Second Australian Acid Mine Drainage Workshop (Eds. N.J. Grundon and L.C. Bell), pp. 127–145.

WETLANDS TO TREAT AMD – FACTS AND FALLACIES

D.R. Jones and B.M. Chapman
CSIRO Minesite Rehabilitation Research Program,
Division of Coal and Energy Technology, P.O. Box 136, North Ryde, NSW 2113.

ABSTRACT

Wetlands have many attractions for the treatment of mine drainage since they have the potential to provide an aesthetically attractive low cost, low maintenance, and sustainable alternative to expensive chemical treatment plants. The need for self sustaining systems becomes much more critical following site decommissioning since not only will there be few, if any, staff on site but there will no longer be a direct cash flow to support high operating costs. The objective of this paper is to synthesise what is known about the fundamental physical, chemical, and biological processes in wetlands to provide the basis for designing effective and sustainable systems for the treatment of AMD. The roles, functions, and limitations of different components (for example, anoxic limestone drains, surface and sub-surface flow wetlands, ponds, and riffle zones) of a wetland treatment system are highlighted and discussed.

1.0 INTRODUCTION

Wetlands are a complex assemblage of abiotic chemical processes, aerobic and anaerobic bacteria, emergent rooted plants, floating plants, epiphytes on the surfaces of plants, and algae co-existing in a functional relationship. They exist wherever the underlying soil is permanently saturated. Constructed wetlands can range from a marsh or pond created in a natural setting where one did not exist permanently before to formed structures involving earth moving and erection of permeable bunds and impermeable containment barriers.

Although research on wetlands has been conducted over the past 50 years, it is only over the past decade that major fundamental advances have been made in designs for the treatment of acid mine drainage (AMD). This has been the result of multidisciplinary collaboration between hydrologists, chemists, and microbiologists. The primary impetus for much of this work has been provided by the need to find economic solutions for the treatment of contaminated water draining from abandoned and operating coal and base metal mines. These waters are often acidic and contain high concentrations of metals (Chapman et al. 1983; Wildeman and Laudon 1989). There are several reasons why wetlands are perceived to be attractive options for treating mine water:

(1) They are potentially low maintenance with low requirements for energy and material input compared with chemical treatment plants;

(2) They incorporate a range of physical, chemical and biological processes which can reduce metal concentrations to very low levels;

(3) They can be aesthetically attractive with consequent 'green' appeal; and

(4) There is the potential for recovery of metal values from the wetland substrate.

2.0 PHYSICAL CONSTRAINTS ON WETLANDS

The most important considerations governing the selection and location of a site for a constructed wetland are:

(1) The availability of enough land with a suitable topography to provide the hydraulic head needed to maintain passive flow through the system;

(2) The absence of large water inflows during storm events;

(3) Sufficient year-round supply of water to ensure that the wetland remains in a permanently saturated condition; and

(4) Potential for impact on groundwater quality.

A satisfactory hydrological regime is a critical pre-requisite for the successful implementation of a sustainable wetland. This aspect must be addressed early in the design phase to determine if, on this ground alone, a wetland treatment system will be viable.

The drainage lines of natural catchments are prime candidates for the location of wetlands since the requirements for earthmoving works are minimised. However, an analysis of the catchment runoff yield should be done to ensure that the wetland is not likely to be swept away by a storm event. If the risk of this is high then a diversion structure, or a flow equalisation pond, must be built. Further analysis of the distribution of rain through the year is needed to determine if a supplementary water supply (such as a dam) will be required to maintain flow through the system. This is likely to be the case for a mine located in a tropical monsoonal climate, or a semi-arid region where evaporation exceeds rainfall for most of the year. In the case of a mine which is still operating, this water could be supplied by dewatering bores. If there is potential for contamination of a groundwater resource by downward percolation of partially treated AMD, then the bottoms of the wetland cells and interconnecting flow channels should be sealed with plastic or clay liners.

3.0 PERFORMANCE CRITERIA FOR WETLANDS

The maximum concentrations of residual metals permitted in treated AMD will be governed by the locally applicable water quality criteria for discharge to surface and/or groundwaters. These criteria may be based on end-of-pipe concentrations, or concentrations in the receiving waterway downstream of a specified mixing and dilution zone. The target discharge criteria must be specified before a wetland can be designed since they will impact critically on not only the size of the system, but also on the range of different types of treatment cells that may need to be incorporated. The ANZECC water quality criteria for irrigation, stock watering, and protection of ecosystem health are most likely to be used to specify performance targets in Australia.

4.0 PHYSICAL, CHEMICAL AND MICROBIOLOGICAL PROCESSES IN WETLANDS

Since the oxidation of iron disulfide minerals (pyrite and marcasite) is the primary source of the acidity, dissolved forms of iron (Fe^{2+} and Fe^{3+}, ferrous and ferric ions respectively) are usually a major component of AMD. The composition of the parent mineral assemblage determines the levels of other metals in AMD. In addition to Fe^{2+} and Fe^{3+}, and sulfuric acid, high concentrations of potentially toxic ions such as Al^{3+}, Cu^{2+}, Cd^{2+}, Zn^{2+}, Pb^{2+}, and AsO_4^{3-} may also be present (Table 1). The data in Table 1 show that the concentrations, and relative levels of metals vary widely from one minesite to another in Australia. AMD from coal mines typically contains much lower concentrations of toxic heavy metals than base metal or gold mines. In this context it is important to note that most of the 'success stories' reported for the use of full scale wetlands have been for systems treating such low strength and relatively 'benign' AMD.

Table 1. Composition of AMD from different mines in Australia.

Analyte	Abandoned Mine (NSW)	Gold Mine (North Aust.)	Mt Morgan (Qld)	Coal Mine (NSW)	Coal Mine (Qld)
pH	2.6	3.1	3.72	3.1	3.14
Conductivity ($\mu S\ cm^{-1}$)	2000	6000	13600	ND[A]	8644
Sodium ($mg\ L^{-1}$)	1.61	229	213	31.7	1168
Potassium ($mg\ L^{-1}$)	0.7	61.4	ND	ND	22
Magnesium ($mg\ L^{-1}$)	25.6	700	1700	ND	36.7
Calcium ($mg\ L^{-1}$)	12.4	268	437	ND	526
Sulfate ($mg\ L^{-1}$)	1160	5880	16100	1340	3700
Iron ($mg\ L^{-1}$)	212	20.0	1460	12.6	12.1
Aluminium ($mg\ L^{-1}$)	34.6	179	803	18.1	10.3
Manganese ($mg\ L^{-1}$)	1.98	55.6	131	3.57	7.9
Copper ($mg\ L^{-1}$)	8.89	6.62	106	0.06	< 0.05
Zinc ($mg\ L^{-1}$)	150	187	45	0.65	< 0.05
Cadmium ($mg\ L^{-1}$)	0.78	0.73	0.36	< 0.02	< 0.02
Lead ($mg\ L^{-1}$)	0.62	0.67	< 0.02	< 0.02	< 0.02
Arsenic ($mg\ L^{-1}$)	4.72	0.13	< 0.05	< 0.005	< 0.05

The composition of the drainage is the single most important factor which will determine the size and sustainability of a wetland treatment system. To appreciate why this is so, it is necessary to consider the nature of the chemical and biological processes which remove acidity and metals from AMD.

The processes that can reduce the concentrations of metals in wetlands are summarised in Table 2. It should be noted that, although the physical processes of dilution and dispersion act to reduce the concentration of a pollutant downstream from a source, they do not *per se* decrease the total load or flux of the metal being transported.

Since Fe^{2+}, Fe^{3+}, and Al^{3+} ions are the most abundant non-alkali and alkaline earth cations in AMD, the chemical reactions of these species are crucial to the initial fate of the metals in AMD. The Fe^{2+} oxidises to Fe^{3+} (the rate of which depends on the prevailing pH and the extent of bacterial activity), which subsequently hydrolyses and precipitates. The rate of oxidation of Fe^{2+} increases exponentially above pH 5 in oxygenated water.

