1). Depending on conditions, shading from the wind by trees may be effective for lakes of a size up to 1 km in the direction of the winds. For larger lakes, forests will have limited effect on mixing.

To reduce turbulence and mixing in the bottom zone, Klapper et al. (1996) proposed the obstruction of the flow path by curtains attached to the lake bottom. In this way, appropriate conditions for sulfate reduction can be created in deep waters. To these authors' knowledge, this system has never been tested. The advantage of this approach is that is can result in a defined and controlled volume of the monimolimnion. Alternatively shaping the lake basin prior to flooding in a way to promote meromixis requires early and good communication with mining companies at a time long before decommissioning.

If a lake protected from wind is fertilized sufficiently, it may turn meromictic as a result of enhanced biological productivity (endogenic meromixis; Scharf and Oehms 1992; Wüest et al. 1992). Organic material is produced in the surface layer, then settles and decomposes at the sediment water interface. Part of the material is dissolved as carbon dioxide (CO_2) and contributes to the density of the deep water (see Chapter 5). More substances, like iron, calcite, or manganese, can be involved in the process and contribute their part to the density difference. In general, endogenic meromixis occurs under conditions of small rates of accumulation in monimolimnetic water and weak density differences. Consequently, endogenic meromixis will predominantly become established in lakes well protected from the wind.

Overturns may also be impeded by temperature stratification. Ikedako Lake is a very deep (233-m) caldera lake of a nearly round surface of 11 km^2 on the southern Japanese island of Kyushu. It has not turned over since 1985 because winters have not been cold enough to force surface temperatures colder than the abyss of the lake. By 1989, anoxic conditions were established in the deep water. Similarly, if a deep enough lake is flooded with river water during the cold season, introduced water can be colder than surface temperatures during the rest of the year. If deep enough, such a lake will not overturn until geothermal heating and diffusive transports have raised deep-water temperatures sufficiently.

Lake Malawi in East Africa also is permanently stratified by temperature. The reason for the permanent stratification is suspected to be related to the production of cold waters at the southern end of the lake during southern hemisphere winter (Vollmer et al. 2002). Without overturning the lake, these cold waters proceed to the monimolimnion and sustain the temperature stratification. As mine lakes are smaller, channelling of cold water, perhaps through pipes, would be needed to keep a lake permanently stratified.

RISKS OF MEROMIXIS

Because of precipitation out of the mixolimnion or release from the sediment surface, substances are enriched in the monimolimnion. Partial or complete overturns caused by extreme weather events or huge landslides of pit walls can mix these substances into the mixolimnion and affect the living domain of organisms dramatically.

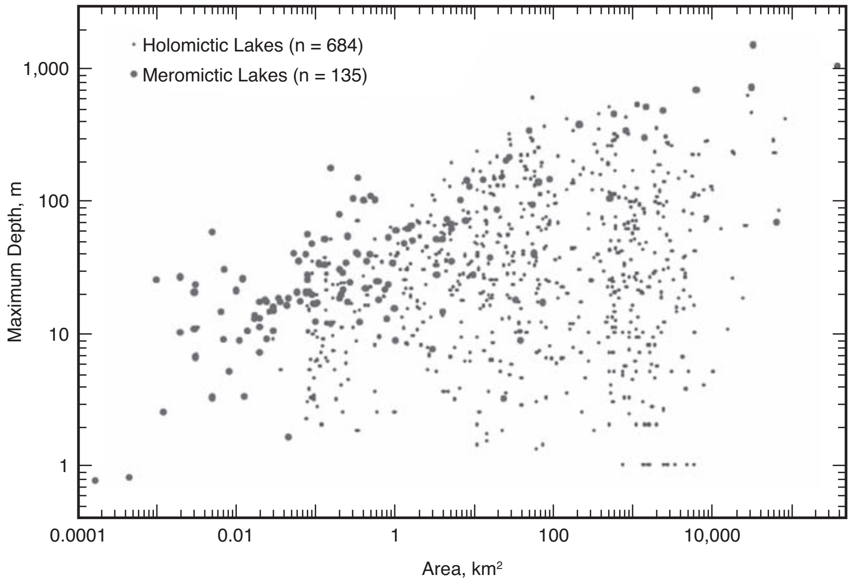
If metals are insufficiently high in concentration to form sulfides, high hydrogen sulfide (H_2S) can accumulate in a monimolimnion. Concentrations of more than 300 mg/L H_2S are found in the monimolimnion of Lake Hufeisensee, Germany. A sudden release of larger amounts into the mixolimnion would cause fish kills through its toxicity, or later through oxygen depletion following oxidation of H_2S through bacteria. Methane resulting from microbial decay of organic substances would cause strong oxygen depletion as well. On the contrary, if dissolved metals are highly enriched on account of missing sulfate reduction, they may also cause toxic effects or strong oxygen depletion (only iron or manganese) when suddenly mixed into the mixolimnion.

Accumulated CO_2 was released from the monimolimnia of Lake Monoun in 1984 and Lake Nyos in 1986, both located in Cameroon, Africa, in a catastrophic event, called limnic eruption, which cost the lives of more than a thousand humans (Kling et al. 1987). Murphy (1997) made predictive calculations about whether such a catastrophe could also happen in pit lakes and concluded that such an event was unlikely but could not be excluded. Extreme weather conditions, landslides, earthquakes, and rock bursts or blasts in neighboring mines might be possible trigger mechanisms for limnic eruptions in pit lakes.

PREDICTION OF PERMANENT STRATIFICATION

Based on information of basin shape and density stratification, the effects of vertical fluxes from turbulent transport and erosion of a chemocline can be estimated probably better than within one order of magnitude (von Rohden and Ilmberger 2001). More accurate predictions can be made if observations exist from a neighboring lake of similar size under the same weather conditions (Boehrer et al. 2000), or if observations are available from the lake itself under different conditions, such as at a low water level prior to complete filling or before implementation of treatment.

In consideration of the fact that the degree of wind force is related to the surface area of a lake, researchers have used ratios of surface area and maximum depth to indicate tendency to



Source: Adapted from Joehnk 2001.

FIGURE 21.2 Maximum depth versus surface area of holomictic and meromictic lakes

meromixis. For lakes within a similar climate, correlations have been found between recirculation depth and surface area. However, Joehnk (2001) included 135 meromictic and 684 holomictic lakes (most of them of natural origin) in a display of maximum depth versus surface area (Figure 21.2). The clear conclusion was that meromictic lakes could not be separated from holomictic lakes by means of a simple line. Castendyk and Webster-Brown (2007) drew the same conclusion for pit lakes after comparing the relative depths of holomictic and meromictic pit lakes (relative depth is the ratio of maximum depth to mean diameter of the surface, expressed in percent; see Chapter 6).

At a typical mining site, the aquatic environment of mine lakes has been dramatically impacted in the decades before mine closure. Recovery of water table and solute transport of substances set free by geochemical weathering will take years, decades, or even centuries to settle into a new equilibrium. Hence, it can be suspected that geochemical cycling is more pronounced than in most natural lakes. High concentrations of dissolved solids in waters often flow into pit lakes. Differences of concentrations and densities are high between inflow water from different origins. If high-density waters reach the lake bottom and recharge fast enough, they may form a monimolimnion. As a consequence, mining lakes have been found to be more prone to develop meromixis than natural lakes (Boehrer and Schultze 2006; Castendyk and Webster-Brown 2007; Sanchez Espana et al. 2008). Hence, without quantifying the effect of these processes for a particular setting of a mining lake, the situation is even more complex. Simple rules of thumb are not suited for a meromixis prediction.

If prognostications of stratification and circulation patterns of a lake are required before final filling, accuracy of numerical predictions will greatly profit if observations from a nearby existing lake are available. A quantitative prediction for the chemocline in two neighboring meromictic mining lakes was done by Böhrer et al. (1998). The stratifying agent was conservative; that is, no geochemical transformations needed to be included. In addition, groundwater information was detailed. The prognostication aimed at a time period of 100 years. Ten years after flooding the mines (2008), the predictions and observed evolution of the lake stratification are still very close (Schultze and Boehrer 2008).

Stevens and Lawrence (1998) numerically investigated the meromixis in the Brenda mine lake in Canada. Joehnk (2001) did a predictive modeling for the onset of meromixis in the natural Lake Ammersee in Germany under the assumption of a changing climate. As both simulations were based on hypothetical assumptions about groundwater inflow and climate change, a critical comparison with the real situation was not possible. Colarusso et al. (2003) reported predictive modeling of a pit lake in Tennessee (United States), using the CE-QUAL-W2 model. Their model results agreed with field data relatively well. However, the calculated weakening of the chemocline was not realistic, and suitability for long-term prediction might be questioned. The complexity of such a prediction could be inferred from Jellison et al. (1998), who conducted a numerical simulation for the future evolution of the stratification in Mono Lake, California, United States. Castendyk and Webster-Brown (2007) used the limnologic model DYRESM (described in Chapter 9) to predict future meromictic conditions in a pit lake that will develop in the Martha gold-silver mine, New Zealand. This prediction can only be checked after the mine closes and the pit is flooded at some time in the next decade.

None of the previous simulations included geochemical transformations and their effect on stability of meromixis. Recently, models have been developed to include geochemistry, such as MODGLUE by Müller (Müller 2003; Werner et al. 2006; see also Chapter 14) and DYRESM-CAEDYM (Salmon et al. 2007; see also Chapter 11). These models can be used for simulating the geochemical evolution in mining lakes. The link between geochemical transformations and resulting changes of the density stratification, however, has not been properly implemented yet, let alone tested against real data. Hence, it is not yet established whether these models are capable of predicting the long-term (decades) meromictic status of a lake in which stratification is controlled by geochemistry.

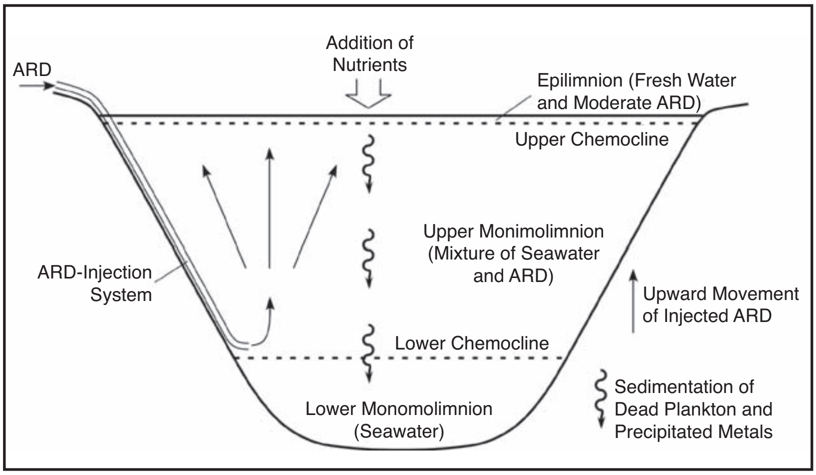
The importance of basin shape for forming monimolimnia was mentioned previously in this chapter. Quantifying morphometric effects for fostering meromixis is difficult. Only under special conditions can some aspects of meromixis be simulated, and specialists may draw some conclusions for the evolution of meromixis in particular cases.

Predictions require knowledge about future conditions of external factors impacting on the lake stratification, such as the vegetation around the lake, climate and weather (temperature, radiation, wind, precipitation), hydrology (inflow and outflow of surface water and groundwater, local runoff, evaporation), water chemistry of all inflows, geology and geochemistry of the lake environment, and planned or already established human activities. The temporal variability and extreme events, such as storms and floods, have to be considered for any reliability of a meromixis prediction. After a meromictic lake is established, monitoring programs should be implemented.

EXAMPLES FOR INDUCED MEROMIXIS

Intended Implementation of Meromixis for Subaqueous Deposition and Treatment of Mining Influenced Water—Island Copper Mine, British Columbia, Canada

The most prominent, and possibly the only example for an intentionally induced meromixis, is the 340-m-deep Island Copper Mine pit lake (see also Chapter 17). In 1996, the pit was filled with seawater, and finally capped with a 7-m layer of fresh water. Acid rock drainage (ARD) that had collected above the lake surface was injected at a depth of about 220 m for disposal. Because of its low density and positive buoyancy, the ARD rose through the seawater and mixed the saline water body above the introduction depth. A three-layer system was formed (Figure 21.3). Density



Source: Adapted from Fisher and Lawrence 2000, 2006; Poling et al. 2003.

FIGURE 21.3 Conceptual depiction of Island Copper Mine pit lake

differences between layers were high, and, as a consequence, the exchange of dissolved substances between layers was low (Fisher and Lawrence 2000; Muggli et al. 2000).

The idea behind the meromictic design was the creation of a surface layer that fulfilled legal requirements for the water quality (Poling et al. 2003). A layer was formed below for the deposition of ARD contaminated with loads of copper and zinc. The pH of the stored seawater and the expected generation of alkalinity by sulfate reduction would precipitate iron and aluminium as oxyhydroxides. The flocculation would absorb and co-precipitate heavy metals. Cycling of manganese and iron between oxic surface water and the anoxic monimolimnion would also contribute to the co-precipitation of heavy metals according to the mechanism described in Figure 5.10 of Chapter 5. Under expected anoxic conditions of the monimolimnion, sulfate-reducing bacteria could transform heavy metals to insoluble sulfides.

