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Chapter Thirty-Three

CONSTRUCTING WETLANDS FOR PASSIVE TREATMENT OF COAL MINE DRAINAGE

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INTRODUCTION

As has been stated elsewhere in this book, wetlands have an inherent, though limited capability, to treat polluted waters. This capability has been recognized due both to the all-too-common past practice of discharging partially-treated or untreated effluents into a convenient wetland area, and the many volunteer wetlands that have developed at point and nonpoint discharge areas. Often ecologists who were attempting to evaluate potential adverse consequences have often found that the wetlands not only tolerated such pollution well, but actually removed significant levels of contaminants. This was certainly true for coal mine drainage (CMD). The concept of passive treatment of CMD can be traced back to independent studies of Sphagnum wetlands by graduate students at two different universities (1, 2). Scientists and engineers quickly realized that this potentially low-cost water treatment technology would be forfeit if they degraded or destroyed natural wetland habitats. However, if wetlands were created primarily for water treatment, the additional potential environmental benefits they provided would be welcome (3, 4). Wetlands have therefore been constructed to receive various effluents, including domestic and industrial sewage and treatment plant waste water, as well as waters that discharge from active and abandoned coal mines (5-8).

Contaminated CMD is an extremely common problem in the Appalachian coal fields of the United States. At active operations, U.S. mining companies spend over one million dollars a day to treat mine water chemically so that it can be legally discharged. The high cost of water treatment, places a serious financial, burden on many mining companies and has contributed to the bankruptcies of many others.

At most U. S. mines abandoned prior to 1978, no one is legally required to treat mine water discharges. As a result, over 19,300 km (12,000 miles) of rivers and streams and over 730 km² (180,000 acres) of lakes and reservoirs in the U.S. are adversely affected. Over a third of this total is the result of acidic mine drainage (AMD) which is generated by the weathering of pyrite exposed during the mining process (9). Acidic CMD is a major problem in northern Appalachia (in Pennsylvania, eastern Ohio, northern West Virginia and western Maryland) as well as many other sites, worldwide. In the U.S., State and Federal reclamation agencies, local conservation groups and watershed associations all consider the treatment of contaminated coal mine drainage to be a high priority. Unfortunately, available funds are insufficient to chemically treat such discharges, except in a few watersheds of exceptional value.

However, natural processes commonly ameliorate mine drainage pollution. As contaminated coal mine drainage flows into and through receiving systems (streams, rivers, and lakes), its toxic characteristics decrease naturally as a result of chemical and biological reactions and by dilution with uncontaminated water. Metal contaminants of coal mine drainage precipitate as oxides and hydroxides under the aerobic conditions found in most surface waters. Dissolved Fe precipitates as an oxyhydroxide, staining the bottoms of many streams orange and often accumulating to sufficient depths to suffocate benthic organisms. Less commonly, dissolved Al precipitates as a white hydroxide. Passive treatment systems are designed to facilitate these natural ameliorative processes, and to limit the volume of precipitates that reach the receiving stream.

MECHANISMS OF PASSIVE TREATMENT

Application of Natural Processes

During the last decade, the possibility that mine water might be treated passively in wetland-based systems has developed from an experimental concept to full-scale field implementation at over a thousand active and abandoned mine sites. The early constructed wetlands were planted with *Sphagnum*, in an attempt to simulate natural bog-type wetlands. *Sphagnum* has a tremendous ability to accumulate metals, and tolerates acidic conditions well, but field tests proved that if exposed to even moderately-contaminated AMD for prolonged periods, it absorbs so much metal that it petrifies.

As a result, most drainage are now aday *Typha* does not accu Bhumbla determined influent iron removed Instead, such plants p aerate sediment and l removal in the wetlan

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As a result, most wetlands constructed to treat contaminated coal mine drainage are nowadays planted with emergent plants, especially cattails (*Typha*). *Typha* does not accumulate metals efficiently. In one study, Sencindiver and Bhumbla determined that Typha bioaccumulation accounted for only 0.2% of the influent iron removed in a wetland that received drainage with 10 mg/L Fe (10). Instead, such plants provide an appropriate environment for bacterial activity, aerate sediment and help filter out precipitated floc, all of which favor metal removal in the wetland substrate (11).