Depending on the flow regime, and rate of pH increase resulting from neutralisation of the acidity, 'ferric hydroxide' can be deposited by concretionary growth on an existing coating of the material adhering to the bed of the wetland, or precipitated *via* rapid nucleation. In the latter case it forms a low density floc which must be trapped in a pond or a dense reed bed. The aluminium ion, Al^{3+}, behaves similarly but precipitation occurs at a higher pH (see Fig. 1).

Table 2. Processes in wetlands that reduce metal concentrations and acidity.

Process	Nature	Controlling Variables
Dilution	Physical	Volume of Water
Dispersion	Physical	Flow velocity, channel geometry and roughness.
Oxidation	Chemical Microbiological	Concentrations of oxygen and organic carbon
Precipitation/co-precipitation	Chemical	Concentration of components, pH, redox potential
Adsorption on precipitates	Chemical	Amount of precipitate, concentrations of major cations and anions, pH
Adsorption/ion exchange on suspended and bed sediments	Physical Chemical	Concentration of suspended sediment, particle size and mineralogy, pH, concentrations of major cations and anions
Sulfate reduction	Microbiological	pH, organic carbon, concentrations of oxygen and sulfate
Uptake by biofilms, algae, and aquatic macrophytes	Chemical Biological	Density of plants, temperature, light intensity, availability of nutrients

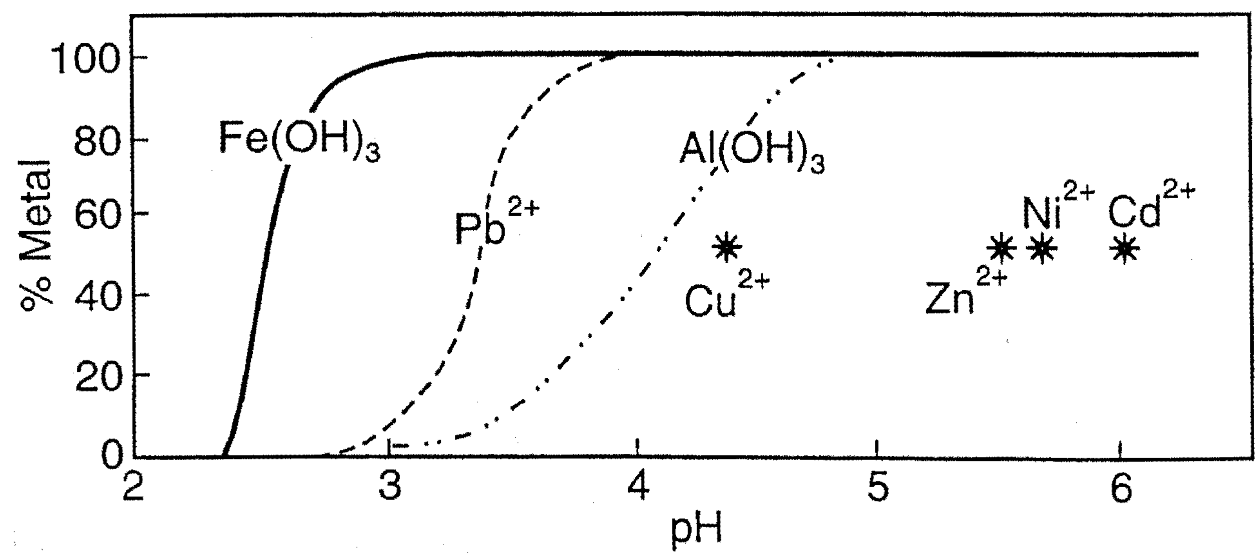


Fig. 1. Schematic showing precipitation of $Fe(OH)_3$ and $Al(OH)_3$, and pH_{50} values for adsorption of Pb^{2+}, Cu^{2+}, Zn^{2+}, Ni^{2+} and Cd^{2+} on $Fe(OH)_3$.

The chemical equations for the oxidation and subsequent hydrolysis and precipitation of Fe^{3+}, and for the precipitation Al^{3+} are depicted in Eqn. [1] to Eqn. [3]. The precipitation of the hydroxides generates acidity. This incipient source of acid must be taken into consideration when calculating the total acid load to be treated by a wetland. A pH value of at least 6 should be achieved prior to moderate strength AMD entering a surface flow wetland. If the pH is less than 6 there will be insufficient alkalinity present to prevent the pH falling substantially as a result of the oxidation and hydrolysis of Fe^{2+}.

$$Fe^{2+} + H^{+} + \frac{1}{4}O_2 \rightarrow Fe^{3+} + \frac{1}{2}H_2O \quad [1]$$

$$Fe^{3+} + 3H_2O \rightarrow Fe(OH)_3 + 3H^{+} \quad [2]$$

$$Al^{3+} + 3H_2O \rightarrow Al(OH)_3 + 3H^{+} \quad [3]$$

Under oxidising conditions, and for the pH range 4 to 7, the dissolved concentrations of trace levels of Cd, Cu, Ni, Pb, and Zn in mine water are likely to be limited by adsorption on precipitated ferric and aluminium hydroxides. Indeed the high scavenging capacity of these hydroxide forms the basis of most industrial chemical treatment processes for the removal of heavy metals. Adsorption occurs over a typically fairly narrow range of one to two pH units and is manifested by a large decrease in the concentration of metal in solution as the pH increases. The pH range over which this decrease in concentration occurs is called the pH-adsorption edge, and the pH at which the dissolved metal ion concentration declines to 50% of its initial value is termed the pH_{50} value. The pH_{50} value for a given metal is a complex function of the nature of the sorbing phase (the sorbent), the ratio of the concentrations of sorbate (the metal ion) to sorbent, and the concentrations of major cations, anions, and dissolved organic matter. For crystalline goethite (an iron oxide which is typically found in AMD systems) the pH_{50} values for Pb, Cu, Zn, Ni and Cd are 3.2, 4.3, 5.5, 5.7 and 6.0 respectively for initial metal concentrations of 20 $\mu g\ L^{-1}$ (Fig. 1). As the initial metal concentration to iron oxide ratio increases the pH_{50} value also increases. Thus a higher pH will need to be achieved before 50% of the metal is removed from solution. This fact has important implications for AMD which contains high concentrations of heavy metals relative to the initial concentration of dissolved iron.

The pH of AMD should be raised to at least pH 6 prior to the oxidation of Fe^{2+} to Fe^{3+} to make the best use of the metal adsorbing capacity of the ferric hydroxide precipitate.

For pH values greater than 7, the solubilities of metal cations present at trace levels can also be limited by the precipitation of hydroxide/oxide and carbonate compounds. However, it should be noted that the solubilities of amphoteric metals such as Zn and Pb decline to a minimum with increasing pH, but then start to increase again at higher pH values as a result of the formation of more soluble negatively-charged hydroxy complexes. Thus too high a pH can be detrimental for the removal of these metals.

Metals can also be removed from solution by adsorption on the surfaces of biofilms, on algae, or by epiphytes growing on the surfaces of aquatic plants. This route may be very significant if there is extensive in-stream vegetation.

Under reducing conditions, in the presence of sulfide ion, the solubilities of Cu^{2+}, Pb^{2+}, Zn^{2+}, Ni^{2+}, and Cd^{2+} will be limited by the precipitation of the respective insoluble metallic sulfide compounds or by co-precipitation with ferrous sulfide. The sulfide salts of most heavy metals are much less soluble than their hydroxide and carbonate counterparts within the normal pH range for soil environments (Patterson 1985). This is particularly important for Cd^{2+} and Ni^{2+} where the pH would have to be raised above 10 in order for discharge quality criteria to be satisfied by the precipitation of the metal hydroxides.