Adding nutrients to the surface water boosted the production of organic material. Dissolved metal ions were incorporated by plankton and removed from the water column with settling organisms. The decomposition of this organic material required oxygen in deeper waters, and finally the organic material was used for the reduction of metals and sulfate. In the intermediate layer of Island Copper Mine lake, oxygen consumption was smaller than anticipated. As a result, further fertilization was implemented, so that the intermediate layer was close to anoxia by December 2004. In addition, fresh water was directed through the epilimnion to stabilize the density stratification of the lake. A quantitative evaluation of the transports of dissolved substances can be found in works by Poling et al. (2003), Fisher and Lawrence (2006), and Wen et al. (2006). Chapter 17 provides a detailed discussion of the Island Copper Mine pit lake.

Examples of Unintended Meromixis in Pit Lakes

In several pit lakes, meromixis was unintentionally created by filling or treatment. Flooding or treatment of pits has turned the lakes meromictic (e.g., Lake Goitsche, Germany, Boehrer et al. 2003; Anchor Hill pit lake, South Dakota, United States, Park et al. 2006). The water quality of some lakes has profited from the meromictic conditions, even if it was not induced intentionally.

In some cases, meromixis had become established before treatment measures occurred. Regarding meromictic conditions as an advantage, treatment was implemented in these cases to preserve the stratification. In Lake Vollert-Süd, Germany (Stottmeister et al. 1999, 2002), the contaminants were deposited in the lake sediment and in the monimolimnion after flocculation without breaking the permanent stratification of the lake. In other cases, meromixis was present and used as a beneficial condition for remediation (e.g., South mine pit, Tennessee, United States, Wyatt et al. 2006; Anchor Hill Pit lake, South Dakota, United States, Park et al. 2006; Lake Rävlidmyran, Sweden, Lu 2004).

CONCLUSIONS

Until 2007, the Island Copper Mine lake in Canada has been the only example where meromixis was induced on purpose. There are several lakes where meromixis occurred as an unintended consequence of flooding or treatment. One example is Lake Vollert-Süd, Germany, which was meromictic before the remediation was implemented. Conserving meromixis was part of the remediation strategy to confine undesirable substances to the monimolimnion (Stottmeister 2008). Especially in cases where meromixis would be undesirable, considerations of the factors causing meromixis as described previously may be valuable.

Though accessibility of the stored monimolimnetic waters for survey and further treatment is a definite advantage, the difficult prognostication for the temporal evolution of meromictic behavior limits the cases where induced meromixis can be implemented. Risk assessment has to include extreme scenarios, such as the potential for huge landslides from side walls into the pit lake or extreme weather conditions, such as heavy storms or unusually cold or windy winters, because these events can affect the estimated release of dissolved substances to the mixolimnion (Böhrer et al. 1998). Only if the consequences of a partial or total overturn of the monimolimnion are acceptable can induced meromixis be considered as a remediation strategy.

Usually strong density stratification limits the transport between monimolimnion and mixolimnion. However, this only lengthens the time scale for the release of substances. Over long enough time periods, all substances not degraded or permanently deposited in the sediment will be released into the mixolimnion by diffusion and erosion of the chemocline or into the groundwater domain, where treatment is more complicated and where they may become a hazard for a wider area. Only in cases where these constraints are properly considered can induced meromixis be implemented as a remediation strategy.

Intensive monitoring of meromictic pit lakes and further research are necessary to improve numerical pit lake models. Table 21.1 lists studies of induced meromixis in pit lakes. Improved predictive tools may result in a much broader applicability of induced meromixis as a treatment and management option for contaminated pit lakes and mine site effluents.

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TABLE 21.1 Case studies on induced meromixis

Pit Lake Name and Location	Reference
Island Copper Mine, Vancouver Island, BC, Canada	Boehrer and Stevens 2005 Fisher and Lawrence 2000, 2006 Muggli et al. 2000 Pelletier et al. (Chapter 17 of this handbook) Poling et al. 2003 Stevens et al. 2005

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CHAPTER 22

Sustainable Development of Open Pit Mines: Creating Beneficial End Uses for Pit Lakes

C. McCullough, D. Hunt, and L. Evans

INTRODUCTION

Factors governing the decision to rehabilitate pit lakes to reduce or remove impact, or to go further and develop them into a tangible environmental or social benefit, fall largely into either regulatory requirements or development incentives. Because of the nonscientific nature of end use development processes, reports on most pit lake developments for social or environmental beneficial end uses occur predominantly within the nonscientific literature in a case-study format (Walls 2004). Consequently, examples of pit lakes relinquished as public amenities are more inaccessible than other forms of pit lake research. Reports on the processes that companies have followed to achieve pit lake relinquishment as a public amenity are even rarer, although they are likely to be the same as those used in general rehabilitation planning.

In order to ascertain the effect of these influences on end use development, a literature review was undertaken and e-mail enquiries requesting opinion and experience were made to other researchers, stakeholder groups, regulatory agencies, and industry representatives. Because of commercial sensitivity, details of actual companies or operations have not been given. This chapter outlines incentives and regulatory requirements, describes recommended planning and implementation procedures for various end use options, discusses opportunities and challenges presented by developing postclosure beneficial uses, and provides case studies of pit lakes developed for social, economic, and environmental end uses.

REGULATORY REQUIREMENTS FOR DEVELOPMENT OF A PIT LAKE AMENITY

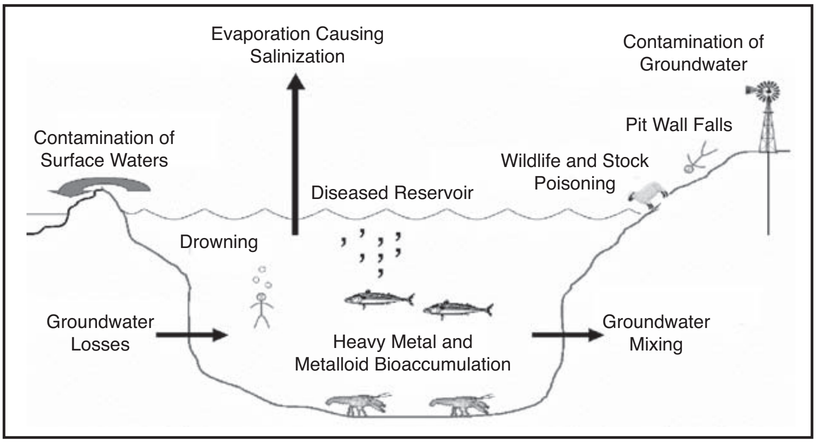
Many variables (often poorly predicted in themselves) remain that require consideration in developing and interpreting the accuracy and reliability of geochemical models. For example, pit lake volume and water quality evolve over time, whether the lake is slowly filled by groundwater, or more rapidly by river diversions. Changing rainfall and groundwater levels may then alter drainage budgets to pits that have formed lakes, particularly in arid and semi-arid areas. In addition, pit lake bedrock geochemistry and water quality often vary significantly between different pits, even within the same region. There is often little consideration given to the long-term effects of erosion and weathering around the pit. The duration of time that a pit lake may maintain a particular water quality may also differ between regions, regulatory agencies, and geochemical models (Miller et al. 1996). Consequently, the main stumbling block for developing a postclosure pit lake amenity in many cases is that water quality issues (although they may have been carefully modeled) may remain too uncertain to provide assurance to regulators and stakeholders to allow or support certain end use development proposals, such as for wildlife habitat.

Although mine closure guidelines from different countries and states generally target long-term liability for pit lakes in their various forms of amenity, regulatory agencies often do not have a clear approach to resolving this issue of long-term uncertainty. For example, water quality guidelines for a particular site may depend on the type of end use chosen for the site and site-specific variables. The ANZECC/ARMCANZ (2000) guidelines provide a good example of this, with separate water quality guidelines for drinking water, aquatic ecosystems, recreation, livestock water, irrigation, and so on. As such, it seems that a mining company could first predict water quality, then review available water quality guidelines for different end uses, and then choose an appropriate end use. Conversely, there are examples where pit lake end uses have been chosen by a mining company, but water quality modeling has shown that water quality guidelines will not be met for this end use (Oldham et al. 2006).

Mining companies are in the business of producing ores and metals and cannot maintain unnecessary ongoing involvement in their completed project areas and require clarification of project expense and liability associated with acceptable postclosure land-use options. Where a clear regulatory process exists, regulations influencing end use development may be expressed across a diverse range of criteria. These criteria may range from simple, generic, local water resource protection to drinking water standards (especially with groundwater resources in arid areas) (Miller 2002) to a more complex ad hoc approach dependent on pit location and local industry such as agriculture and silviculture (Bolen 2002).

Even though it is generally assumed that pit lakes will follow an autochthonous succession from young to mature lakes, resulting in lakes with a well-developed ecosystem (Kalin and Geller 1998), without confident predictability, pit lakes may still present significant health and safety issues for both the mining company and adjacent human and wildlife communities, for many hundreds of years following cessation of mining operations (Doupé and Lymbery 2005). Although long-term issues may not be considerations in shorter time frames that many regulatory authorities may be interested in, these longer time frames may represent a significant issue for the long-term sustainability of communities and environments. As such, the environmental and social liability that pit lakes represent to communities and the environment can be a significant legacy of the regional geography after mine closure.

Regulatory requirements governing available pit lake end use options also differ both between and within different countries as a result of both different regulatory regimes and the many different potential risks that pit lakes may represent to that particular region (Figure 22.1). For example, as many mines exist in remote, low-rainfall regions, such as central Australia's and similar arid regions of the United States and China, inappropriately managed pit lakes may represent a significant risk to the local human and environmental water resources (Brown 2003). Where communities reside nearby, pit lakes may also present hazards for recreational swimmers where there is a risk of drowning with the limited shallow margin (Hatch 2007). In agricultural areas, pit lakes may lead to poisoning and drowning of stock and wildlife where there is a risk of falls from the pit highwalls. In environmentally sensitive areas, mixing of local water resources with contaminated pit waters may lead to loss of biodiversity or ecosystem function (McCullough and Lund 2006). Particularly in drier regions, pit lakes may also be an ecological liability through supporting populations of feral animals. Consequently, significant rehabilitation may be required to turn a pit lake landscape from an industrial site to an acceptable public amenity or wildlife habitat (Kruger et al. 2002).



Source: Adapted from McCullough and Lund 2006.

FIGURE 22.1 Potential liabilities of pit lakes to communities and the environment

A select few countries and regions do clearly regulate end uses for pit lakes. This end use choice is usually determined by existing local economic or environmental interests. For example, end use amenities are broadly regulated in the Lower Lusatian lignite mining area of Germany on the basis of regional planning targets for land use for economic, environmental, and social (recreation) concerns (Dähnert et al. 2004). Nevertheless, these overall strategies may be inflexible for other end uses, including more local-scale interests (Kruger et al. 2002). For instance, social amenity end uses are less commonly specified as end uses for pit lakes, as these end uses are typically novel for the area.

In many countries, mining companies are given no prior advice on acceptable end use options. Rather, mining companies interested in developing an end use approach regulatory authorities with their preferred option, and then these agencies make judgments as to whether the end use is acceptable. Nevertheless, most developed countries and states are consistent in their requirement for mining companies to plan and/or rehabilitate to minimize or prevent entirely any potential deleterious effects of the pit lake water body on regional ground and surface resources (Miller 2002). Special regard from most general or ad hoc pit lake regulation is also especially given to protecting human and ecological communities from effects of the pit lake. For example, in Australasia, closure guidelines are based on the Australian and New Zealand Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand (ANZECC/ARMCANZ 2000) criteria (generally for a combination of drinking water, recreation, or ecological requirements). Such guidelines generally emphasize either a demonstration of null-negative effects of the lake when treated passively or active management for a point of compliance (Kuipers 2002). Nevertheless, where regulatory guidelines for natural systems such as natural lakes are applied to pit lakes (Axler et al. 1998), end use opportunities may be rendered unavailable as inappropriately high environmental and social values are placed on these artificial systems.

Lack of knowledge of state-of-the-art rehabilitation strategies and capabilities (such as remediation techniques) by regulators may also produce a strong deterrent for companies wishing to engage in end use development activities (McCullough et al. 2006). In some of these cases, mining companies may even need to incur themselves the risk of liability in order for beneficial end use development to be approved by regulators.

PERFORMANCE BONDS

Performance bonds may be collected prior to mine permit approval and held by a government pending appropriate rehabilitation and final relinquishment of mining leases on public lands to enforce protection of these surrounding resources and environments. Nevertheless, even in developed countries, bonds are often insubstantial relative to acceptable standards of rehabilitation. In addition, bonds are often particularly inadequate to achieve rehabilitation standards that meet social and environmental end uses. Performance bonds may present little direct incentive to a company's relinquishment performance; furthermore, these performance bonds often occur in regulatory environments with few specific guidelines for relinquishment of mine leases with pit lakes (Nguyen 2006). Even if bonds are taken, these bonds may even be released prior to achievement of adequate pit lake standards for relinquishment, such as when a regulatory agency believes that in the event of rehabilitation performance failure, the country/state can legally recoup enough monies to rehabilitate the site appropriately.