Ideally, passive treatment systems require no input of chemicals and little or no operation and maintenance requirements. The costs of passive treatment systems are generally measured in their land use requirements. Passive treatment systems use contaminant removal processes that are slower than that of conventional treatment and thus require longer retention times and larger areas in order to achieve similar results.

The goal of passive mine drainage treatment systems is to enhance the natural amelioration processes so that they occur within the treatment system, not in the receiving water body. Two factors that determine whether this goal can be accomplished are the kinetics of the contaminant removal processes and the retention time of the mine water in the treatment system. The retention time for a particular mine site is often limited by available land area. However, the kinetics of contaminant removal processes can often be affected by manipulating the environmental conditions that exist within the passive treatment system. Efficient manipulation of contaminant removal processes requires that the nature of the rate-limiting aspects of each removal process be understood.

Contaminants in Coal Mine Drainage

The common contaminants of concern in CMD are iron, manganese, aluminum and acidity. To be legally discharged, water originating at sites mined since May 4, 1984 must have no net acidity (alkalinity must equal or exceed acidity), a pH between 6.0 and 9.0, total iron levels that never exceed 6.0 mg/L or an average of 3.0 mg/L for 30 consecutive days and total manganese levels that never exceed 4.0 mg/L or an average of 2.0 mg/L for 30 consecutive days. Manganese is regulated as a surrogate for other more toxic metals that are occasionally present in AMD. Manganese removal in a conventional treatment plant operation requires that the pH be raised above 10, which also removes the other more toxic contaminants, if present. Ironically, passive removal of manganese at circumneutral pH does not necessarily assure removal of these more toxic metals.

Unless the pH of the mine water is below about 3, most of the dissolved iron is present in mine water as ferrous iron, Fe²⁺. To precipitate Fe²⁺, it typically must be oxidized (to ferric iron—Fe³⁺) and hydrolyzed, to ferric hydroxide or oxyhydroxide. At neutral or alkaline pH, these reactions proceed fairly rapidly, with or without bacterial activity. However, the hydrolysis reaction can lower pH, and every unit decrease in pH slows the abiotic oxidation rate as much as 100 fold.

At lower pH, the precipitation of dissolved iron is rate-controlled by the activity of the various iron-oxidizing bacteria.

The ferric hydroxide sludge that forms at slightly acidic to circumneutral pH is much denser than that associated with the alkaline conditions of conventional chemical treatment, and may prove to have sufficient economic value to justify recovery (12). Aluminum is also precipitated as a hydroxide in passive systems, typically over the pH range of 4.5-8.

Manganese can be removed passively, by both abiotic and bacteriallycatalyzed oxidation reactions, but is stable only after almost all of the Fe²⁺ has been oxidized or removed, and when the pH is 6 or above. Manganese-oxidizing bacteria are involved, especially initially, but once the Mn is precipitated, the oxidation reaction is autocatalytic (13), so it is not clear whether the long-term rate is controlled by bacterial activity or not.

Acidity is a measurement of the base neutralization capacity of a volume of water. Three types of acidity exist: proton acidity associated with pH (a measure of free H⁺ ions), organic acidity associated with dissolved organic compounds, and mineral acidity associated with dissolved metals (14, 15). Mine waters generally have a very low dissolved organic carbon content, so organic acidity is negligible. The acidity of coal mine drainage arises from free protons (low pH) and the mineral acidity from dissolved Fe, Mn, and Al. These metals are acidic because they can undergo hydrolysis reactions that produce H⁺.

Acidity is normally determined by titration, after adding hydrogen peroxide and heating the sample to assure that all the mineral acidity is accounted for. However, acidity can also be calculated for coal mine drainage using the pH and the measured concentrations of acidic metals:

$$Acid_{cate} = 50(2Fe^{2+}/56 + 3Fe^{3+}/56 + 3A1/27 + 2Mn/55 + 1000(10^{\cdot pH}))$$
(1)

where all concentrations are in mg·L⁻¹ and 50 is the equivalent weight of CaCO3, and thus transorms mg/L of acidity into mg/L CaCO₃ equivalent (16).