Recent research has highlighted the critical importance of facilitating reduction of sulfate in wetlands in order to ensure indefinite operation at high levels of efficiency (Hedin et al. 1989; Brierley 1990; McIntire et al. 1990; Dvorak et al. 1992; Eger 1992; Hammack and Edenborn 1992; Wildeman et al. 1992). Sulfate reduction rates have been measured in marine and freshwater sediments and have ranged from 0.038 $\mu g\ cm^{-3}\ d^{-1}$ to 290 $\mu g\ cm^{-3}\ d^{-1}$. Rates can vary over several orders of magnitude across a pond or lake owing to heterogeneity in redox status and the amounts of organic matter present in the sediments. Fine-grained sediments are likely to exhibit higher rates owing to reduced penetration of oxygenated water from the overlying water column). Bacterial sulfate reduction rates have been measured in a constructed wetland containing mushroom compost as a carbon source and were found to range from 0.19 $\mu g\ cm^{-3}\ d^{-1}$ to 58 $\mu g\ cm^{-3}\ d^{-1}$ (McIntire et al. 1990).

Sulfate reducing bacteria represented by the genera *Desulphovibrio* and *Desulphatomaculum* are obligate anaerobes that require oxygen concentrations to be less than 0.16 mg L^{-1}. Their pH optimum lies between 5 and 9, and, with acclimation, they can tolerate temperatures from 0 °C to 50 °C (Gloyna and Espino 1969; Postgate 1979). Since the lower concentration limit of sulfate to support their activity is approximately 30 mg L^{-1}, sulfate limitation will not be an issue for the treatment of even very dilute AMD. The reduction of sulfate is coupled to the oxidation of organic compounds to bicarbonate:

$$SO_4^{2-} + 2\ CH_2O + 2H^+ \rightarrow H_2S + 2H_2O + 2CO_2 \qquad \text{(pH < 7.0)}$$

$$SO_4^{2-} + 2CH_2O + OH^- \rightarrow HS^- + H_2O + 2HCO_3^- \qquad \text{(pH > 7.0)}$$

This alkalinity provides additional neutralising capacity in the system and will help to buffer it against transient shock loadings of increased acidity. There is generally no shortage of sulfate in drainage from minesites. However, these waters are generally deficient both in nutrients (P and N) and in organic carbon that is the 'fuel' for sulfate reducing bacteria. Consequently, it has been found that readily degradable organic matter such as compost needs to be added to the gravel substratum of a wetland in order to initiate the formation of an anoxic zone (McIntire et al. 1990; Hedin and Nairn 1992).

Sustainable removal and retention of sulfate in wetlands depends on the maintenance of anaerobic, strongly reducing conditions. Wetlands constructed for the removal of metals as sulfides must be designed to maintain these conditions and avoid exposure of the sediments to oxidising conditions.

Manganese (Mn^{2+}) has proved to be one of the most difficult metals to remove from mine drainage. It does not readily form sulfide minerals and will not precipitate as a carbonate or hydroxide until relatively high pH values are attained. It has been found that the most effective removal occurs above pH 7 to 8 when a combination of abiotic and biotic processes are operative (Chapman et al. 1988). The Mn^{2+} is oxidised and precipitated as a Mn^{4+} or a mixed Mn^{3+}/Mn^{4+} hydrous oxide. This oxidation can be promoted by bacteria (Batal et al. 1989). However, once a thin film of Mn-oxide has been deposited, an abiotic autocatalytic reaction will proceed readily if the pH is above 8.

In summary, pH and the presence or absence of oxygen are the key parameters governing the removal of metals and acidity from AMD. The pH must be higher than 6 in order for there to be effective treatment by aerobic or anaerobic processes. If sulfate reduction is a major component of the treatment process then the wetland must not be allowed to dry out or else the sulfides will re-oxidise to produce acid, and remobilise the initially trapped metals.

5.0 OPERATIONAL COMPONENTS OF WETLAND TREATMENT SYSTEMS

A fully functional and self-contained constructed wetland treatment system must, of necessity, contain a number of process units in addition to a series of linked ponds containing plants. These include pre-neutralisation systems (active and passive chemical), aeration zones, at least one of four different types of 'wetland' cell designs, and algal-filters. The functional role of each of these components, and the way in which they should be incorporated into a wetland treatment system is explained below.

5.1 Pre-Neutralisation of AMD

As stated above, pre-neutralisation of AMD is the key to efficient removal of metals by aerobic or anaerobic wetland treatment cells. Ideally the pH should be raised to 6 before the effluent contacts the more sensitive biological components of the wetland proper.

5.1.1 Active chemical

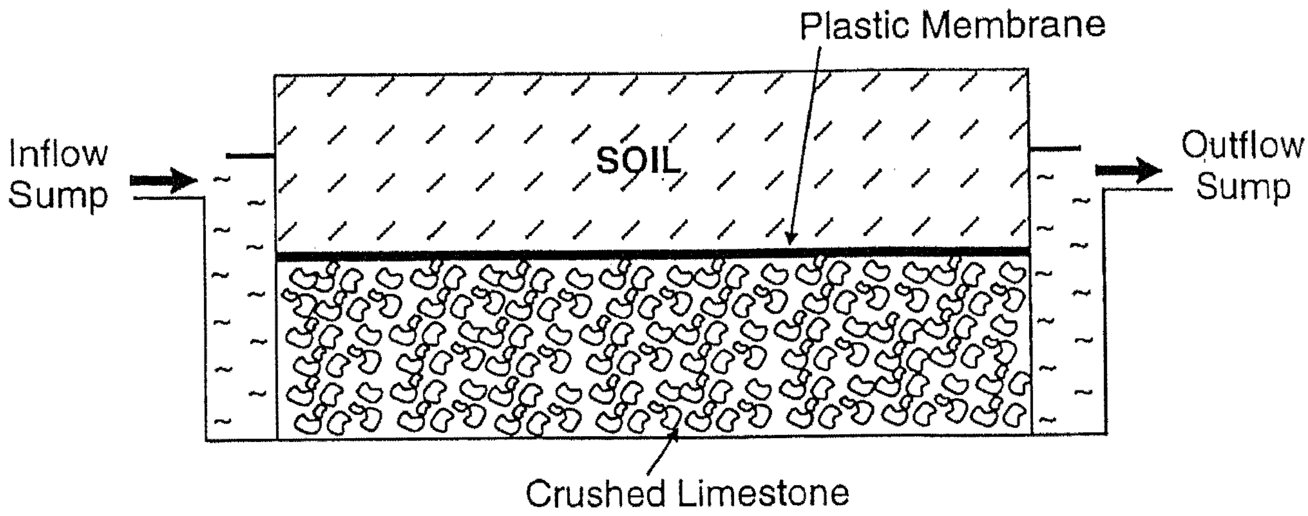
For highly concentrated AMD (such as occurs at many Australian mines), there may be no alternative but to pre-treat the initial drainage with lime or magnesia. Sodium hydroxide is generally prohibitively expensive for this purpose. Unlike chemical treatment plants, where the final pH may need to be as high as 10, to completely remove the metals, the target pH for pre-neutralisation for a wetland treatment system only needs to be about 6. The reagent should be added to the effluent prior to a cascade system to promote oxygenation of the treated stream.

5.1.2 Passive chemical (Anoxic Limestone Drains)

The installation of limestone-containing drains prior to a wetland can help to overcome the problems caused by low pH water (Nairn et al. 1991; Nairn et al. 1992). Experience has shown that an anoxic environment should be created in these drains. Anoxic conditions prevent the oxidation of Fe^{2+} to Fe^{3+}, which will passivate limestone under aerobic conditions by coating it with a layer of ferric hydroxide precipitate. Ferrous hydroxide and ferrous carbonate are much more soluble than ferric hydroxide at pH 7 to 8, and thus Fe^{2+} is unlikely to precipitate in an anoxic drain.