Under these circumstances, there may be little regulatory pressure to develop an end use benefit from a pit lake. Furthermore, many mining companies, researchers, and regulators themselves currently perceive regulation to often be more of an impediment than an incentive to development of social and environmental end uses from pit lakes. Nevertheless, although there may be genuine risks associated with some end uses (Doupé and Lymbery 2005), many pit lakes may represent a potential boon to their local community and environment (McCullough and Lund 2006).

INCENTIVES FOR DEVELOPMENT OF A PIT LAKE AMENITY

Although the target of a postmining landscape is generally to restore the affected areas to the environment of the previous landscape (Lögters and Dworschak 2004), this is not often practicable on account of high expenses for earthworks, extended nonoperational times to relinquishment, or simply because topsoils or other earth resources not available. Consequently, achieving a planned landscape of equal or even greater social and environmental value may be one way in which mining can positively contribute to a region's long-term sustainability.

The Mining, Minerals, and Sustainable Development (MMSD) Australia project (MMSD 2002) identified seven key issues facing the mining industry and proposed action to enhance the mining industry's contribution to Australia's sustainable development. These issues are as follows:

1. **Sustainability of the minerals sector** and its capacity to support social, economic, and environmental processes that underpin sustainable development;
2. Need to improve **governance** of the industry to deliver on the commitment to sustainable development and engage with stakeholders over issues of community concern;
3. Need to improve **resource valuation and management** of minerals resources and ecological values that drive development, and improve the social and cultural heritage values that enrich quality of life;
4. Need to improve **stakeholder engagement** in decision making relating to mining projects and operations;
5. **Fair distribution of costs and benefits** to ensure lasting equitable social benefits from the exploitation of mineral resources;
6. Promotion of **inter-generational benefits** by improving understanding of impacts on health, economic well-being, cultural and social relations of communities; developing

social baseline data; and establishing effective monitoring systems to measure long-term benefits to local communities; and

7. Promotion of the rights and well-being of **indigenous communities** by ensuring prior informed consent, incorporating traditional owners' consultation to mining proposals, and equitable distribution of mining benefits.

Adopting sustainability practices requires mining companies to plan for mine closure and work with communities to promote the development of appropriate and productive end uses of previous mine sites (McCullough and Lund 2006). In addition to the points itemized previously, this report also emphasized (a) community negotiation, (b) development of a site closure plan at the start of mining with active remediation during mining, and (c) postclosure resource use.

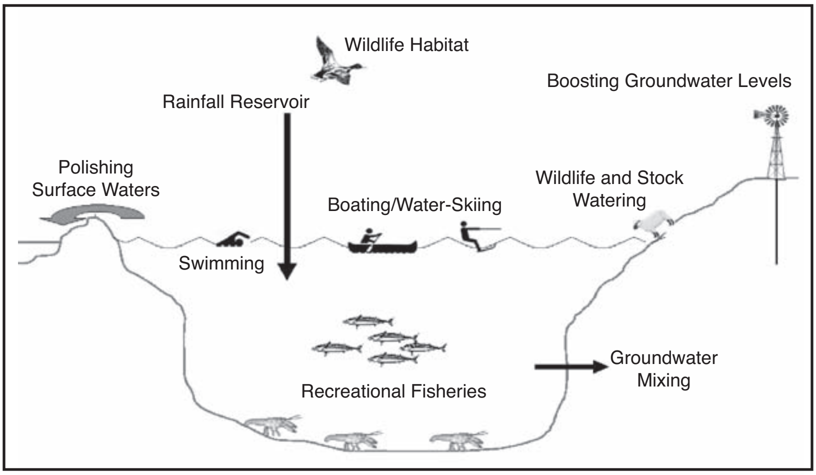
Encouraged by the global mining industry in the MMSD report (2002), the application of sustainable development as a growing business practice provides a framework through which incentives may be provided for developing a pit lake amenity. Sustainable development was a relatively unknown concept in the resources sector only 10 years ago. However, many resource companies—mindful of environmentally minded stock investors, and possibly responsive to criticism and embarrassed by a legacy of environmental and social problems—are increasingly responding positively to community pressure to move toward sustainability (Table 22.1). This is especially so of the large blue-chip miners (Davidson 2005). Companies in the mining sector are especially inclined to emphasize sustainability in their performance because of the greater level of environmental controversy and public scrutiny of their activities (Department of Environment and Heritage 2005).

It follows that pit lake management that only considers minimization of liability may therefore miss significant opportunities for realizing significant benefits, which these water sources can offer both during mine operation and in the future after mine closure (Johnson and Wright 2003; Doupé and Lymbery 2005; McCullough and Lund 2006) (Figure 22.2). A large incentive for development of beneficial end uses for some pit lakes may be that these lakes may yield an important, yet little-recognized way in which true mining sustainability to most or even all major stakeholder groups can be made over an indefinite or long period of time and in a mutually beneficial fashion.

Although there are usually few and in most areas no direct incentives for companies to develop postclosure lake social and environmental amenities, corporate sustainability offers a significant incentive process through which many different pit lake beneficial end uses may be realized. If there are no natural lakes in these regions, there may be little value in constructing pit lakes for an environmental end use unless migrating birds are likely to utilize the lake. However, in regions where natural water bodies such as wetlands or lakes have been degraded and lost, development of pit lakes for the benefit of local ecology may provide a way in which a rehabilitated project

TABLE 22.1 Examples of the increasing sustainability report size (numbers of pages) from some international mining companies from their increasing recognition of the importance of sustainability practices

Company	2002	2003	2004	2005	2006
Alcoa	44	70	56	25	26
BHP Billiton	66	116	164	384	522
Xstrata	Not produced at this time	Not produced at this time	88	108	110



Source: Adapted from McCullough and Lund 2006.

FIGURE 22.2 Some potential benefits of pit lakes to communities and the environment

area may conceivably contribute more to the biodiversity and ecological health of a region. For example, appropriately rehabilitated pit lakes have been found to provide large amounts of valuable wildlife habitat, such as for waterfowl (Walls 2004).

Social amenity opportunities for pit lakes, such as recreational swimming, are organic outgrowths of their communities and, although often unrecognized and unregulated by local authorities, are already well-established in many mining regions with reasonable pit lake water quality. However, health risks associated with pit wall failures can limit public access to lakes (see Chapter 23). The development of other pit lake amenities will require more foresight and planning. For example, sailing and water-skiing activities require access roads, boat ramps, and other such infrastructure. To be successful, such planned opportunities require specific and direct support from mining companies, regulatory authorities, and, most importantly, the willingness and acceptance of the local communities that they are intended to benefit. Where social and economical benefit can be demonstrated to communities from such ventures, strong support is likely. Infrastructure also already exists at mine sites for future industry to develop. For example, aquaculture and irrigation may provide for either new industries, or extensions to existing industries, directly contributing to local business ventures, employment, and income.

PLANNING FOR A PIT LAKE AMENITY

For pit lakes to be a viable relinquishment option for a company, the community, and the environment, a management strategy for the development and final form of the pit lake should be considered well before rehabilitation operations have begun (Evans and Ashton 2000; Evans et al. 2003). If a new mine is considered from the perspective of a beneficial landscape following mining in addition to extracting resources, then there is a much greater chance of obtaining an economically feasible and sustainable outcome (Viertel 2005). Generally, unless the company is conducting multiple mining operations in the vicinity of an abandoned mine, it becomes much more difficult economically to undertake major rehabilitation or remediation works as plant and labor for earthworks then must be specially brought in (Beutler 2003). A systems approach aimed at facilitating progressive mine rehabilitation over the life of the operation to return landforms to beneficial end uses in the shortest possible time is often recommended for this reason (e.g., ANZMEC/MCA 2000). Consequently, pit lakes need to be planned for, not only to minimize risks, but also to maximize opportunities for end use benefit.

A partnership approach to planning for beneficial outcomes is the preferred strategy, with mining companies, community groups, and government agencies having a significant input into the decision-making process (Evans 2006). It is essential that planning for pit lake relinquishment involves dialogue with all relevant stakeholder groups. Conflicts between end uses are likely, for example, where a well-sculpted lake may represent risks to the environment from low but ecologically significant levels of contaminants, although the community is not at risk and is able to utilize the lake for recreation.

The U.S. Environmental Protection Agency (EPA 2008) has identified four general but key conditions that have proved effective in the development of successful beneficial end uses for contaminated lands and, by inference, pit lakes (www.epa.gov/superfund/programs/aml/revital/index). These conditions include

1. The creation of a site reuse vision through community consultation,
2. A sustainable community involvement that is inclusive and driven by a community champion,
3. A process for monitoring outcomes, and
4. A close liaison with regulatory authorities (Table 22.2).

Compliance with regulatory requirements for the pit lake must be a priority consideration for relinquishment to a public amenity. These regulatory requirements typically address general considerations pertinent to long-term offsite risks to ground- and surface waters and other resources of nearby and regional communities and environments. Clearly, companies that show a willingness and competence to rehabilitate (i.e., through a thorough understanding of current processes, clear rehabilitation goals, and examples of ongoing rehabilitation) will demonstrate to regulators and stakeholders that relinquishment targets are both achievable and sustainable.

Given the technical nature of most regulatory and rehabilitation issues, scientific research through university and government multi/interdisciplinary research centers has much to contribute with respect to best practice processes of pit lake rehabilitation and end use (Beutler 2003). Where such collaborations do not deliver relevant, quality results to their funding organizations, it is likely that regulators and industry supporters may become disenchanted with future research collaborations of this nature. It is therefore important that industry and government both maintain an involved and executive role in directing research activity (Grünewald and Uhlmann 2004).

TABLE 22.2 Conditions and actions that contribute to a successful reuse of mined land and pit lakes

Condition	Description
Sustained community involvement	Includes at least two elements of a site champion and an inclusive stakeholder process
Creation of site reuse vision	Achieved through meetings, visioning sessions, and workshops involving local communities
Reuse process oversight	Requires the formation of a sponsor group (e.g., advisory committee) to oversee the development and identification of key individuals with responsibility for reuse development and stakeholder communication
Coordination with regulatory agencies	Reuse options will be governed by regulations. Information flow and advice between regulatory authorities and the development group is essential for a successful outcome.

Source: Adapted from EPA 2008.

TABLE 22.3 Factors influencing the viability of pit lake end uses

Factors Requiring Consideration	Implications of Factor
Lake area and volume	Places significant limitations on possible and desirable end uses
Water quality	A significant parameter affecting end use options, economics of remediation, and long-term pit lake liability
Geographical location	Climate and proximity to population centers influence choice of end use.
Proximity to natural water bodies	Implications of pit waters mixing with natural waters may be desirable or undesirable.
Pit lake development and/or remediation costs	A cost–benefit analysis for different end uses will be required if costs are significant.
Short- and long-term risks to mining companies and postclosure managers	Requires an explicit and formal risk assessment prior to development that is updated during remediation and postclosure, the results of which must be incorporated into the decision-making process
Stakeholder political pressure	May positively or negatively influence end use options
Regulatory requirements for rehabilitation and/or end use	Will generally exclude some end uses; however, may also direct end uses to align with regional end use models
Support from stakeholders for end use options	Although lack of support may not exclude some end uses, support will increase viability of other end use choices.
Water resource needs at local, regional, and national levels	Will influence regulatory and also stakeholder (e.g., community) decision making

Regular outputs of productivity, whether they are peer-reviewed journal articles or industry conference presentations, should also be encouraged, as these further serve to control quality regarding the relevance and usefulness of the research process.

There are a number of factors likely to affect the viability and choice of beneficial end uses (economical, social, or environmental) for any pit lake (Table 22.3).

An ongoing process of stakeholder agreement is also likely to be a notable facilitator to end use option development (RWE Power AG PBF Division 2004). The community consultative process often begins with a conceptual plan being developed by the mining company and presented to either an established or ad hoc community consultative committee. As well as the need for ongoing consideration of the changing stakeholder structure, there will also be a need to account for the changes to economic and technological aspects of development over the life of the mining project. This iterative process constantly refines the end use development plan so that operations are well directed both during and after mining operations cease (Figure 22.3).

CASE STUDIES

The following are some case study examples of best-practice mine pit closures from around the world that incorporate elements of a more holistic approach to closure than simple economics and regulation. In nearly all of the examples provided, closure was planned early in the life of the mine site, and the companies involved engaged with the local communities to determine final uses for the pits.

Collie Coalfield Lakes, South-Western Australia (Coal)

Collie township is located close to two major urban areas (50 km from Bunbury and 202 km south from the capital city of Perth). Although unintended, two acidic Collie pit lakes, Stockton Lake and Black Diamond Lake (pH 3.5–5), are used for recreational and tourism purposes and have become important tourist attractions for visitors to the region (Shire of Collie 2003).