When mine water has pH > 4.5, it has acid neutralizing capacity and is said to contain alkalinity. Alkalinity can result from hydroxyl ions (OH), carbonate, silicate, borate, organic ligands, phosphate, and ammonia (14). The principal source of alkalinity in mine water is dissolved carbonate, which can exist in a bicarbonate (HCOs²) or carbonate form (COs²). Both can neutralize proton acidity. In the pH range (5-8) of most alkaline waters, bicarbonate is the principal source of alkalinity.

Alkalinity and acidity are not mutually exclusive. When water contains both mineral acidity and alkalinity, a comparison of the two measurements results in a determination as to whether the water is net alkaline (alkalinity > acidity) or net acidic (acidity > alkalinity). Net alkaline water contains enough alkalinity to neutralize the mineral acidity represented by dissolved ferrous iron and manganese. As these metals oxidize and hydrolyze, the proton acidity that is produced is rapidly neutralized by bicarbonate.

A major source of bicarbonate in many anaerobic environments is the dissolution of carbonate rocks, such as limestone. Carbonate dissolution can result in higher concentrations of bicarbonate in anaerobic mine water environments than in aerobic environments for two reasons. First, the absence of Fe³⁺ in most anaerobic environments limits the formation of FeOOH or Fe(OH)₃ coatings that armor carbonate surfaces and inhibit further carbonate dissolution in aerobic environments (17). Second, the solubilities of carbonate compounds are directly affected by the partial pressure of dissolved CO₂ (13, 14, 18). Anaerobic mine water environments commonly contain high CO₂ partial pressures because of the decomposition of organic matter and the neutralization of proton acidity. Limestone is often added to substrate materials during wetland construction to generate alkalinity, since organic decomposition in this environment generally produces anaerobic conditions. Alternatively, anoxic mine water can be pretreated using a buried bed of limestone, so that wetland treatment is made more efficient. Such a system is referred to as an anoxic limestone drain (ALD).

Another approach to passively generating alkalinity in constructed wetlands is to foster bacterial sulfate reduction. In this natural process, bacteria oxidize organic compounds and release hydrogen sulfide and bicarbonate. Sulfate-reducing bacteria require the presence of sulfate, suitable concentrations of lowmolecular weight carbon compounds, pH greater than 4, and the absence of oxidizing agents such as oxygen, Fe³⁺ and Mn⁴⁺. These conditions are commonly satisfied in treatment systems that receive coal mine drainage and contain organic matter. High concentrations of sulfate (> 200 mg·L⁻¹) are characteristic of contaminated coal mine drainage. The oxygen demand of organic substrates causes the development of anoxic conditions and an absence of oxidized forms of iron or manganese. The low-molecular weight compounds that sulfate-reducing bacteria utilize (lactate, acetate) are common end products of microbial fermentation processes in anoxic environments. The pH requirements can be satisfied by alkalinity generated by the microbial activity and carbonate dissolution.

Bacterial sulfate reduction directly affects concentrations of dissolved metals by precipitating them as metal sulfide solids. The removal of dissolved metals as sulfide compounds depends on pH, the solubility product of the specific metal sulfide, and the concentrations of the reactants. For CMD, where metal contamination is generally limited to Fe, Mn, and Al, the hydrogen sulfide produced by bacterial sulfate reduction primarily affect dissolved iron concentrations. Aluminum does not form any sulfide compounds in wetland environments and the relatively high solubility of MnS makes its formation unlikely.

The precipitation of metal sulfides in an organic substrate improves water quality be decreasing the mineral acidity without causing a parallel increase in proton acidity. Proton-releasing aspects of the H₂S dissociation process (H₂S \rightarrow 2H⁺ + S²) are neutralized by an equal release of bicarbonate during sulfate reduction. An organic substrate in which 100% of the H₂S produced by sulfate reduction precipitated as FeS would have no effect on the mine water pH or alkalinity (although acidity would decrease). In fact, however, the chemistry of

pore water in wetlands constructed with an organic substrate characteristically has pH 6-8 and is highly alkaline (19). These alkaline conditions result, in part. from reactions involving hydrogen sulfide that result in the net generation of bicarbonate. In most wetlands (constructed and natural), surface waters are aerobic while the underlying pore waters in contact with the organic substrate are anaerobic. When sulfidic pore waters diffuse from the organic substrate inte zones that contain dissolved ferric iron, dissolved oxygen, or precipitated Fe and Mn oxides, the hydrogen sulfide can be oxidized.