Anoxic limestone drains (ALDs) are relatively simple to construct (Fig. 2). They consist of a trench filled with limestone, which is covered with a layer of plastic and low permeability soil or clay to prevent the ingress of oxygen. It is recommended that high purity limestone (> 90%) be used for the construction of the drains since the rate of alkalinity generation (dissolution of the carbonate) is greatest for this material. The bed should be flooded with water prior to the installation of the capping layer to minimise the volume of air entrained within the drain. The initial development of anoxic conditions could be accelerated by the provision of a good carbon source for bacteria. The drain is ready to receive AMD once the dissolved oxygen concentration drops to near zero. In order to minimise the ingress of oxygen into the drain, the inflow and outflow points should be protected by submerging them in sumps. It is also very important to locate the inflow for the drain as close as possible to the source of the AMD to limit the extent of oxidation of Fe^{2+} once the AMD contacts the atmosphere.

The critical parameters which ultimately determine whether an anoxic limestone drain will be viable are the concentrations of Fe^{3+} and Al^{3+} in the source water, since both of these metal ions will form insoluble precipitates at the prevailing pH. Aluminium hydroxide is especially troublesome since it forms a gelatinous mass which can rapidly block the flow paths through the drain. Based on US experience, the concentrations of Fe^{3+} and Al^{3+} should each be less than 2 mg L^{-1} to ensure long term performance. Reference to Table 1 shows that, according



$$CaCO_3 + 2H^+ \Longrightarrow Ca^{2+} + H_2CO_3$$

$$CaCO_3 + H_2CO_3 \Longrightarrow Ca^{2+} + 2HCO_3^-$$

Fig. 2. Cross section through an anoxic limestone drain.

to these criteria, none of the examples given would be suitable for treatment by an ALD. In practice, however, lifetime which is somewhat less than the ideal might be an acceptable compromise.

Since ALD technology is only about five years old, there are no data on long term performance. The effects of armouring reactions and changes in permeability of the bed are unknown. This means that design criteria are still very much in the formative stages. However, the **minimum** volume of limestone required for a given lifetime can be calculated using, as the starting point, the total acidity (proton plus metal) expected to pass through the system over this period. On this basis, it is recommended (Hedin et al. 1994) that, for a design life of 30 years, an ALD should contain about 8 t limestone L^{-1} min^{-1} of AMD containing 400 mg L^{-1} acidity ($CaCO_3$ equivalent). In order to realise the full potential of an ALD for the generation of alkalinity, the residence time of water in the system should be about 14 h. Actual performance will depend critically on mass transfer limitations which will be a function of flow rate, geometry, and the rate of armouring.

5.2 Types of Wetland Treatment Units

Four basic types of wetland treatment units can be identified:

(1) Free water surface (FWS) systems which have predominantly surface flow with shallow water depths and extensive growths of emergent aquatic plants throughout;

(2) Subsurface flow (SSF) systems in which most of the water flows laterally through a bed of sand or gravel, which may be planted with emergent aquatic plants;

(3) Subsurface flow (SSF) systems in which the water flows vertically upwards or downwards through a permeable substratum, which does not contain plants; and

(4) Lagoons - ponds several metres deep with floating plants in middle of basin and rooted emergent plants around periphery. Lagoons serve primarily as sedimentation basins. However, if sufficient organic matter is present in the bottom sediments, microbial respiration can lead to anaerobic conditions which favour the immobilisation of many elements.

Each of the wetland cell designs is discussed below and the scale at which operational experience has been gained is indicated in italics.

5.2.1 Free water surface wetlands (*full scale operational*)

The first generation of constructed wetlands were of the surface flow type in which predominantly aerobic zones were in contact with the water (Fig. 3). In these systems adsorption of metals by hydrous oxides of Fe and Mn and surface organic matter, and precipitation as hydroxide or carbonate salts are the primary removal mechanisms. Many of these wetlands have been installed in the United States to treat low strength AMD from coal mines. The most important operational requirement for these systems is that the pH of the water flowing into them must be near 6, or there will be insufficient removal of the metal ions by precipitation or adsorption to prevent the ultimate exhaustion of binding sites in the substrate. This breakdown in aerobic surface flow wetlands is especially likely to occur if the input water has a pH lower than about 6. At lower values than this there will be insufficient bicarbonate alkalinity to buffer the acidity generated by the precipitation of ferric hydroxide which is the dominant metal ion-adsorbing solid in this type of wetland (Brodie 1990).

Thus pre-neutralised AMD and anoxic seepage water, at pH 7 to 8, are the best input sources for FWS wetlands since such waters contain an abundant supply of iron, which will precipitate as ferric hydroxide and bind the metals. Although the substratum a short distance beneath the soil-water interface in a FWS wetland may become anaerobic, removal of solutes via the precipitation of sulfides will be strongly limited by diffusion. Hence longer retention times are required for these systems to make effective use of this removal mechanism than for subsurface flow wetlands. For related reasons, the addition of limestone to the substrate of a FWS wetland is unlikely to be of much benefit. A similar situation pertains in a pond or lagoon.

Surface flow wetlands most closely resemble natural systems and are likely to be the most robust and require the least long term maintenance of any of the options for wetland cell design, provided that the inflow water is at the correct pH. Their one major disadvantage is that they are less efficient, and hence require a larger area, than the subsurface designs discussed below.

5.2.2 Subsurface lateral flow (*pilot scale experimental*)

In contrast to FWS wetlands, SSF systems are more costly to construct, and rigorous control of substrate type and placement is required. However, it is now recognised that SSF wetlands can be much more compact than their FWS counterparts, for a given throughput, by virtue of the higher surface area to volume ratio presented to the mine water. In lateral flow SSF wetlands the water flows horizontally through a gravel bed, after being introduced via a distribution system. The design may include provision for a shallow layer of water on top of the substrate.

SSF wetlands are particularly attractive for those cases where sulfate reduction is to be used for the removal of heavy metals since anaerobic conditions can readily be created by the addition of a supply of organic matter (such as compost) to the substrate. Wetlands of this type which have been supplemented with added organic matter have been referred to in the literature as 'compost' wetlands. Unlike FWS wetlands, the addition of limestone to the substrate will be