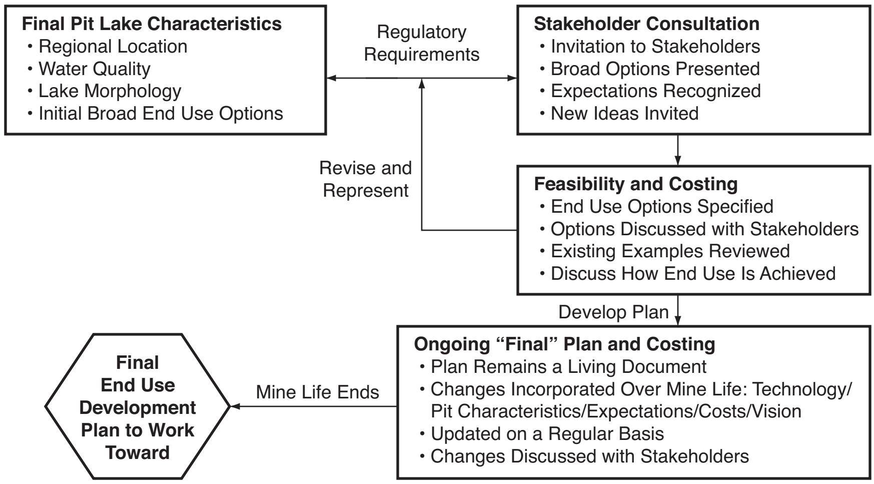


FIGURE 22.3 Planning steps for successful pit lake development and relinquishment to an accepted end use

However, visitors to the lake are warned to limit their exposure to the water owing to the high acid content (Shire of Collie 2003).

Premier Coal has developed the Lake Kepwari pit lake as a major recreation and conservation facility (Figure 22.4). Kepwari was formed after the closure of the Western No.5 mine site that was mined from 1970–1997 (APP Project Management 2003). It has a depth of 75 m with a water volume of approximately 32 GL (Ashton and Evans, n.d.). As part of the rehabilitation process and to make the site safe for recreational use, Premier Coal reshaped the former mine pit and battered the steep slopes and sharp drops that characterize many of the older pits in the Collie region (Department of Minerals Petroleum and Resources 2006). Substantial amounts of topsoil were added to the sides and an island refuge for waterfowl was also created, adding an additional $2 million to the rehabilitation costs (Ashton and Evans, n.d.). In addition, a motor sports and driver training complex has been developed on land near the lake. Premier Coal received the Banksia Award for Sustainable Development Leadership in the Minerals Industry in 2004 in recognition of its rehabilitation efforts around Lake Kepwari.

Premier Coal also needed to ensure that acidic water, which is a common feature in Collie pit lakes (Lund et al. 2006), was controlled and able to meet water quality guidelines for recreational activities (ANZECC/ARMCANZ 2000). If the lake had been allowed to fill slowly with groundwater, then it would have taken up to 100 years to fill naturally, with the likely result of a pH level of 3 to 3.5. This acidic pH range is typical of other new mining lakes in the region (Lund et al. 2006). Premier Coal, along with the relevant state government agency, decided to divert the Collie River South Branch into the lake to rapidly fill it. The company is hoping that the lake will eventually form a continuous part of the Collie River system (Ashton and Evans, n.d.). In the meantime, however, the lake may need to be allocated an annual 1.5 GL refill amount to maintain a pH level of 5.0 that is recommended for recreational use.

Planning for Lake Kepwari developments occurred close to cessation of mining activities, which has probably increased the cost of the development. Although there are some outstanding concerns with regard to water quality, this remains a good example of development of a pit lake for recreational end uses.

Photo courtesy of Clint McCullough.

FIGURE 22.4 Rehabilitation at former Premier Coal s mine site, Collie, Western Australia. Lake Kepwari with waterfowl island hidden by near shore.

Flambeau Mine, Wisconsin, United States (Gold/Silver)

Operated by Kennecott Mining Company (KMC), the Flambeau mine was located only 3 km from the city of Ladysmith in Wisconsin and only 40 m from the Flambeau River. The Flambeau River is an important resource for the community for recreational fishing, tourism, and as a wildlife habitat with aesthetic appeal (Fox et al. 2000). Because of its location near the river, the mine had the potential to be both an environmental and social disaster if the mine operation and closure phases were not handled in a sustainable manner (Chapter 20, Figure 20.10). The initial plan was for a mine life of 11 years and an open pit approximately 90 m deep, with the rehabilitation plan leading to a recreational lake as a beneficial end use (Fox et al. 2000).

Importantly for the operation, the closure of the mine was given priority from the initial planning stages, providing the local community with a sense of security with regard to the cessation of mining. It also provided the community with time to plan the development of alternative industries to provide employment when the mine closed. The local community, however, reacted unfavorably to the proposal, and the company chose not to proceed with a recreational lake. Stakeholder involvement had been identified as an important consideration for sustainable development according to MMSD (2002) (Fox 2004). KMC began planning for the mine again during the mid-1980s with a focus on protection of the Flambeau River as a cornerstone to gaining community approval and acceptance for the new development. This new plan allowed for a smaller pit with backfilling of the pit and contouring of the site back to its original condition upon closure and rehabilitation.

KMC established a Local Agreement and Conditional Land Use Permit with the three local government areas responsible for the development: City of Ladysmith, Rusk County, and the Town of Grant (Fox et al. 2000). Rehabilitation then began during autumn of 1996, and backfilling of the pit was completed the following year. The site was then contoured back to its approximate original state, along with the construction of wetlands for a wildlife sanctuary (Chapter 20, Figure 20.11).

KMC also entered into an agreement with the City of Ladysmith to establish a series of hiking trails on the mine site. A public walking trail was established in 2001 that demonstrates the rehabilitation of the site, passing through various wetlands, grasslands, and wooded areas (Fox et al. 2000). Using infrastructure developed by the mining company, including a rail line to service the park, KMC also developed an industrial park on part of the mining site in conjunction with the Ladysmith Community Industrial Development Corporation.

Golden Cross, Waihi, New Zealand (Gold/Silver)

Golden Cross is located near the mouth of the Waitekauri River, approximately 8 km from the township of Waihi in the North Island of New Zealand (Castendyk and Webster-Brown 2006). The mine is surrounded by farmland, native forests, and a pine plantation. A joint venture partnership between Coeur d'Alene Mines and Viking Mining (New Zealand) established a joint task force to examine sustainable postmining land uses for the site, with the process involving community consultation both prior to and during closure. One of the major issues for the site was the potential for cyanide contamination of local water supplies on account of runoff from the metallurgical processing facilities in this area of high annual rainfall of as much as 4 m/year (Department of Environment and Heritage 2003).

Environmental regulation for mining across all of New Zealand is undertaken through the Resource Management Act 1991 (RMA). The RMA focuses mainly on issues of environmental importance, with little direction on the economic and social issues surrounding mine operation and closure.

The closure and rehabilitation of the mine centered around stabilizing the site works around the mine, and ensuring that water quality was of a high standard, in order to provide for the possible use for recreational and possibly commercial trout fishing (Ingle 2002). The Coeur Gold New Zealand Limited/Viking Mining Company Limited Joint Venture established a peer review panel and the joint task force (also known as the Community Consultation Group or CCG) during the operational phase of the mining project to undertake an analysis of the closure and rehabilitation process. Members of the CCG included regional and district councilors and staff members, professional peer reviewers (a geotechnical engineer, an environmental scientist, and geochemists), environmental groups, local residents of the valley, and Maori traditional owners (Department of Environment and Heritage 2003).

The final rehabilitation has seen the mine site become a wetland and native animal habitat, as well as being used for grazing and recreational purposes (Chapter 20, Figures 20.12 and 20.13). Previously cleared areas surrounding the mine site were revegetated with native species. The rehabilitated mine site now has walking trails, picnic facilities, footbridges and information panels along the walking trail that discuss the history of the Golden Cross Mine.

Despite the successful initial plans for closure of the site, the final concept cost many millions more than had been allowed for in the original closure plan. The mining companies in this case, however, decided that it was wiser to bear the extra rehabilitation costs rather than risk future environmental problems and the potential negative publicity associated with it.

Rother Valley Country Park, United Kingdom (Coal)

Rother Valley Park was planned in the early 1970s, but mining did not commence until the mid-1970s. Sheffield City Council began to determine the feasibility of the park in 1972 after the National Coal Board announced that it intended to establish open cut coal mining in the region. The proposed park covered the local council areas of Rotherham, Sheffield, North and East Derbyshire, and the South Yorkshire county councils (Rotherham Metropolitan Borough Council 2004).

A joint committee of the five county councils oversaw the funding for development and running of the park. The joint committee throughout this period established a community consultation program. In 1977, they produced a development options report (unpublished)

Source: Rotherham Metropolitan Borough Council 2004.

FIGURE 22.5 Rotherham Country Park, after rehabilitation

that allowed for significant community input into the final design of the park. From the outset it was envisaged that the site would be rehabilitated in such a manner that it would become a recreational facility that would attract tourists to the region. By 1978, the final report was published and used as the basis for the final rehabilitation design (Rotherham Metropolitan Borough Council 2004).

The restoration program achieved three main objectives:

1. Create four river-filled main lakes with adjacent open land to provide a wide variety of recreational pursuits, both water and land based (Figure 22.5);
2. Provide several different habitats for the many different plants and animals that inhabit or migrate through Rother Valley; and
3. Create an efficient flood control system to protect areas of housing and industry downstream (Rotherham Metropolitan Borough Council 2004).

Although all five councils initially offered rehabilitation funding, the Rotherham Metropolitan Borough Council continues to be the sole entity responsible for the funding and maintenance of the park (Rotherham Metropolitan Borough Council 2004). Rother Valley incorporates many of the features of world's best practice rehabilitation such as early planning, community involvement, strong local council involvement, a long time frame for the development, and commitment from all of the parties involved in the closure process. Furthermore, Rother Valley Park has now developed strong commercial outcomes through tourism-based activities.

Collinsville Coal Project, North Queensland, Australia (Coal)

The Bowen region in North Queensland of Australia is currently facing a long-term drought, and large volumes of water, even of low quality, are of great benefit (Côte et al. 2006). Mining pit lakes therefore represent significant short-term resources to adjacent operations and furthermore reduce pressure on regional natural water resources over a longer period (McCullough and Lund 2006).

Photo courtesy of Clint McCullough.

FIGURE 22.6 Reference (upper) and treatment (lower) sections of the experimental acidic pit lake, Garrick East

A major use of water in the Collinsville operation in North Queensland is for road dust suppression of haul roads and processing of coal to suitable quality for sale. However, to protect offsite natural surface watercourses, only the use of low-salinity waters are permitted by regulatory authorities, making highly acidic pit lake water unsuitable.

Treatment of Collinsville waters using sewage and municipal green waste (bulk waste issues themselves) has been demonstrated in laboratory- and field-scale trials (McCullough et al. 2006, 2008a). This treatment approach is now being undertaken in a whole-lake experiment on a sectioned-off part of the Garrick East lake, which has a volume of approximately 70 ML (McCullough et al. 2008b) (Figure 22.6).

This approach stimulates naturally occurring sulfate-reducing bacteria to reverse the process that initially generated the acidity. The warm Collinsville climate, typical of many of large-scale mining operations in remote arid Australian areas, may facilitate such "passive remediation" treatments (McCullough et al. 2008a). After 6 months of additions with organic materials, hypolimnion pH has increased, electrical conductivity has decreased markedly, and concentrations of sulfate, iron, aluminum, and heavy metals have also decreased. These treatment results have shown that even highly acidic pit lakes have potential to be inexpensively remediated with low-grade waste organic materials to provide a large supply of water for low-grade end uses.

Laverton, Goldfields, Western Australia

Laverton is a small service town in the arid northeastern Goldfields of Western Australia. Fresh groundwater in the region contains unacceptably high levels of nitrate for potable use, and, being an arid region, there are no other natural water resources nearby. Consequently, local mining pits have been employed as water resources by the state domestic water supply company (Water Corporation). Although never planned for as a beneficial end use, nitrate-contaminated regional groundwater is blended with low-nitrate water in the Wedge pit lake (Figure 22.7) to improve water quality and provide a potable resource for the local community. Residual pit lake water arsenic is easily treated by conventional means. This arrangement removes the alternative of expensive reverse osmosis treatment, which is used in many other Goldfields area towns.

Photo courtesy of Clint McCullough.

FIGURE 22.7 Wedge pit lake is used as a potable water supply by the Goldfield's mining town of Laverton

Nevertheless, potential risk of contamination possibly caused by disposal of wastes, livestock deaths, or native fauna deaths within the unmanaged mining catchment remains high. Although a levee bank has been constructed to protect the pit against flooding from intermittent flows in the nearby creek, it is believed that Wedge pit lake water is fresh on account of large inputs of surface runoff when it rains. The meteoric water forms a low-saline groundwater lens above the regional hypersaline aquifer, as has been observed with other Goldfields pit lakes (Connolly and Hodgkin 2003). Wedge pit lake can store 800–850 ML water when full; however, total annual recharge of both ground- and surface water to the Wedge pit remains unknown. Clearly, exploitation of this resource needs to be carefully managed to ensure that sufficient fresh water is maintained in the pit to prevent more brackish groundwater intrusion. There are currently 1,800 mining pits in Western Australia with more than 150 currently operating below groundwater level (Johnson and Wright 2003). However, given the high rate of previous and future development across arid regions of Western Australia, this water resource remains a likely beneficial end use option, especially for lower-quality water requirements such as industrial use, including process waters of new and current mining operations currently limited by water resource availability (Barger 2006).

SUMMARY OF OTHER PIT LAKE BENEFICIAL END USE DEVELOPMENTS

As has been shown in the previous case studies, many mining companies throughout the world are actively engaged in the creation of postclosure beneficial end uses for pit lakes and there are many more examples of pit lakes that have been utilized for beneficial end uses (Table 22.4). Some of these are planned beneficial uses and others unplanned. However, most pit lake developments are reported predominantly within the nonscientific literature, and, consequently, examples of pit lakes relinquished as public amenities are more inaccessible than for other forms of pit lake research (Walls 2004). Nevertheless, Table 22.4 summarizes some of these other examples that have been achieved, or are currently being developed. Note that some pit lake developments may be achieving, or may be attempting to achieve, more than just the primary end use listed.