Table 1 shows the chemistry of surface water and substrate pore water samples collected from a wetland constructed with limestone and spent mushroom compost. Spent mushroom compost consists of a composted mixture of spoiled hay, horse manure, corn cobs, wood chips, and limestone. At the wetland used in this examples, 10 to 15 cm of limestone sand was covered with 20 to 50 cm of compost and planted with cattails. Water flowed through the wetland primarily by surface paths, and no efforts were made to force the water through the compost. This design is typical of many compost wetlands constructed in northern Appalachia during the late 1980's. The data shown in table 1 were collected 15 months after the wetland was constructed.

Surface water at the study site had low pH and high concentrations of iron. aluminum and manganese. Compared to the surface water, the substrate pore water

Surface and pore water chemistry at the Latrobe wetland (16)						
5,	Pore water ¹		Surface water ²			
Parameter	Mean	Std dev	Mean	Std dew		
A.1	1	5	35	5		
Al	467	188	308	29		
	215	183	73	39		
Fe ²⁺	215	9	24	16		
Fe ³⁺	27	.75	<1	0		
H_2S	175	48	166	9		
Mg	175	10	42	2		
Mn	24	10	5	1		
Na	11	522	1 967	115		
SO4	1,6/4	240	503	86		
Acidity ³	439	340	505	0		
Alkalinity	885	296	502	Nan		
Net Alkalinity ⁴	446	NAp	-503	1 Nap		
рН	6.8	.8	3.1	.1		

TABLE 1		
Constant none water chemistry at th	e Latrohe	wetland (16)

Nap - Not applicable

Std dev - standard deviation

¹A total of 52 water samples were collected by the dialysis tube method. Metals were analyzed for every sample. Field pH was measured for 29 samples. Alkalinity was measured for nine samples.

²Six samples collected.

³Calculated for pore water samples and measured by the H₂O₂ method for surface water samples. ⁴Average alkalinity minus average acidity.

had higher pH, higher concentrations of alkalinity, ferrous iron, calcium, and hydrogen sulfide, and lower concentrations of sulfate, ferric iron and aluminum. On average, the pore water had a net alkalinity while the surface water had a net acidity. The alkalinity of the pore water resulted from a combination of limestone dissolution and sulfate reduction. The average alkalinity calculated to result from these processes was 703 mg•L⁻¹, a value that corresponds well with the measured difference in acidity, 684 mg•L⁻¹. Compared to surface water, substrate pore water contained elevated concentrations of ferrous iron. High concentrations of Fe²⁺ likely resulted from the dissolution of ferric oxyhydroxide at the redox boundary (16).

The solubilization of ferric hydroxides results in the release of OH, which acts to raise pH to circumneutral levels and also reacts with dissolved CO2 to form bicarbonate. However, reduction of ferric hydroxide has no effect on the net acidity of the mine water because the increase in alkalinity is exactly matched by an increase in mineral acidity. If the iron-enriched pore water diffuses into an aerobic zone, the ferrous iron content should oxidize, hydrolyze, and reprecipitate as ferric oxyhydroxide. Because the pore water has circumneutral pH and is strongly buffered by bicarbonate, the removal of iron by oxidation processes from pore water as it diffuses into aerobic surface waters should occur rapidly. Indeed, during the summer months, when the data in table 1 were collected, comparisons of the wetland influent and effluent indicated that the wetland decreased concentrations of both iron and total acidity on every sampling day. The decrease in acidity indicates that alkaline pore water was mixing with surface water and neutralizing acidity. The decrease in concentrations of iron in the surface water indicates that elevated concentrations of Fe2+ observed in the pore water were rapidly removed in the surface water environment.

Constructing Wetlands to Treat Coal Mine Drainage

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Although many of the wetlands constructed to treat mine water also serve as a habitat for local wildlife, these systems are first and foremost water treatment systems, and are principally designed to utilize the previously discussed mechanisms as efficiently as possible. To accomplish this, both flow rate and water quality must be determined prior to construction. Since both may vary seasonally and during storm events, sampling and flow measurements must reflect the anticipated range of contaminant loading (flow x concentration), so that the system can be appropriately engineered.