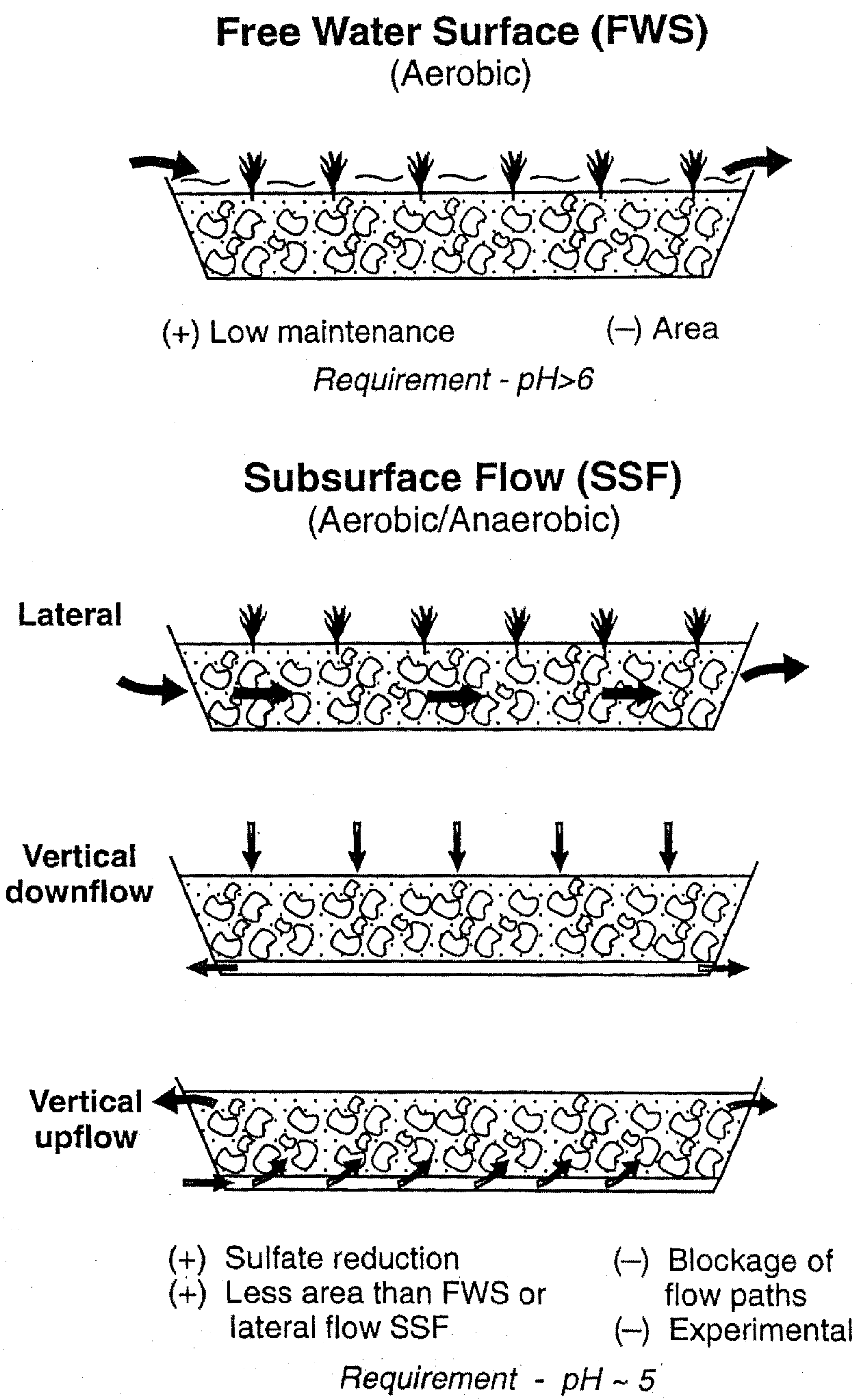


Fig. 3. **Schematic diagrams of four different types of constructed wetland cells.**

very beneficial owing to the much greater contact area, and the reducing conditions which prevent armouring by ferric hydroxide (c.f. anoxic limestone drains). The limestone may help to protect the wetland from damage if a pulse of much poorer quality water than usual enters the cell, and is likely to be especially beneficial during the initial establishment phase when the substrate is being colonised by sulfate reducing bacteria.

Although organic matter must initially be added to the substrate, the long term supply of decaying organic matter can be assured by growing aquatic plants in the substratum. One of the major unanswered questions about long term performance relates to the blockage of flow paths by plant roots, precipitates, or bacterial biomass, with consequent reduction in performance. For this reason SSF wetland cells should be located in series after a FWS cell to minimise the loading of particulates in the inflow.

5.2.3 Subsurface vertical flow (*small scale experimental*)

Recent research has indicated that wetland cells for the removal of sulfate and toxic metals from AMD should be designed as upflow units for maximum removal efficiency/unit area. In these systems the water is injected at the bottom and percolates upwards through an anaerobic zone (Morea et al. 1989; Wildeman et al. 1992). Crushed limestone can be added in a layer at the bottom of an upflow cell to initially supplement the acid neutralising capacity of the cell in the period before sulfate reducing activity is fully established.

Although water can be transmitted horizontally through a bed of organic substrate (as in the subsurface lateral flow systems discussed above), the hydraulic head is much larger and is more easily controlled in a vertical system. A vertical flow unit can be designed with a ten times higher flow rate than an identically sized horizontal system (Eger 1992). The maximum flow rate that can be used in practice before there is an unacceptable deterioration in performance will be determined by the balance between residence time and the rate of removal of the metal ions in the bed. However, the long term effective operation of a vertical flow unit will depend even more critically on the rate of occlusion of the flow paths. For this reason, vertical flow cells should be located downstream of FWS cells to minimise the concentration of particulates in the inflow. It is recommended that plants not be grown in the substrate of upflow cells Consequently, periodic additions of readily degradable organic matter will need to be made to these cells to sustain the population of sulfate reducing bacteria. Thus, the advantages of smaller size and higher efficiency of upflow cells is offset by higher maintenance requirements.

5.2.4 Ponds (Lagoons)

The main role of ponds (up to 2 m deep) in wetland systems is as sedimentation basins to trap precipitates of ferric and aluminium hydroxides generated from neutralisation and oxidation reactions occurring further upstream. This sedimentation and trapping function is typically enhanced with stands of *Typha* and *Eleocharis*. Organic matter, either added or produced over time by the decay of plants, on the bed of the pond will facilitate sulfate reduction. Ponds should precede FWS or SSF wetland cells to reduce the loading of solids into these components of the system. Low solids loading is especially critical for the long term functioning of SSF systems. Whilst ponds are simple to construct, and are vital components of a wetland treatment system, thermal stratification can considerably reduce operating efficiency. If stratification occurs, a thin layer of water moving at relatively high velocity can transport untreated water across the surface of the pond to the discharge point. Thermal stratification is especially likely to occur when there is a large diurnal range in air temperatures. To prevent a stratified layer from forming, a curtain hanging from a floating boom can be installed near the upstream end of a pond to force the incoming surface water to mix with the lower layers.

5.3 Riffle Zones and Algal Mats

Effective oxygenation of AMD is critical for the rapid oxidation of Fe^{2+} to Fe^{3+} at the beginning of the wetland treatment system, and the oxidation of Mn^{2+} to Mn^{4+} in the final polishing section.

The dissolution of oxygen is enhanced by creating turbulence in the water. This can be achieved by linking a series of cascades with a shallow channels lined with coarse rock (rip rap). The length of the cascade that is required will depend on the input concentration of Fe^{2+}. The pH of the water entering the cascade should be near 6 to not only maximise the rate of oxidation but to make maximum use of the adsorptive capacity of the ferric hydroxide that is produced. Growth of algae in the riffle zone will further enhance the process of precipitation and metal removal.

Riffle zones should also be located after subsurface flow cells to oxygenate the water and to provide optimum conditions for the removal of Mn^{2+}. The pH of the water leaving the final wetland cells should lie between 7 and 8. At this pH, algae can very effectively oxidise Mn^{2+} to Mn^{4+}, which will precipitate. Manganese is regarded as the most difficult metal to remove from AMD. Indeed some overseas authors have recommended that the size of an FWS wetland for the removal of Mn should be three times larger than that required for the removal of an equivalent amount of iron. However, the specific conditions needed for the efficient removal of manganese have not been provided in most wetland systems that have been reported in the literature.

Based on work done in a passive treatment system for water from the Hilton mine (Chapman et al. 1988), the authors of this paper found that Mn^{2+} can be very efficiently removed from pH neutral water. Filamentous blue green algae (cyanobacteria) attached to the bed of a shallow stream created a high pH (> 9) oxygen-rich microenvironment during the day, and Mn oxides precipitated on the surface of the algal cells. The work of the algae was enhanced by an abiotic process of autocatalytic oxidation of Mn^{2+} that occurred on the surfaces of the gravel on the bed of the stream.

6.0 ROLE OF EMERGENT PLANTS IN WETLANDS

The role of plants in wetlands has been somewhat controversial. Detailed studies have shown that, in contrast to the floating plants such as duckweed and water hyacinths used in lagoon-based systems, uptake of metals into the tissues of rooted emergent aquatic plants (e.g. Typha) in wetlands is not a major route for metal removal from mine waters containing higher than trace level concentrations of metals. It has been estimated that less than 5% of the metals input into a wetland bed are incorporated into the plants (Sencindiver and Bhumbla 1988; Machemer et al. 1990). Plants do, however, enhance the removal of solutes by indirect means. In the case of FWS systems plants reduce water velocity and lead to the sedimentation and trapping of metal enriched particulates, and provide a substrate for epiphytic algae. The reduced velocity also increases the residence time and lengthens the treatment period for a given parcel of water.

The second important role of plants is as a long term source of the decaying organic matter required to sustain sulfate reduction after the initial organic amendment in the wetland has been consumed. However, for units designed to treat minewater at high throughput, periodic supplementation with additional organic matter may be periodically required to maintain the metal removal rate.