CONCLUSIONS

The last few years have seen a plethora of documentation in relation to mine pit closure, and today, both industry and regulators are paying greater attention to the environmental, social, and visual impacts surrounding mine pits. Development of social and environmental end uses from relinquished lakes requires a strong vision and commitment from regulatory authorities and the willingness as well as acceptance of local communities. Opportunities from existing pit lakes or pit lakes in development will require direct support from the mining companies operating the

TABLE 22.4 Examples of existing and potential postmining end uses of pit lakes

Location	Type of Mining Operation	Description	Reference
		Recreation and Tourism	
New Federal States, Germany	Lignite	Numerous lignite surface mining and refinement plants in the territory of the former German Democratic Republic closed and being remediated for recreational use.	Kruger et al. 2002
TXU Mining, Tatum mine, Beckville, Texas, United States	Lignite	Reclaimed pit lake using a pond-in-series design to create five wetland areas. Native grasses and forbs planted and more than 40 acres of hardwood species established. Fish stocking also made for seven local sports-fishing species.	MII 2009a
Stone Mining Company, Pikeville, Kentucky, United States	Coal	Coal slurry impoundment converted into a recreational lake, approximately 21 acres in size. Lake stocked with bluegill, channel catfish, and largemouth bass.	MII 2009b
Collie, Western Australia	Coal	Final open void shaped and contoured to create a recreational lake with central island, Lake Kepwari. Government funding requested to provide boat launching facilities, ablution blocks, parking bays, and barbecue facilities.	Premier Coal 2009; Ashton and Evans 2005
City of Gilbert, Minnesota, United States	Iron ore	Lake-ore-be-gone. Created through natural flooding of three iron ore mines: the Gilbert, the Schley, and the Pettit. City of Gilbert constructed boat landings, docks, and sandy beach swimming area to create a recreational lake.	City of Gilbert 2009
		Wildlife Conservation	
Arch Coal, Mingo Logan mine, Southern West Virginia, United States	Coal	More than 200 acres of ponds and wetlands created on previously mined land. Attracts waterfowl, aquatic species, and other wildlife.	MII 2009c
Alford Field mine, Petersburg, Indiana, United States	Coal	Two separate pits were mined and reclaimed with several peninsulas and coves, as well as some islands. Numerous ponds and small wetlands included in rehabilitated area to maximize the area's potential for use as a fish and wildlife habitat.	MII 2009d
Oxford Mining, Muskingum County, Ohio, United States	Coal	Large pit impoundment and two large wetland impoundments constructed as part of the reclamation plan. Developed year-round and part-year storage areas, heavily vegetated pasture, and areas planted with native trees to enhance specific wildlife habitat.	MII 2009e
Flambeau mine, Wisconsin, United States	Gold/silver	Site contoured back to its approximate original state, along with the construction of wetlands for a wildlife sanctuary. Also established a series of hiking trails and less-intensive walking tracks on the mine site.	Fox et al. 2000
Golden Cross, Waihi, New Zealand	Gold/silver	Surrounding pit lake rehabilitation for revegetation with plant communities. Although primary intention of the lake is wildlife habitat, there is also development occurring for recreational/commercial fishing facilities.	Ingle 2002l Castendyk and Webster-Brown 2006
Universal Mine Slurry Wetland Area, Universal, Illinois, United States	Coal	More than 80 acres of wetland created from a coal wash slurry deposit. Includes 20 acres of permanently impounded water and surrounding wildlife habitat.	OSMRE 2005a
		Aquaculture	
Arch Coal Mingo Logan mine, Thacker Fork, West Virginia, United States	Coal	Cold water from mine used for fish breeding in an arctic char hatchery. Dissolved gases extracted and liquid oxygen added before water is pumped to incubators and trays in the hatchery.	OSMRE 2005b
Premier Coal mine site, Collie, West Australia	Coal	Acid mine lake water treated by a fluidized limestone chip treatment system and then gravity-fed into six polyculture ponds used for silver perch and freshwater crayfish aquaculture research.	Whisson and Evans 2003; Stephens and Ingram 2006

Table continued next page

TABLE 22.4 Examples of existing and potential postmining end uses of pit lakes (continued)

Location	Type of Mining Operation	Description	Reference
		Irrigation	
Klien Kopje Colliery, South Africa	Gypsiferous	Mine water used for irrigation of agricultural crops. Sugar beans and wheat irrigated with three center pivots, on both virgin and rehabilitated land.	Annandale et al. 2001
Freedom mine, Bismarck, North Dakota, United States	Lignite	Mined land reclaimed to croplands for wheat, grasslands for livestock grazing and hay production, and the creation of wetlands for wildlife. Irrigation water sourced from surface water collected in sediment pond.	MII, 2009f
Belle Ayr coal mine, Northeast Wyoming, United States	Coal	Land has been reclaimed for the premine use of livestock grazing and wildlife habitat Success from planting of trees and shrubs enhanced through use of trickle irrigation and water harvesting.	U.S. Forest Service 2007
Oxbow mine, Northwestern Louisiana, United States	Lignite	Permanent ponds incorporated into the postmining topography to increase landowner value and provide water for cattle and wildlife.	MII, 2009g
		Potable Water	
Wedge pit, Laverton, Australia	Gold	Remote arid region mining town water supply. Water abstracted from bores at edge of mined-out pit to minimize liability of pit involvement (see case study).	Australian Labor Party 2004
		Industrial Water	
Garrick East pit, Collinsville	Coal	Remediated (reduced salinity) pit lake water for haul road dust suppression to relieve pressure on competitive regional water resources.	McCullough et al. 2006, 2008a, 2008b
		Chemical Extraction	
Piast/Czeczott, Poland	Coal	Different treatment and disposal schemes described and compared from a technical-economical point of view. Recommends drying of sodium chloride (NaCl) and sale in Poland and/or on the export market as preferred option.	Ericsson and Hallmans 1994
		Education Facilities	
Capel Lakes, southwest Western Australia	Mineral sands	Rehabilitated open-cut mineral sand mine lease in a series of dune lakes and wetlands. Boardwalks and interpretive signs established to provide a regional wetland education center.	Doyle and Davies 1999

project. However, in the case of most of these end uses, the local community will also be a direct beneficiary of such opportunities and is likely to be supportive.

Developing closure plans for a pit lake requires thorough technical, economic, and social evaluation across an extensive list of opportunities that may be available. The final end use choice will inevitably be a balance between all the competing factors and the economic benefits (Ingle 2002). Indeed, one of the greatest challenges of the amenity development planning process is to determine a fair compromise for stakeholder groups and economic and environmental concerns (Lögters and Dworschak 2004). Conducting a comprehensive cost–benefit analysis that includes engineering costs, liability risks, public image, environmental benefits, economic and social benefits, and regulatory acceptance may be the best way to achieve this (Mallet and Mark 1995).

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CHAPTER 23

Slope Stability Considerations

D. Van Zyl

INTRODUCTION

Pit slopes are designed to be stable under operating conditions. These slopes are typically optimized to provide sufficient stability to maintain safe working conditions. After closure, many changes in boundary conditions occur in a pit that influence the geotechnical stability of the pit walls. The pore pressures change as the dewatering or depressurization activities cease. New joint surfaces may get inundated, thereby changing the shear strength of these surfaces. Relaxation of stresses in the pit walls may result in shear strength changes. The overall effect is that rock blocks or smaller particles may be dislodged from the surfaces of the pit walls causing raveling of the slope and potential flattening or progressive failure of the slope. Larger slope failures can also occur over time, resulting in a much larger plan area of the pit and the deposition of rock slope debris, often containing mineralized materials, into the pit, and often into the pit lake. Raveling and larger failures may impact the pit water quality through the exposure of fresh mineralized surfaces that are highly reactive.

This chapter presents a very basic introduction to some aspects of the geotechnical stability of the pit walls after closure. The basics of rock slope stability engineering are described in many texts such as Wyllie and Mah (2004), as well as summaries of case studies such as Hustrulid et al. (2000). The geotechnical stability of open pit mine slopes following closure is influenced by:

- Hydrogeological changes,
- Stress relief or relaxation, and
- Shear strength changes of pit wall materials (including intact materials, fractures, and structures).

HYDROLOGIC CHANGES

Hydrologic changes through the mine life cycle result in ongoing pore pressure changes and therefore potential slope stability and other impacts. The simplified model of an open pit mine shown in Figure 23.1 will be used to describe hydrologic changes through the mine life cycle.

In Figure 23.1 on the left, the total vertical stress (σ) and pore pressure (u) at point A can be calculated as shown. The effective vertical stress (σ') (Lambe and Whitman 1969 and other geotechnical engineering texts) at A is then

$$\sigma' = \sigma - u \qquad \text{(EQ 23.1)}$$

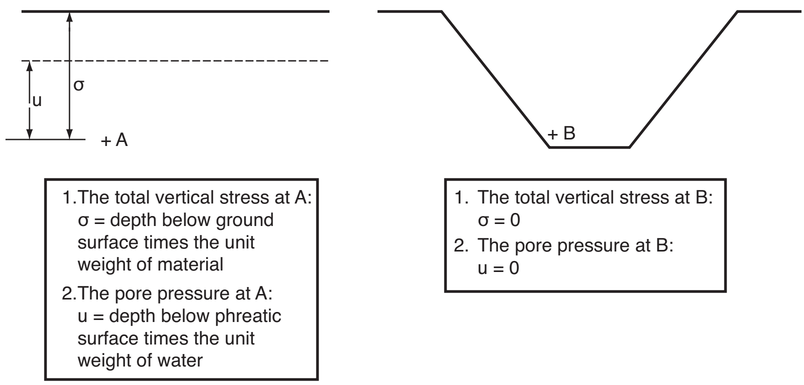

FIGURE 23.1 Premining and mining conditions

The behavior of soil and rock materials is dependent on the effective stress. For example, if the phreatic surface is lowered through dewatering, then the pore pressure at point A will decrease and the vertical effective stress at point A will increase. Such an increase in vertical effective stress can cause compression of the material at point A (especially if the material consists of clay, silty, or sandy materials) and, therefore, overall settlement of the surface. This explains the observations that lowering the phreatic surface can result in surface settlement.

At point A there is also a horizontal stress in the material that depends on the geological history of the site, including rock formation, deposition of sediments, and so forth. The horizontal stress is calculated as a multiple of the vertical effective stress, ranging from about 0.3 to higher than 1; that is, the horizontal effective stress can be less than the vertical effective stress (typically the case for normally consolidated sediments) or it can be higher than the vertical effective stress (typically the case for rock formations that have been subjected to tectonic activity).

When the open pit has reached point B (maybe at a similar depth as the premining point A), the vertical total stress and pore pressure are equal to zero and so is the vertical and horizontal effective stress. However, in the rock directly next to point B, there are still vertical and horizontal stresses in the rock, but the vertical stress is lower because of the removal of the overburden materials and the horizontal stresses are also lower, resulting in "relaxation" of the rock or expansion, however small or significant. This is again a function of the geologic history of the site.

After the pit is filled with water (assume to the original phreatic surface location), the vertical stress at point B is the pore pressure as a result of the depth of water. The stress on the pit wall next to point B is also only the pore pressure. There could still be readjustments as a result of the stress release, and so forth, depending on the time-dependent behavior of the pit wall materials.

Hydrogeological changes at the time of closure can also be quite dramatic. For example, in the case of the Sleeper mine, Nevada, United States (Dowling et al. 2004), the natural rebound of the phreatic surface was short-circuited as the pits were rapidly filled with groundwater and surface water. Pore pressure conditions therefore resulted in seepage into the pit walls until the phreatic surface rebound was complete. Table 23.1 explores a number of pit stability conditions dependent on hydrogeological conditions.

STRESS RELIEF

The stress relief or relaxation described previously can impact the stability of the pit slopes prior to or sometime after pit filling. Such relaxation may increase the size of joints and fractures and

TABLE 23.1 Open pit stability conditions related to hydrogeology

Purpose of Dewatering	Conditions During Mining Operations	Possible Actions and Conditions at Closure	Consequences
Depressurization; this is typically done in low-permeability materials where very little flow is observed into the pit.	Low flow, reduction in pore pressure to stabilize slopes.	Remove pumps and allow pore pressure to increase.	Increased pore pressure that can influence slope stability, even though there is very little flow into the pit as a result of the low-permeability wall rock.
To reduce groundwater table.	Medium to large dewatering rates; water can be treated and discharged.	Stop pumping; large volumes of inflow to the pit.	Increased pore pressure that can decrease slope stability causing the pit to fill with pit wall materials (nonmineralized and mineralized materials).
		Fill pit rapidly from surface water inflows, such as river diversion.	Pit is filled in a short period of time. The phreatic surface outside pit does not recover rapidly and pore pressure conditions can be complicated, e.g., initially flow from the pit results in high pore pressure near the pit rim and lower pore pressure at the dewatering front; over time, a steady-state condition phreatic surface is reached with water flowing into the pit.
To collect water from a concentrated flow zone, such as a fault zone.	Local controls of groundwater flows and pore pressure.	Stop water collection.	Water could collect in bottom of pit, resulting in a shallow pool.

make it possible for blocks of rock or smaller particles to break loose from the pit wall, resulting in raveling of the surface. Although the results of stress relief and other pit wall failure mechanisms can be observed in mined pits (such as shown in Figure 23.2), it does not seem that there is extensive research about the long-term effects on slope stability.