The third function of rooted emergent plants is to create oxidised microzones around their root networks that extend into the anaerobic substratum (Brix 1987; Macfie and Crowder 1987; Dunbabin et al. 1988). These sites promote the precipitation of high surface area iron and manganese oxyhydroxide phases (Tarutis and Unz 1990) which have considerable scavenging capacity for heavy metals (Jenne 1968). Consequently, metal ions that escape from the lower anaerobic zone can be immobilised in the aerobic regions located around the roots and at the top of the gravel bed. The microenvironment in the vicinity of roots also favours the formation of pyrite from the iron monosulfides initially formed under strongly reducing conditions in sediments.

7.0 STATE-OF-THE-ART

Historically, the majority of wetlands constructed for the treatment of mine waters in Australia and overseas have consisted of a combination of FWS and lagoon components, located in existing stream channels. These systems are characterised by a predominantly aerobic environment in contact with the flowing water. The dense stands of reeds growing in these systems serve as a support for bacteria and algae, and act as a hydraulic barrier to reduce the speed of the water, and enhance residence time.

7.1 Overseas

An overview of the literature reveals that most of the full-scale wetlands that have been constructed have been treating AMD that is not only relatively dilute, but which contains relatively low concentrations of toxic heavy metals (Wieder 1989). Moreover, the fact that the oldest functioning wetland for treating AMD has been operating for only ten years, means that there is a lack of long term performance data even for relatively low grade AMD. Most large wetlands built thus far have been of the FWS type. Subsurface flow systems for the treatment of AMD are still at the experimental or pilot scale stage.

7.2 Australia

The use of constructed wetlands for treating mine drainage in Australia has so far been quite limited and mainly confined to situations where the effluent entering the wetland is close to neutral pH. There is virtually no information available for experimental systems which may be in the process of being trialled for low to moderate strength AMD.

Examples of functioning wetlands treating near-neutral pH mine water are located at the Ranger uranium mine (NT), Tom's Gully gold mine (NT), Woodcutters base metal mine (NT), the Hilton base metal mine at Mt Isa (Qld), the Hellyer mine (Tas) and the Westralian Sands Ltd synthetic rutile plant (WA).

At the Ranger mine, the solutes of interest are uranium (in the form of UO_2^{2+}), Mg^{2+}, SO_4^{2-}, and Mn^{2+} (D.R. Jones unpublished data). The constructed wetland consists of three shallow ponds, (each 1.5 m deep and 20 x 20 m in area) in series which discharge into a retention basin. Results so far indicate effective removal of U at flow rates up to 1,000 $m^3 d^{-1}$.

The Tom's Gully gold mine discharges drainage into an ox-bow billabong containing *Typha* (Noller et al. 1992). This overflows into the Mount Bundy Creek. A range of transition and base metals as well as As are effectively removed in this system. This is an example of a natural wetland in the vicinity of a mine that can be used to improve the water quality of mine drainage.

At Woodcutters mine, effluent with near-neutral pH passes through a settling pond and then discharges to a pond and wetland channel, prior to being released into a creek. Metals removed include Cd, Pb, Cu and Zn.

The water at the Hilton mine is slightly acidic as a result of being supersaturated with CO_2

(Chapman et al. 1988). However, the pH rises to over 8 after degassing. The wetland consists of a riffle zone feeding into a 4.2 ha retention pond (average depth of 1.3 m), which overflows into a natural creek channel. This system, populated by reeds and algae, has been studied in detail and its efficiency optimised to an extent that the water can be reused for ore processing. Elements being removed include Fe, Zn, Mn and Tl. The important processes that have been identified include degassing, precipitation, coprecipitation, adsorption, sedimentation, autocatalytic oxidation of Mn^{2+} and algal photosynthesis leading to a microenvironmental rise in pH. This system has functioned successfully for flow rates of up to 3,000 m^3 d^{-1}.

At the Hellyer mine, a large wetland consisting of a number of FWS cells (populated with *Juncus sp.*) arranged in a parallel configuration was constructed in late 1992. This followed trials with a small pilot scale system. The wetland receives pH neutral input containing Pb, Fe, Mn, and sulfate from the tailings dam. This has been a relatively expensive system to construct.

The Westralian Sands synthetic rutile plant (Masters 1988) produces an acidic effluent which is neutralised with lime before it is discharged into a 2 ha wetland. This system, commissioned in 1986, was designed as a backup for the chemical treatment plant, rather than as a primary treatment facility. The elements removed from the usually pH neutral water include Fe, Mn and sulfate. However, following process malfunction the pH of the water can be as low as 3 to 4. Under these circumstance the wetland has been found to provide about 4 days of treatment, before its buffering capacity is exhausted. The system has required extensive maintenance involving the replacement of vegetation and substrate (Ryan and Hosking 1992). This is possibly a result of the damaging effects of the acute shock loads to which it is periodically exposed.

8.0 DESIGN FACTORS AND COST

The design of a wetland will depend critically on the quantity of effluent to be treated, the types of contaminant to be removed, the local climatic conditions, and the geomorphology and land area available. A comprehensive overview of the many facets needed to be considered when designing wetland systems for the treatment of wastewaters is provided in a book written by Reed et al. (1988).

The aspect ratio most often used for wetland cells lies between 5:1 and 10:1 (L:W) to provide maximum path length for a given area. Single or multiple cells in series or in parallel can be used, but care has to be taken to avoid channelling or shortcircuiting. Serpentine arrangements using channels or baffles increase contact time and mass transfer between substrate, biota and water. In some cases, clay or synthetic liners are necessary if protection of the water table is of concern. Depending on the requirements for the wetland, substrates such as clay, organic soils or mushroom compost may need to be introduced and macrophyte species planted. Aesthetic considerations can also be important in design and the wetland can eventually become a valuable wildlife refuge. Wetlands should be designed to avoid drying out as this can dramatically reduce the biomass, and mobilise metals initially trapped as sulfides.

As stated above, most operational experience in the use of constructed wetlands for the treatment of AMD has been obtained with FWS systems. Not only are the design factors for this category of wetland still largely empirical, but those factors which have been published are primarily for the removal of Fe and Al from low strength AMD. Brodie and others have proposed guidelines (Table 3) based on their work on FWS wetlands constructed by the Tennessee Valley Authority (Brodie et al. 1988). These are intended as a guide only and depend on many factors not specifically considered in this table. The criteria for Mn are generally based on sub-optimal configurations for the removal of this element.

Table 3. Sizing guidelines for FWS constructed wetlands (from Brodie et al. 1988).

For target discharge limits:		Fe ≤ 3 mg L^{-1} Mn ≤ 2 mg L^{-1}
Influent pH < 5.5		Design Factor
Fe	2 m^2 of wetland mg^{-1} Fe in inflow min^{-1}	2 m^2 mg^{-1} min^{-1}
Mn	7 m^2 of wetland mg^{-1} Mn in inflow min^{-1}	7 m^2 mg^{-1} min^{-1}
Influent pH > 5.5		
Fe	0.75 m^2 of wetland mg^{-1} Fe in inflow min^{-1}	0.75 m^2 mg^{-1} min^{-1}
Mn	2 m^2 of wetland mg^{-1} Mn in inflow min^{-1}	2 m^2 mg^{-1} min^{-1}

There has been little quantitative evaluation of the effect of temperature on wetland performance. This is particularly relevant for the Australian context where many mining provinces have year round temperatures that are considerably higher than in the USA and Europe from where published design guidelines have originated.

Costs of constructing a wetland can vary enormously and depend particularly on the topography of existing land forms, whether natural or synthetic liners are required, and the composition of the effluent. Typical price ranges quoted for FWS wetlands in the United States are US$8 to US$56 m^{-2} (Skousen et al. 1992), US$3 to US$32 m^{-2} (Wieder 1989), and US$66 to US$139 m^{-2} (Hiel and Kerins 1988).