Stacey et al. (2003) find that there can be large zones of extension strain forming near the toe of the slope, either in the slope or the floor of the pit. Excavation of the materials to form the pit as shown in Figure 23.1 results in large stresses at the bottom of the slope, and the "stress relief" mechanisms previously described are related to this failure mechanism. Stacey et al. (2003) find that these extension strains are large enough to fracture intact rock. Surface layers can fail and can lead to larger slope failures. Sudden failures can occur after limited deformation. Although these observations were made in operating mines, it is unclear whether similar behavior will continue to occur in the postclosure period.

SHEAR STRENGTH

The shear strength of the pit wall materials may also change because of reduced normal stress on the materials near the pit walls or as a result of water flow along fractures or geologic structures. The large failure in Figure 23.2 is most probably the result of displacement and a reduction in

FIGURE 23.2 Examples of pit slope failure mechanisms in mined-out pit. The larger failure is probably because of structural controls; the effects of smaller raveling failures can be seen on some of the benches as they are filled with finer materials.

shear strength along geologic structures. Small pit slope failures are not unusual during the pit filling process.

CONCERNS WITH GEOTECHNICAL STABILITY

The major concerns with geotechnical stability are listed as follows:

- Safety concerns for people accessing the pit, whether dry or having a pit lake. Pit wall failures can be of different magnitudes, ranging from dislodging rock particles to larger, structurally controlled failures. Ongoing monitoring and observations can help in identifying some of these failures before they occur; however, they may still pose hazards to individuals entering open pits. Some mine pits are targeted specifically for recreational activities, such as swimming and boating. The Martha mine in New Zealand is an example where the mine owners intend to build a boat ramp to the site to encourage public access and use (Ingle 2002). Ongoing monitoring and visual observations will be required at all these sites to confirm that the slopes remain stable. Unexpected large-volume failures may result in waves on the lake and may be sufficiently large to impact the safe handling of boats.
- Potential impacts on pit lake water quality as pit wall failures take place. Pit wall failures that dump mineralized materials into a lake can influence water quality by releasing mineralized materials into the lake or exposing fresh surfaces of mineralized materials to surface water runoff.
- Enlarging the top area of the pit over time so that it is very difficult to establish the "ultimate" pit limits for access control by embankments or fences. It is difficult to predict the ultimate plan area of a pit if there are ongoing failures of the pit walls. It is expected that most pit walls will not undergo ongoing failure; however, the experience base with the long-term stability of mine pits, especially large pits, is not extensive.

Based on discussions with regulatory agencies, they are presently interested in pit lake water quality and not the stability of the pit walls. This author could not find any publicly available reports from mining companies in Nevada that evaluate the postclosure stability of open pits. It is an area of large uncertainty and will require future research. Such research should include a better understanding of the long-term behavior of jointed rock masses after unloading, analysis of pit walls following closure, ongoing monitoring and observations of open pit slopes to develop a set of case studies, and analysis of the effects of large failures on wave development and dissipation in pit lakes.

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PART VIII: CONCLUSIONS

CHAPTER 24

The State of the Art of Pit Lake Research

L.E. Eary and D.N. Castendyk

INTRODUCTION

The overall purpose of this handbook is to provide a thorough description of the current state of the art of pit lake research, particularly those formed in hard-rock surface mines for the extraction of base and precious metals. This chapter summarizes the state of the art in the following areas:

- Current accomplishments in predicting pit lake water quality
- Data gaps and research needs in developing reliable models of pit lakes
- Recommendations for the mining industry
- Strategies used in pit lake remediation

In addition, this chapter presents a set of guidelines for developing and presenting the results of pit lake predictive models based on consideration of the various types of information and approaches contained in the chapters of this handbook. The primary goal of these guidelines is to develop a template of topics that clearly explain all aspects about how a pit lake model is constructed and applied to a particular mine, and thereby facilitate reviews of pit lake models by mine operators, regulators, technical experts, and researchers, who may have the role of judging the reliability and validity of predictions.

ACCOMPLISHMENTS IN PREDICTING PIT LAKE WATER QUALITY

The information is this section is presented as a series of lists of major findings from the chapters in this handbook for the main topics of hydrology, limnology, geochemistry, microbiology, and management decisions. The order of the items does not infer a level of importance but merely follows the handbook organization.

Hydrology

- Analytical and numerical (e.g., MODFLOW) models exist that can generate fairly reliable predictions of groundwater input volumes and pit lake filling over time after cessation of mine dewatering systems (Chapter 8).
- Pit lake hydrology can be fairly well defined based on the amount of information gathered about subsurface aquifer systems and surface watersheds (Chapter 8).
- Additional complexity and time in generating a numerical model may not necessarily increase overall understanding without validation of the model (Chapter 8).
- Groundwater inputs influence the water chemistry of a pit lake (Chapter 8).

Limnology

- Numerical limnologic programs (e.g., DYRESM and CE-QUAL-W2) are available and have successfully modeled the circulation of existing pit lakes (Chapter 9).
- It is possible to develop numerical predictions of limnology for pit lakes that do not yet exist (Chapter 9).
- The temperature and concentration of total dissolved solids affect the density of lake inputs, which strongly influences the depth and frequency of seasonal turnover (Chapter 9).
- Morphology-based limnology predictions (e.g., relative depth and surface-depth ratios) oversimplify limnologic processes and may generate incorrect depictions of water column stability (Chapters 9 and 21).
- Monimolimnion in meromictic lakes have the ability to receive settling metals, isolate these metals from the surface environment, and precipitate metal-sulfides, thereby improving overall lake water quality (Chapter 21).

Geochemistry

- It is possible to define geochemical reactions and specify kinetic reaction rates between lake water and wall rock minerals. However, developing robust and representative kinetic expressions for the release of solutes on account of wall rock leaching, oxidized groundwater zones, and sulfide minerals remains the single most challenging and generally mine-specific endeavor in developing pit lake predictive models (Chapters 10 and 12).
- Geochemical computer programs (e.g., PHREEQC) are available (Chapter 10) and readily adaptable to representing biogeochemical and geochemical processes important for controlling pit lake water quality.
- Predictions have been generated based on the combination of physical and ecological limnology models with geochemical speciation models that reproduce observed trends in pit lake turnover and geochemical parameters (i.e., Cl^-, NH_4^+, dissolved organic carbon, pH, and Al^{3+}) (Chapter 11).
- Predictions have been generated based on the integration of hydrologic, limnologic, and geochemical models that accurately predict lake acidity and pH over time (Chapter 14).
- Comparisons between predictions and observations show that geochemical predictions can be made with good confidence for major cations and anions (Ca, Mg, Na, K, SO_4, HCO_3, H) and major metals (i.e., Al, Fe, and Mn) in addition to limnologic factors of temperature, salinity, and water density (Chapter 15).
- Predictions of metal and metalloid concentrations (e.g., As, Cd, Cd, Ni, Se, Sb, Pb, and Zn) are problematic because distinct solubility, adsorption, and biologic controls are site specific and difficult to quantify.
- Comparisons of batch test simulations and predicted water quality show good agreement between common ions that do not precipitate from solution (Chapter 15).
- The thermodynamic database in prediction models can be adjusted to match predicted results to batch test observations (Chapter 15).
- Broad geochemical trends have been identified for some pit lakes based on ore deposit geology (Chapter 15).

- Prediction of pH is a priority for models so that maximum metal concentrations can be assumed to be correct. Incorporation of empirical kinetic data is important to establish correct metal concentrations (Chapter 15).
- Detailed studies of existing pit lakes indicate the importance of limnology on controlling vertical redox (oxidation–reduction) conditions (Chapter 15).

Microbiology

- Basic microbial processes such as iron oxidation, iron and sulfate reduction, and photosynthesis for primary production are well established (Chapter 13).
- Knowledge of microbial processes can guide the development of geochemical predictions (Chapter 13).
- Initial geochemical model predictions can indicate which of the microbial processes are likely to be important (Chapter 13).
- Natural bioremediation processes occur in pit lakes (Chapter 18).
- It is possible to support and enhance natural bioremediation processes through the addition of suspended solids, increasing growth substrate and increasing fertilizers (Chapter 18).
- The trophic status of a pit lake needs to be evaluated as a part of the risk assessment process. Morphology and nutrient balances of many pit lakes indicate that they tend to be oligotrophic. Risks in pit lakes with low biological productivity and less than acutely lethal contaminant concentrations may have insignificant impact on populations of resident and migrant species that are associated with the lake (Chapter 16).

Management Decisions

- Rapid filling of pit lake water can help control water chemistry through the addition of alkaline water with low total dissolved solids and will reduce the time between closure and steady-state water elevation (Chapter 17).
- It is best to not flood a pit lake to overflowing immediately to allow researchers the opportunity to understand the chemistry of lake water before discharge begins (Chapter 17).
- Meromictic pit lakes can be manufactured by filling pit lakes with inputs of different density (Chapters 17 and 21).
- It is best to remove highly oxidized waste rock from the pit before flooding to reduce the initial load of dissolved metals in waste water (Chapter 17).
- Mining companies should perform limnologic and geochemical modeling before lake filling begins (Chapter 17).
- Engineering meromictic conditions can improve the quality of surface water discharge by allowing sulfide precipitation in the monimolimnion (Chapter 17).
- When lake filling is complete, companies should evaluate that potential for bioremediation (Chapter 17).
- It is important to recognize that pit wall failures do occur. The stresses that create such failures can be identified (Chapter 23).
- Backfilling open pits can improve visual impacts and avoid pit lake formation; however, this is an expensive approach and may result in initial groundwater quality degradation (Chapter 20).

- Several options exist for the postclosure utilization of pit lakes in fulfillment of sustainable development objectives, including public recreation areas and wildlife conservation areas. These options may require good water quality and stable pit walls (Chapter 22).
- Modifications to the design of the open pit *after* hydrologic, limnologic, and geochemical modeling can affect the accuracy of each of these models and ultimately change the water quality predictions. Major changes to mine plans, such as expansions to the open pit design, may necessitate new modeling.

PRESENT DATA GAPS AND RESEARCH NEEDS

Many of the chapters in this handbook have identified areas where key assumptions had to be made to predict or interpret important processes affecting water quality in pit lakes. These assumptions represent areas where there are data gaps and a need for further research to improve the state of the art in pit lake studies. The key areas presented in the chapters are summarized below for hydrology, limnology, geochemistry, microbiology, and management decisions.

Hydrology

- Given the evidence of global climate change, an accurate prediction of future climate condition (e.g., rainfall and evaporation rates) is needed to improve long-range hydrology predictions (Chapters 3 and 8).
- There is a need for expanded understanding among mining companies, groundwater practitioners, regulatory agencies, nongovernmental organizations, and the public in regard to uncertainty analysis (Chapter 8).
- Modelers should quantify the probable ranges of predicted water chemistries for pit lakes to allow for better judgment of decisions that may be made based on predictions (Chapter 8).
- Longer validation periods are needed in predictions. Modelers should validate models based on two or more observation points rather than one observation point in order to lower uncertainty (Chapter 8).
- Groundwater models need to account for the influence of historic mine tunnels, which may conduct the bulk of groundwater input to a lake.

Limnology

- There remains some uncertainty in defining the extent of mixing within lake water, notably the influence of eddy diffusion across lake layers and high-density surface water inputs occurring along the walls of the pit (Chapter 12). Lateral variations in surface water inputs may invalidate one-dimensional models such as DYRESM.
- Limnologic predictions are limited by inaccuracies in future climate data, variations in the lake water balance, and variations in the temperature and composition of lake inputs over time. More short-range prediction models should be generated and validated to confirm the appropriate approach (Chapter 9).
- It is uncertain how to generate meromictic conditions using methods other than water density manipulation during lake filling (Chapter 21).
- It is difficult to identify how long meromictic conditions may last (Chapter 21).

- Efforts should be made to predict and avoid instantaneous lake turnover that results in the degassing of H_2S, CO_2 and CH_4, and the transportation of metal-rich water to the surface environment (Chapter 21).

Geochemistry

- More effort should be made to test model results against well-constrained water and mass balances for future pit lakes (Chapter 11).
- Comprehensive comparisons of model predictions to high-quality data sets from existing pit lakes are needed to validate models. In particular, there is a need for testing model performance against long-term data sets for systems with well-constrained groundwater inflow, sediment fluxes, and aquatic food web data (Chapter 11).
- Considerable uncertainty remains in defining the specific surface area of pyrite in contact with lake water (Chapter 12).
- More information is needed on changes in geochemical reaction rates as a function of temperature (Chapter 12).
- Comparisons between predictions and observations show it can be very difficult to quantify rates of trace elements released from kinetically controlled processes like wall-rock leaching and metal cycling (Chapter 15).
- Large differences between predictions and observations from experimental simulations of pit lake chemical compositions indicate the need to improve thermodynamic and kinetic data for mineral phases that either sequester or release metals in pit lake environments (Chapter 15).
- More long-term studies of broad trends in pit lakes based on ore deposit type are needed (Chapter 15).
- Broad trends have not been established for solubility controls on trace metals (e.g., Cd, Cu, Pb, and Zn), although surface adsorption reactions roughly approximate concentrations of As and Se (Chapter 15).
- More studies are needed of pit lake sediment mineralogy to constrain lake precipitates (Chapter 15).
- More studies are needed of biological productivity and organic carbon combined with studies of density stratification to understand redox controls (Chapter 15).
- It is necessary to constrain reactions occurring during the initial filling period with observations. The release of metals from oxidized wall rocks and the unsaturated groundwater zone as the water table recovers is an area that warrants research (Chapter 15).

Microbiology

- More knowledge about microbial reaction kinetics and dynamics in the pit lake environment is needed, as well as the ability to use this understanding predicatively (Chapter 13).
- It is difficult to measure microbial processes in situ (Chapter 13).
- Our knowledge of the physical and chemical controls on primary production in pit lakes is limited and additional research is needed in this area (Chapter 13).
- A better understanding of the variation of oxidation rates at the water line, and how to predict wetting and drying cycles of wall rocks resulting from wave action, would be useful for incorporation into pit lake models (Chapter 13).

- Long-term (multi-decade) monitoring studies of pit lakes that have been rehabilitated by stimulating microbial reactions are needed. This is the only way that precise predictive models can be developed (Chapter 18).
- Methods to increase sediment particulate size will aid the removal of metals from the water column (Chapter 18).

Management Decisions

- Uncertainty currently exists on the prediction of pit wall failures. No research is currently being done on this topic (Chapter 23).
- Uncertainty exists regarding the water quality impacts associated with pit lake backfilling (Chapter 20).
- Cost–benefit analyses are required to assess the option of pit backfilling, such as the cost of trucks and of gas use during remediation, and the overall cost of pit lake impacts versus backfilling impacts (Chapter 20).

RECOMMENDATIONS FOR THE MINING INDUSTRY

The insight gained from preparing this handbook has allowed the editors to formulate the following list of recommendations for the mining industry. If implemented, these recommendations will lead to a greater understanding of existing pit lakes and an improved ability to predict pit lake water quality in the future.

- The mining industry and environmental consultants should endorse a general approach to modeling pit lake water quality that is universal and easy for reviewers to follow (i.e., transparent). Suggested guidelines for this approach are presented later in the "Guidelines for Pit Lake Predictive Models" section of this chapter.
- Organizations such as the International Network for Acid Prevention (INAP), the International Mine Water Association (IMWA), and members of the Global Alliance, which includes the U.S. Acid Drainage Technology Initiative (ADTI), the Australian Centre for Minerals Extension and Research (ACMER), the Canadian Mine Environment Neutral Drainage Program (MEND), the Partnership for Acid Drainage Research in Europe (PADRE), and the South African Water Research Commission (WRC), should invest in collecting more observations of pit lake water quality over time from a variety of geologic and climatic settings and disseminating these data in conferences such as the International Conference on Acid Rock Drainage. A large online database will allow modelers to anticipate future water quality based on reported trends, compare observed values to initial predictions, and scrutinize models to identify accuracies and inaccuracies in modeling approaches. In general, the best way to determine the validity of a model's prediction of future pit lake chemistry is a comparison to trends observed in existent pit lakes.
- Companies in coordination with such organizations as INAP, IMWA, and the Global Alliance should consider selecting representative mines that are closing imminently and that have been studied in detail prior to closing (e.g., the Martha mine, New Zealand) for long-term monitoring. These mines should be subject to a 20-year quarterly water quality monitoring program with the results made publicly available. This data set will allow modelers to investigate the accuracy of pit lake prediction methods made prior to closure, and will address some of the current data gaps listed previously.

- Companies, regulators, and stakeholders should jointly define realistic outcomes for pit lake predictions. It is unrealistic to expect models to provide an exact concentration of a particular element in solution 50 years in the future. Rather, companies, regulators, and stakeholders should expect models to provide a range of likely outcomes, and base their risk assessment and management decisions for mitigating risk on the predicted range of outcomes.

CONTEMPORARY REMEDIATION STRATEGIES

Several treatment methods have been applied to existing pit lakes that have improved water quality to some degree. The appropriate treatment strategy for a given lake depends on (1) the specific treatment goal(s) for the lake (e.g., raise pH, reduce arsenic concentrations); (2) climate conditions that dictate the volumes of water added to and removed from the lake, as well as local water resources available for alternative treatment methods; and (3) the time period when treatment activities will be initiated (i.e., prefilling, filling, or postfilling). It is important to recognize that not all of these strategies would be successful at all pit mines; site geology, local climate, the concerns of local citizens, and sustainability objectives (if any) will determine the appropriate remediation strategy for the pit lake.

To save time and expenses, and to increase the likelihood of success, the mining company should extensively model the hydrology, limnology, and geochemistry of the future or existing lake prior to the implementation of any remediation strategy and use these models to test the likely outcome of each proposed treatment strategy. A cost–benefit analysis should also be conducted to evaluate the costs and benefits of a particular strategy against the risks associated with taking no action.

Prefilling Treatment Strategies

Backfilling. By backfilling an open pit, pit lake development can be entirely avoided. Experiences at the Golden Sunlight mine in Montana, United States, and the Golden Cross mine in New Zealand have shown that this is an effective yet expensive option that may be appropriate for some mines and not for others (Chapter 20). Groundwater contamination may result from backfilling, and ongoing groundwater monitoring may be required. Backfilling reduces the visual impact of mining and may be favored by the general public.

Perpetual pumping. In extremely arid environments where the annual volume of water discharging to the pit is minimal, it may be cost-effective to perpetually pump and treat water from the pit as a means of avoiding pit lake development. Chevron is considering this method at the Questa molybdenite mine in New Mexico, United States. This strategy would not be suitable in wetter climates where the volume of groundwater needing to be pumped and subsequently treated would make this cost-prohibitive.

Wall-rock covers or runoff collection. Wall rocks exposed above the lake surface can contribute the majority of mining influenced water (MIW) added to a pit lake. By covering these rocks with a geotextile or nonreactive rock and soil, pit wall runoff may have significantly higher water quality than otherwise. Ideally, this cover would extend from just below the lake surface to the top of the pit catchment area; however, isolated patches that cover the most easily eroded, sulfide-rich material may also prove effective (see Castendyk and Webster-Brown 2006). Alternatively, some portion of pit wall runoff can be collected in surface diversions before it flows into the lake, depending on pit wall geometries. This variation could require collecting and treating runoff indefinitely, if the drainage does not meet surface water quality requirements, adding to annual operating costs.

Excavation of reactive material. Pit lake water quality can be improved by removing acid-generating rocks from the pit prior to lake filling, as demonstrated at the Island Copper pit lake in British Columbia, Canada (see Chapter 17).

Expansion of the littoral zone. The littoral zone is the portion of the lake floor that receives sufficient sunlight to allow for organic productivity (i.e., photosynthesis). Given recent advances in biochemical treatments (see next section), it may be advantageous to engineer a shallow water zone around the perimeter of the lake that will enhance the growth of rooted (i.e., vascular) plants. Existing pit benches may facilitate this design. The potential disadvantage of this approach is that improving productivity may attract more wildlife, possibly increasing their exposure to metal-impacted drainage water.

Filling Treatment Strategies

Density-driven meromixis. Meromictic lakes are permanently stratified lakes with a surface water layer and a bottom water layer that do not mix. Meromictic pit lakes may be advantageous to well-mixed, holomictic pit lakes in that (1) dissolved oxygen is not transported to deep, submerged wall rocks, which reduces the subaqueous production of acid mine drainage by dissolved oxygen; (2) the deep layer is permanently isolated from the environment, provided there is no groundwater discharge; and (3) reducing conditions in the deep layer lead to sulfide precipitation, which removes metals from lake water and produces alkalinity. One technique to engineer meromictic conditions is to flood most of the lake with high-density water, such as seawater, and cap the lake with fresh water. The density contrast between these layers strongly resists vertical mixing. Extensive limnologic modeling is required to implement this method. Moreover, as Cameroon's Lake Nyos demonstrated, the unexpected turnover of a meromictic lake due to seismic events or landslides can have catastrophic consequences because of the sudden release of dissolved gases (i.e., H_2S, CH_4, CO_2, etc.). This method has been successfully applied at the Island Copper pit lake in British Columbia (see Chapters 17 and 21).

Surface water flooding. Rapidly filling a pit with surface water minimizes the contribution of groundwater to the lake and reduces the time until steady-state hydrologic conditions are achieved. The latter may accelerate regulatory approval of site closure, thereby speeding the return of environmental bonds and/or the release of liability. This method is only appropriate if groundwater has a lower quality, or a greater potential to generate MIW than surface water. This may be a short-term solution, as eventually groundwater inputs will exchange with surface water, modifying lake water chemistry. This method was also implemented at the Island Copper pit lake (see Chapter 17).

Postfilling Treatment Strategies

In situ chemical treatment. The addition of a lime slurry to pit lake water can raise lake pH and reduce dissolved metal concentrations via mineral precipitation and adsorption. However, the effects of lime treatment are temporary unless the source material is removed; therefore, lime treatment may become a perpetual expense. A lime treatment plant recently constructed in Britannia, British Columbia, Canada, cost $12 million to construct and has an annual operating cost of $1 million (Chapter 17). It is also possible that lime treatment can transform an acid-drainage problem into a neutral-drainage problem, owing to the desorption of anionic metals at high pH (e.g., arsenic, selenium, and molybdenum). Extensive geochemical modeling should be conducted prior to the implementation of this strategy. Successful lime treatment has been conducted at the Anchor Hill pit lake in South Dakota, United States (Lewis et al. 2003; Park et al. 2006), and also in the South mine pit lake in the Copper Basin of Tennessee, United States (Wyatt et al. 2006). The in-pit treatment system in the South mine pit lake is estimated to cost only 15% of a conventional water treatment plant and has been successful in substantially reducing metal loads to a downgradient stream that receives discharge from pit overflow (Wyatt et al. 2006).

Biochemical treatment. Biochemical treatment of pit lakes has shown considerable promise in recent years. In this method, an organic material (e.g., molasses, methanol, organic waste) and/or fertilizer are added to the lake. The added organic material will typically sink to the bottom of the lake where aerobic microorganisms will decompose it, consuming dissolved oxygen in the process. Once dissolved oxygen is depleted from the bottom layer, anaerobic microorganisms will continue to decompose the organic material, resulting in reducing conditions at the bottom of the lake. In theory, reducing conditions stimulate the growth of sulfate-reducing and iron-reducing bacteria, leading to an increase in pH and Fe^{2+} and H_2S concentrations. Bacteria cultures can be added to the lake to kick-start these reactions, yet these bacteria are thought to be ubiquitous. Eventually, sulfides begin to precipitate, removing undesired metals from the lake water. The addition of fertilizer stimulates the growth of phytoplankton with three notable benefits: (1) photosynthesis slightly raises pH, (2) plankton directly integrate dissolved metals into their body mass, and (3) the surface area of the plankton provides a significant adsorption site for trace metals. Upon their death, plankton settle to the bottom of the lake, thereby removing integrated and adsorbed metals from the surface water. Decomposition of phytoplankton at the bottom of the lake leads to the processes described previously. Meromictic pit lakes provide the optimal environment for these processes as reducing conditions occur in bottom waters. Biochemical treatments have been successful at the Anchor Hill pit lake (Lewis et al. 2003; Park et al. 2006), the Island Copper pit lake (Chapter 17), the Sweetwater pit lake, Wyoming, United States (Paulson and Harrington 2004), and others. Because pit lakes are generally nonproductive (oligotrophic), it is likely that organic material and/or fertilizer will need to be applied in perpetuity. Fertilizer application costs $300,000 per year at Island Copper, which is considerably less than chemical treatment options (see Chapters 17–19).

Structurally imposed meromixis. By suspending circulation barriers from the lake surface, or floating barriers from the bottom of the lake, it is hypothetically possible to inhibit annual circulation and produce meromictic conditions. This method has not yet been applied to a pit lake (see Chapter 21).

Discharge management. Pit lake discharges should be managed to minimize the impacts on downstream and downgradient water resources. To prevent groundwater discharge from a lake, a surface water discharge point can be constructed just below the level of the premining water table such that groundwater will perpetually discharge into the lake. A passive wetland can be constructed at the lake discharge point to enhance trace metal adsorption prior to off-site discharge. Wetlands have been shown to be effective treatment systems at the reclaimed Golden Cross mine in New Zealand and elsewhere. In addition, active treatment methods are often necessary to mitigate direct discharge from small pit lakes, particularly during spring freshet. Research needs to be focused on how to design flexible equipment configurations for active treatment of specific problem trace constituents found in some pit lakes on a site-specific, variable time-frame basis.

GUIDELINES FOR PIT LAKE PREDICTIVE MODELS

Predictive models are useful for evaluating the potential for future impacts to water quantity and quality and are an important part of the regulatory process for permitting new mines, permitting mine expansions, and developing alternatives for mine closure. Pit lake models may range in complexity, depending on their intended use in the permitting process. In addition, individual mines represent unique sets of physical and environmental conditions, potentially requiring unique modeling approaches and data. As a result, there is no one exact modeling methodology that is applicable to all pit lakes. A direct reflection of these factors is evidenced by a large portion of this handbook being devoted to descriptions of various modeling approaches and applications to different pit lakes.