9.0 ASSEMBLING THE WETLAND

Based on the above discussion, a conceptual design can now be formulated for a wetland to treat moderate strength AMD. This design is shown in Fig. 4. The abiotic, pre-neutralisation, oxidation, and settling components are located before the wetland components to ensure maximum removal of acidity and metals before the more sensitive biological processes are exposed to the drainage. Experience on pilot scale systems has indicated that FWS cells should precede SSF cells to minimise the potential for blockage of the substrate. The latter serve to remove the residual concentrations of heavy metals via the precipitation of sulfides. However, in particular cases it may be necessary to install a SSF cell prior to the first FWS unit. This could occur for the situations where the first part of the system cannot achieve a pH near 6, or where the concentrations of heavy metals are still too high to permit the growth of higher plants. Sulfate reduction will proceed at a pH which is lower than that needed for efficient removal of heavy metals ions by adsorption on ferric hydroxide. This process is not only more tolerant to elevated concentrations of these metals, but also generates alkalinity.

If the concentrations of manganese in the treated AMD are likely to exceed discharge criteria, then a riffle zone incorporating an algal mat should be installed as the final stage of the system. Filamentous blue green algae are preferred for this role since they can create and maintain a higher pH in the water column during daylight hours than can green algae.

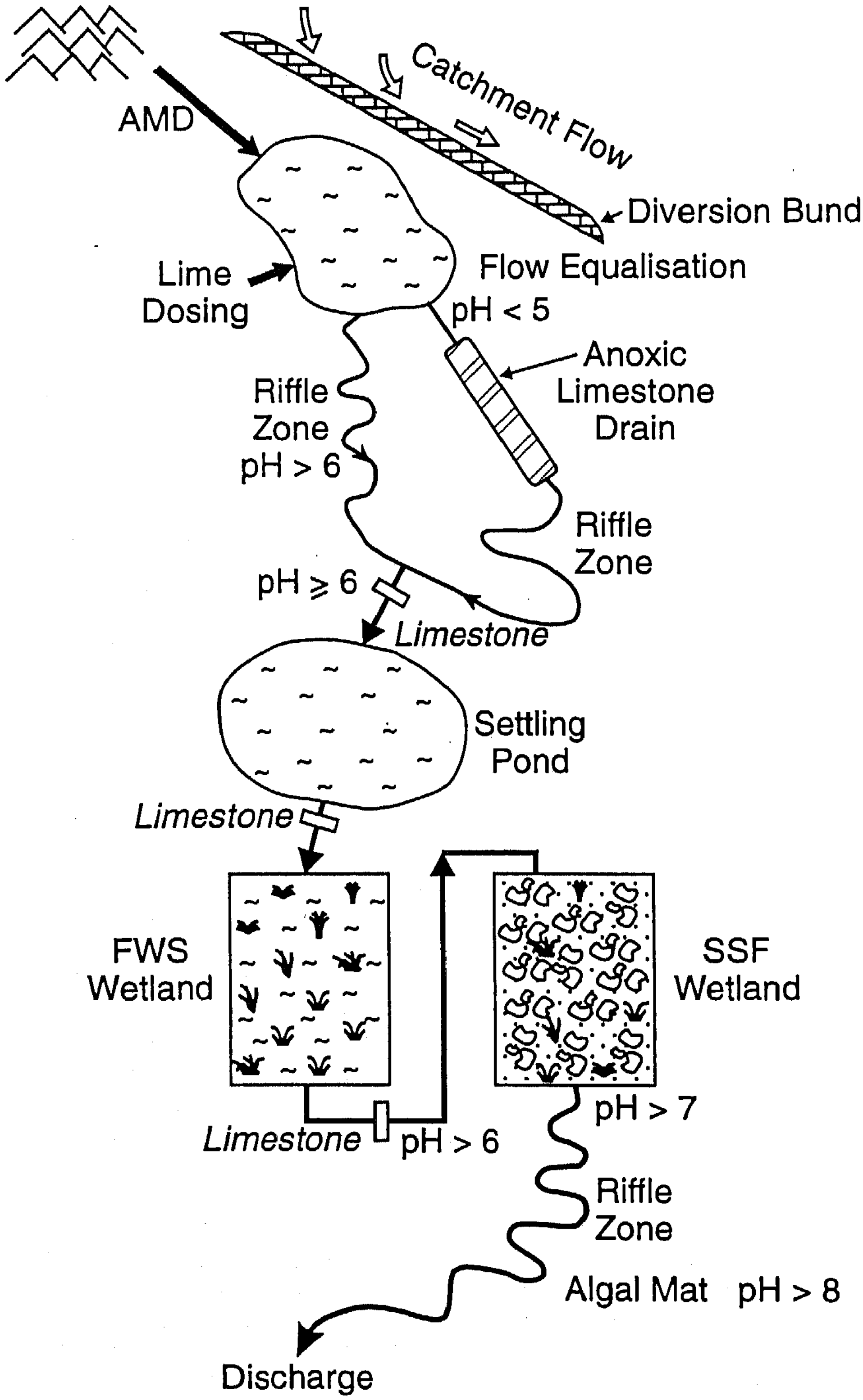


Fig. 4. **Conceptual design of a complete wetland system for treating AMD.**

10.0 LONG TERM CONSIDERATIONS

The design lifetime of a wetland treatment system is a key issue. In some cases significant volumes of AMD requiring treatment may only be produced during the life of the mine, prior to rehabilitation of source material (e.g. waste rock dumps). In this case there would obviously need to be less emphasis on long term sustainability. Wetlands, by their very nature will accumulate heavy metals and so the long term implications of a contaminated site should be addressed when a wetland system is being planned. In all cases performance monitoring will be required throughout the life of the system. Typically the frequency and spatial intensity of monitoring would need to be highest during the initial establishment and proving phase. The monitoring program should be designed to not only measure the end-of-pipe quality but also to provide an early warning of a decline in performance in the upstream components of the system.

11.0 CONCLUSIONS

A properly constructed wetland treatment system does have the potential to be self sustaining with construction and operating costs that are lower than for a conventional chemical treatment plant. The hydrological regime and the chemical composition of the drainage are the key factors which will determine the complexity and size of the system required to produce effluent suitable for discharge to receiving waters. It is important to recognise that most full scale systems that have been built to date have either been treating low strength AMD or mine drainage which has an initial pH above 4. Systems to treat moderate strength, low pH AMD are still at the developmental stage.

Wetlands for the treatment of more concentrated sources of AMD are unlikely to be walkaway and maintenance free. For these cases pre-neutralisation systems will require replenishment, sedimentation ponds will need to be periodically cleaned, and the substrate in some of the more heavily impacted upstream wetland cells may need to be replaced.

In all cases, long term monitoring of performance will be required.

12.0 REFERENCES

Batal, W., Laudon, L.S., Wildeman, T.R., and Mohdnoordin, N. (1989). Bacteriological tests from the constructed wetland of the Big Five Tunnel, Idaho Springs, Colorado. In 'Constructed Wetlands for Wastewater Treatment; Municipal, Industrial and Agricultural'. (Ed. D.A. Hammer.) pp. 550–557. (Lewis Publishers.)

Brierley, C.L. (1990). Bioremediation of metal-contaminated surface and groundwaters. *Geomicrobiology Journal* 8, 201–223.

Brix, H, (1987). Treatment of wastewater in the rhizosphere of wetland plants-the root-zone method. *Water Science and Technology* 19, 107–118.

Brodie, G.A., Hammer, D.A., and Tomljanovich, D.A. (1988). Constructed wetlands for acid control in the Tennessee Valley. In 'Mine drainage and surface mine reclamation (conference proceedings). Volume 1: Mine water and mine waste, Pittsburgh, PA USA, 19–21 Apr 1988'. US Department of the Interior, Bureau of Mines Information Circular IC9183. pp. 325–331.

Brodie, G.A. (1990). Treatment of acid drainage using constructed wetlands, Experiences of the Tennessee Valley Authority. In 'Proceedings, 1990 National Symposium on Mining'. (Ed. D.H. Graves.) pp. 77–83. (University of Kentucky Publications: Lexington.)

Chapman, B.M., Jones, D.R., and Jung, R.F. (1983). Processes controlling metal ion attenuation in acid mine drainage streams. *Geochimica et Cosmochimica Acta* 47, 1957–1973.