Although different modeling approaches may be applicable to different pit lakes, the variety of possible approaches poses a problem for both mine owners and regulators who have to assess the environmental impacts of pit lakes in the permitting and evaluation process for new mines and expansion projects at existing mines. Thus, in reports on pit lake modeling predictions, it is especially important to provide detailed descriptions of how the model was constructed, how it was applied to make predictions, its major assumptions, use of data, and assessment of the uncertainty of the predictions. This detailed information is critical to allow mining companies, regulators, and other reviewers of pit lake models to evaluate the validity of model predictions.

The following guidelines developed from the chapters in this handbook are suggested as necessary components of a pit lake modeling study and report:

- Purpose
- Conceptual model
- Modeling approach data sources and usage, including what data are measured and what data are estimated
- Assessment of reliability of predictions

These guidelines are not intended to be a cookbook for constructing a pit lake model. Instead, they are intended to be guiding principles for developing, applying, and explaining pit lake models, so that the validity of predictions of water quality and quantity can be assessed with confidence. Although the science of pit lake predictive modeling is continually evolving, the careful and detailed description of the models combined with cautious interpretation of results should allow mines and regulators to meet their environmental objectives and minimize liabilities and risks.

Purpose

In most cases, the purpose of developing and applying a pit lake predictive model will be obvious. However, it is still important to clearly state the intended purpose of the model results because the level of detail of the model should be sufficient to make management and regulatory decisions based on the predictions. Typical purposes of pit lake models may include the following:

- Inputs to environmental impact statements for permitting of new mines
- Inputs to environmental impact statements for expansions of existing mines
- Identification of alternatives for mine closure
- Analysis of chemical, physical, and biological processes occurring at existing pit lakes for either research or treatment options
- Assessment of long-term sustainability of water quality in areas where pit lakes may comprise a valuable water resource
- Combinations of the above

For each of these generalized purposes, the primary reason to develop a predictive model is to identify where situations, such as consumption of scarce water resources or generation of poor surface water quality, can be expected to occur so that management decisions on preventive and mitigation measures can be planned prior to their occurrence in the field. This use of predictive models is consistent with the guiding principles for assessing the potentials for acid rock drainage and metal leaching at hard-rock mines through the linked studies of prediction, prevention, mitigation, and contingency for the purpose of developing environmentally sound and economically viable management practices (McLemore 2008; Figueroa and Gusek 2009; MEND 2005; Price and Errington 1998; EPA 1994, 1999).

Conceptual Model

Every pit lake numerical model should be based on a conceptual model. The conceptual model should provide a description of all of the processes that add to, remove from, and redistribute chemical solutes to the pit lake. In situations where detailed limnology models are needed, the energy balance should also be a part of the conceptual model. Depending on the complexity of the model and its intended purpose, the processes affecting water and chemical balance in a pit lake may include

- Hydrology (water balance)
  - Average, dynamics, and variability of precipitation and evaporation rates, and the predicted modification of these rates over time as a function of climate change
  - Groundwater inflow and outflow
  - Surface water runoff from the pit lake catchment and wall rocks
- Geochemical and biological (chemical balance)
  - The compositions, quantity, and distribution of wall-rock minerals
  - Wall-rock oxidation and leaching
  - Inorganic mineral precipitation in the water column and sediments
  - Inorganic adsorption and exchange of metals on mineral surfaces
  - Exchange of $O_2(g)$ and $CO_2(g)$ with the atmosphere
  - Nutrient and metal uptake in the water column
  - Organic carbon production and consumption
  - Redox processes in sediments (e.g., sulfate reduction and metal sulfide precipitation)
- Limnological (energy balance)
  - Probability of seasonal and/or annual turnover of the entire water column
  - Probability of meromictic conditions
  - The depth of the hypolimnion/monimolimnion boundary, and the volume and composition of the monimolimnion layer
  - Probability for whole-lake turnover in a meromictic lake

Chapters in this handbook by Castendyk (Chapter 6), Boehrer and Schultz (Chapter 5), Niccoli (Chapters 4 and 8), and Shafer and Eary (Chapter 10) provide examples of conceptualizations of the important processes that affect pit lake hydrology, limnology, and water quality. These examples can form the starting point for developing conceptual models for most types of pit lakes modeling situations.

Modeling Approach and Data Sources

Any report that presents the results of a pit lake predictive model should contain a detailed description of the numerical approaches used to represent each of the processes identified in the conceptual model. The sources of the data used to parameterize the equations incorporated into the modeling approaches also need to be described in detail. For most pit lake predictive models, these descriptions will likely cover all or parts of the following areas:

- Time steps
- Physical properties of the mine
- Water balance
- Chemical balance
- Limnology
- Uncertainties/sensitivity analysis
- Computational method

Time steps. The goal of most pit lake predictive models is to forecast future conditions. Pit lakes are dynamic systems, especially during the initial period of hydrologic rebound when the pit fills with water. Even under so-called steady-state conditions, changing climate conditions will perpetually modify the pit lake water balance. Hence, most models use mathematical expressions to calculate water and chemical mass balances over discrete time intervals that step forward in time from a starting point. The length of the time step used in these calculations should be consistent with the purpose of the model and level of detail of its input data. For example, a model designed to predict the long-term (decades to hundreds of years) water quality for a proposed mine may employ a one-month time or possibly a one-year step because the primary purpose of the model is an indication of chronic water quality indicators, such as pH, redox, total dissolved solids, and metal concentrations, to assist in decisions about mine closure. In contrast, a model designed to represent episodic chemical processes in an existing mine, such as the effects of stratification and turnover of the water column on biological and chemical processes, assessments of storm events on the hydrology, or assessments of closure alternatives, may need to use a time step of one day or shorter.

Physical properties of the mine. The physical and morphological properties of the pit and sources of data for those properties should be documented because they are used in many of the numerical representations of the water balance, chemical balance, and limnology. These properties may include but are not limited to pit volume as a function of elevation, pit lake surface area as a function of elevation or water volume, fracture density and grain size for pit walls, and level of exposure to sun and wind.

Water balance. The water balance for a pit lake will be defined by the hydrologic components of the conceptual model, such as whether the pit is a terminal, flow-through, or combination of terminal and flow-through system depending on season. The data and equations for the component rates of water inflow and outflow that make up the water balance should be described or referenced. For example, rates of groundwater inflow and outflow may be obtained from a site-wide hydrologic model or from an analytical model based on the local hydrologic domain. Rates of rainfall and evaporation may be obtained from nearby meteorological stations or represented by a synthetic record. Rates of surface capture may be obtained from a local runoff model.

Chemical balance. The chemical balance for a pit lake will be defined by the conceptual model that describes each of the processes that add chemical solutes to the pit lake or remove them. The equations and data used to represent these processes should be described in detail. For example, the chemical composition of a pit lake, especially during the initial period of infilling, is a function of the rates of chemical solute flow to the lake. Solutes enter the lake primarily through groundwater inflow and leaching of wall rocks and backfill material (if present). Solute influxes are functions of flow rates. Hence, they are directly dependent on predicted rates of water inflow and outflow for each of the different hydrologic sources that make up the water balance of the pit lake. A common modeling approach is to assign each hydrologic source of chemical composition based on available site-specific data and then mixing these sources in proportions equal to their rates of inflow to the pit lake minus their outflows, both of which may change over time as the pit lake fills. The source of the data used to represent the chemical compositions of the hydrologic inflows, such as groundwater, rainwater, or wall-rock runoff, should be described in detail. The processes of wall-rock leaching and solute release during the initial stages of groundwater inflow to the pit may often require a detailed explanation because they may be represented by a number of different calculation approaches that vary considerably in complexity. The mixing calculations yield a bulk composition that should be equilibrated using a chemical and/or biological model, according to a set of chemical equilibrium or kinetic processes expected to be applicable to the pit lake. These processes should also be described in detail and may include geochemical reactions (e.g., mineral solubility, mineral oxidation and leaching rates, redox, gas-phase equilibria), biological factors (e.g., metal, nutrient, and carbon cycling), and limnological effects (e.g., stratification, ice cover, overturn, and mixing of the water column).

Limnology. A number of the chapters in this handbook have pointed out the importance of limnology for affecting water quality in pit lakes (e.g., Boehrer and Schultz, Chapter 5; Castendyk, Chapter 9; Pelletier et al., Chapter 17; Schultze and Boehrer, Chapter 21). The approach used to represent the effects of limnological processes on water quality in the pit lake predictive model should be described. This description may range in detail, depending on the purpose of the model. For example, it may be necessary to use a detailed numerical limnology model to explore the seasonal effects of biological processes on metal and nutrient cycling. In contrast, the long-term effects of stratification on water quality may be examined sufficiently by considering the effects of alternate redox conditions on water quality reflective of the different lake layers.

Uncertainties/sensitivity analysis. An analysis of the uncertainties in predictions and sensitivity to inputs should be provided in reports on pit lake models where possible. The discussion about uncertainties should include descriptions of the major assumptions used to construct the pit lake model based on the conceptual model. Uncertainties and sensitivity analyses are areas that are often overlooked but can be important for developing confidence in the results or identifying areas where improved input data to the model are needed. They may be particularly important to reviewers of pit lake predictive models predictions, who may have to make judgments about the validity of the modeling approach. Probabilistic approaches where stochastic formulations are used to represent levels of uncertainties may often be the most practical method for conveying the importance of key data inputs on model results. Castendyk and Webster-Brown (2007a, 2007b) provide an example approach for analyzing sensitivities and uncertainties for a pit lake predictive model.

Computational method. A general description of the computational method used to run the pit lake model should be described. This description should include the model configuration, such as a single model or linked models, the submodels used (e.g., geochemical and limnological models), and methods of handling data inputs and outputs (e.g., spreadsheets or text files). It should also include mention of how the numerical calculations were verified through error checking of water and chemical mass balances, chemical charge balances, and numerical dispersion. The reasonableness of the results should also be considered in terms of an inspection of trends in predictions for obvious errors in conceptualizations, equations, and unit conversions.

Assessment of the Validity of Predictions

The validity of predictions of future pit lake quality may be difficult to quantify. One of the best ways to check the validity of a model is to compare its water quality predictions to trends in compositions observed in existing pit lakes. Compilations of water quality and interpretation for existing pit lakes that can be used for comparison to modeling results can be found in works by Davis and Eary (1996), Eary (1999), Miller et al. (1996), and Price et al. (1995). In addition, Shevenell et al. (1999) provides a compilation of chemistry for different types of deposits, which can provide an excellent means to determine whether a pit lake prediction is consistent with the known trends in geoenvironmental characteristics unique to different ore deposit types.

The following general guidelines can be followed in examining the predictions made with a model that can provide confidence that the model is producing reasonable results.

Predominantly acidic pit lake environments. Pit lakes situated in rocks with high potential for acid generation in the wall rocks, rubble, and backfill in combination with acidic to neutral groundwater inflows with low alkalinity can be expected to evolve to acidic compositions. For example, silicic, massive sulfide, and porphyry copper deposits commonly generate acidic pit lakes. Under these conditions, most metals (e.g., Al, Cd, Cu, Fe, Hg, Mn, Pb, Zn), metalloids (As, Sb, and Se), and sulfate concentrations can also be expected to be elevated. Total dissolved solids concentrations may also become elevated, depending on the hydrologic conditions.

Predominantly alkaline pit lake environments. Pit lakes situated in rocks with low or no potential for acid generation in the wall rocks, rubble, and backfill in combination with neutral to alkaline groundwater inflows can be expected to evolve from neutral to alkaline compositions. Scarn and limestone replacement deposits are examples, depending on the amount of silicification and carbonate content. Under these conditions, most trace metal concentrations (e.g., Al, Cd, Cu, Fe, Hg, Mn, Pb, Zn) can be expected to be low, but metalloids (e.g., As and Se) and sulfate concentrations may be elevated or become elevated over time because of evapoconcentration over time in terminal pit lakes in arid climates.

Indefinite acidic-alkaline pit lake environments. In contrast to the previous two situations, it is more difficult to assess the validity of model predictions for pit lakes located in rocks with a mixture of acidic and alkaline properties (such as some skarn deposits, ultramafic host rocks, and vein replacement deposits with high amounts of carbonate replacement minerals). The acid–base character of these lakes will be dependent on the chemical balance between acid generation and neutralization in the wall rocks and the influx of alkalinity from groundwater. An additional complicating factor is that wall–rock reactions and groundwater inflows may change over time as the pit fills, changing the acid–base balance. Stratification and turnover of the water column may cause additional seasonal changes in the water quality. Assessments of the validity of model predictions for this class of pit lakes may be based primarily on professional judgment about the conceptualizations and approaches used in the model, uncertainty analyses, and comparison with other pit lakes with similar properties.

CONCLUSIONS

Our general understanding of pit lakes has improved over the years due to the need to assess the long-term impacts of mining and mine closure on the environment. Mine pit lakes are dynamic systems that are artificially created, allowing the chance to observe how they evolve over time. However, most pit lakes are young or not yet in existence such that there remain substantial areas of uncertainty in their environmental behavior in the areas described previously. However, these authors are confident that continued research on water quality, hydrologic, and limnologic dynamics of pit lakes will lead to increasingly effective approaches to make sure that pit lakes become water resources wherever feasible rather than liabilities.

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NOTE: *f.* indicates figure; *t.* indicates table.

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