Chapman, B.M., Jones, D.R., Jung, R.F., Jones, C.J., Kadletz, O., and Teague, J.W.S. (1988).

Treatment and Utilisation of Hilton Mine Water. In 'Proceedings of the 3rd Intl. Mine Water Congress, Melbourne, Australia. pp. 147–156. (Australasian Institute of Mining and Metallurgy.)

Dunbabin, J.S., Pokorny, J., and Bowmer, K.H. (1988). Rhizosphere oxygenation by *Typha domingensis Pers.* in miniature wetland filters used for metal removal from wastewaters. *Aquatic Botanical* 29, 303–317.

Dvorak, D.H., Hedin, R.S., Edenborn, H.M., and McIntire, P.E. (1992). Treatment of metal-contaminated water using bacterial sulfate reduction: results from pilot-scale reactors. *Biotechnology and Bioengineering* 40, 609–616.

Eger, P. (1992). Wetland Treatment for Trace Metal Removal from Mine Drainage: The Importance of Aerobic and Anaerobic Processes. In 'Proc. Intl. Specialist Conference Wetlands Downunder-Wetland Systems in Water Pollution Control, Uni. Of NSW, Sydney, NSW, Australia, Nov. 1992'. pp. 46.1–46.9.

Gloyna, E.F., and Espino, E. (1969). Sulfide production in waste stabilization ponds. *Journal of the Sanitary Engineering Division* ASCE, 95, SA3.

Hammack, R.W., and Edenborn, H.M. (1992). The removal of nickel from mine waters using bacterial sulfate reduction. *Applied Microbiology and Biotechnology* 37, 674–678.

Hedin, R.S., Hammack, R., and Hyman, D. (1989). Potential Importance of Sulfate Reduction Processes in Wetlands Constructed to Treat Mine Drainage. In 'Constructed Wetlands for Wastewater Treatment - Municipal, Industrial and Agricultural'. (Ed. D.A. Hammer.) pp. 508–509. (Lewis Publishers: Chelsea, Michigan.)

Hedin, R.S., and Nairn, R.W. (1992). Designing and sizing passive mine drainage treatment systems. Proceedings Thirteen Annual West Virginia Surface Mine Drainage Task Force Symposium - April 8–9, 1992.

Hedin, R.S., Nairn, R.W., and Kleinman, R.L.P. (1994) Passive treatment of coal mine drainage. US Bureau of Mines Information Circular IC-9389, Pittsburgh, PA, USA, 35p.

Hiel, M.T., and Kerins, F.J. Jr. (1988). The Tracey wetlands: a case study of two passive mine drainage treatment systems in Montana. In 'Mine drainage and surface mine reclamation (conference proceedings). Volume 1: Mine water and mine waste. Pittsburgh, PA, USA, 19–21 Apr 1988'. US Department of the Interior, Bureau of Mines Information Circular IC9183. pp. 352–358 (Apr 1988).

Jenne, E.A. (1968). Controls on manganese, iron, cobalt, nickel, copper and zinc concentrations in soils and water: the significant role of hydrous manganese and iron oxides. In 'Trace Inorganics in Water'. (Ed. R.A. Baker.) Advances in Chemistry Series No. 73, 337–387. (American Chemical Society: Washington, DC.)

Macfie, S.M., and Crowder, A.A. (1987). Soil factors influencing ferric hydroxide plaque formation on roots of *Typha latifolia* L. *Plant and Soil* 102, 177–184.

Machemer, S.D., Lemke, P.R., Wildeman, T.R., Cohen, R.R., Klusman, R.W., Emerick, J.C., and Bates, E.R. (1990). Passive treatment of metals mine drainage through use of a constructed wetland. In 'Proceedings of the 16th Annual Hazardous Waste Research Symposium (EPA/600/9–90–037 ed.)'. pp. 104–114. (US Environmental Protection Agency: Cincinnati OH.)

Masters, B.K. (1988). Biological filtration of industrial effluent water, a successful case study: Third International Mine Water Congress Oct. 1988, Melbourne, Australia. pp. 763–770.

McIntire, P.E., Edenborn, H.M., and Hammack, R.W. (1990). Incorporation of Bacterial Sulfate Reduction into Constructed Wetlands for the treatment of Acid and Metal Mine Drainage: 1990 National Symposium on Mining, May 14–18, 1990. pp. 207.

Morea, S.C., Olsen, R.L., and Chappel, R.W. (1989). Assessment of a passive treatment system for acid mine drainage at a Colorado Superfund site, Camp Dresser & McKee: Denver.

Nairn, R.W., Hedin, R.S., and Watzlaf, G.R. (1991). A Preliminary Review of the Use of Anoxic Limestone Drains in the Passive Treatment of Acid Mine Drainage. In 'Proceedings West Virginia Surface Mine Drainage Task Force Symposium (Twelfth Annual ed.)'. (West

Virginia surface Mine Drainage Task Force/West Virginia Mining & Reclamation Association: Morgantown, West Virginia.)

Nairn, R.W., Hedin, R.S., and Watzlaf, G.R. (1992). Generation of Alkalinity in an Anoxic Limestone Drain. In 'Proceedings of the National Meeting of the American Society for Surface Mining and Reclamation, Duluth, Minnesota, American Society for Surface Mining and Reclamation, ASSMR9–92 June 14–18, 1992'.

Noller, B.N., Woods, P.H., and Ross, B.T. (1992). Case studies of wetland filtration of mine water in constructed and naturally occurring systems in northern Australia. In 'Proc. Intl. Specialist Conference Wetlands Downunder-Wetland Systems in Water Pollution Control, Uni. of NSW, Sydney, NSW, Australia, Nov. 1992'. pp. 47.1–47.9

Patterson, J.W. (1985). 'Industrial Wastewater Treatment Technology (2nd ed.).' (Butterworth Publishers: Stoneham, MA.)

Postgate, J.R. (1979). 'The Sulphur-Reducing Bacteria.' (Cambridge University Press: New York.)

Ryan, P., and Hosking, R. (1992). A review of wetland filter systems for mining operations. In 'Proceedings of the International Specialist Conference on Wetlands Downunder -Wetland Systems in Water Pollution Control, Uni. of NSW, Sydney, NSW, Australia, Nov. 1992'. pp. 1–16.

Reed, S.C., Middlebrooks, E.J., and Crites, R.W (1988). 'Natural Systems for Waste Management and Control'. (McGraw Hill Book Company: Sydney.)

Sencindiver, J.C., and Bhumbla, D.K. (1988). Effects of cattails (Typha) on metal removal from mine drainage. Bureau of Mines Information Circular, 9183, pp. 359–366.

Skousen, J.G., Phipps, T.T., and Fletcher, J. (1992). Acid mine drainage treatment alternatives. In 'Land reclamation: advances in research and technology - proceedings of the international symposium, Nashville, TN, USA'. (Eds T. Younos, P. Diplas, and S. Mostaghimi.) pp. 297–303. (American Society of Agricultural Engineers: St Joseph, MI USA.)

Tarutis Jr, W.J., and Unz, R.F. (1990). Chemical diagenesis of iron and manganese in constructed wetlands receiving acidic mine drainage. In 'Constructed Wetlands in Water Pollution Control'. (Eds. P.E. Cooper and B.C. Findlater.) pp. 429–440. (Permagon Press: Oxford.)

Wildeman, T.R., and Laudon, L.S. (1989). Use of Wetlands for Treatment of Environmental Problems in Mining: Non-Coal-Mining Applications. In 'Constructed Wetlands for Wastewater Treatment - Municipal, Industrial and Agricultural'. (Ed. D.A. Hammer.) pp. 221–231. (Lewis Publishers: Chelsea, Michigan.)

Wildeman, T., Brodie, G., and Gusek, J. (1992). Wetland Design for Mining Operations, Colorado School of Mines.

Wieder, R.K. (1989). A survey of constructed wetlands for acid coal mine drainage treatment in the eastern United States. *Wetlands* 9, 229.