Two sets of sizing criteria are provided (Table 2), based on the load of contaminant to be removed daily. The "abandoned mined land (AML) criteria" are intended for agencies or groups that are attempting to cost-effectively decrease

¹ The difference between surface and pore water concentrations of sulfate averaged 293 mg•L⁻¹, which is equivalent to 305 mg•L⁻¹ CaCO₃ alkalinity; the difference in calcium concentrations averaged 159 mg•L⁻¹, which is equivalent to 398 mg•L⁻¹ CaCO₃ alkalinity (16).

contaminant concentrations. In many AML situations, the goal is to improve water quality, not consistently achieve a specific effluent concentration. The AML sizing criteria are based on measurements of contaminant removal by existing constructed wetlands.

In contrast, at sites where effluent requirements must be met consistently, it is wise to double the size of the wetland to allow for seasonal variations in effectiveness and reduced efficiency observed at the lower end of the wetland where the contaminant concentrations are low (20).

The simplest case is when the mine water to be treated is net alkaline. At these sites, wetlands are constructed to maximize air-water interchange, with small waterfalls followed by shallow quiescent areas. The wetland area must be large enough to allow oxidation, precipitation and settling of the suspended floc. It is recommended that the freeboard of aerobic wetlands constructed for the removal of Fe be at least 1 m. Observations of sludge accumulation in existing wetlands suggest that a 1-m freeboard should be adequate to contain 20 to 25 years of FeOOH accumulation. If iron and manganese must both be removed, the wetland must be sized to treat them sequentially. If the pH is above 6 and the iron has been removed, manganese will precipitate on rock surfaces in shallow ponds or basins constructed for air-water interchange.

In these aerobic passive treatment systems, plant activity is relatively irrelevant to the rate of metal removal. Basins are planted to prevent flow channelization. facilitate the filtration of particulates and to benefit wildlife.

Treatment of acidic mine water requires the generation of enough alkalinity to neutralize the excess acidity. Currently, there are two passive methods for generating alkalinity: pretreatment of acidic drainage by use of an ALD or construction of a compost wetland. In some cases, the combination of an ALD *and* a compost wetland may be necessary to treat the mine wate.

ALDs produce alkalinity at a lower cost than do compost wetlands. However, not all water is suitable for pretreatment with ALDs. The primary chemical factors that limit the utility of ALDs are the presence of Fe³⁺, AL³⁺ and dissolved oxygen (DO). When acidic water containing Fe³⁺ or Al³⁺ contacts limestone, metal hydroxides (FeOOH or Al(OH)₃) will form. No oxygen is necessary. Ferric hydroxide can armor the limestone, limiting its further dissolution. Whether aluminum

TABLE 2. Recommended sizing for passive treatment systems (16)						
	AML Criteria g·m ⁻² ·d ⁻¹		Compliance Criteria g·m ⁻² ·d ⁻¹			
	Alkaline	Acid	Alkaline	Acid		
Fe Mn Acidity	20 1.0 NAp	NAp NAp 7	10 0.5 NAp	NAp NAp 3.5		

NAp - Not applicable.

hydroxides armor limestone has not been determined. The buildup of both precipitates within the ALD eventually decrease the drain permeability and cause plugging. The presence of dissolved oxygen in mine water will promote the oxidation of ferrous iron to ferric iron within the ALD, and thus potentially cause armoring and plugging. While the short-term performance of ALDs that receive water containing elevated levels of Fe³⁺, Al³⁺ or DO can be spectacular (total removal of the metals within the ALD) (21), the long-term performance of these ALDs is questionable.

Mine water that contains very low concentrations of DO, Fe^{3+} and Al (all <1 mg•L⁻¹) is ideally suited for pretreatment with an ALD. As concentrations of these parameters rise above 1 mg•L⁻¹, the risk that the ALD will fail prematurely also increases. Recently, two ALDs constructed to treat mine water that contained 20-30 mg•L⁻¹ of Al became plugged after only 6-8 months of operation.

In some cases, the suitability of mine water for pretreatment with an ALD can be evaluated based on the type of discharge and measurements of field pH. Mine waters that seep from spoils and flooded underground mines and have a field pH>5 characteristically have concentrations of DO, Fe³⁺, and Al that are all less than 1 mg•L-1. Such sites are generally good candidates for pretreatment with an ALD. Mine waters that discharge from open drift mines or have pH<5 must be analyzed for Fe³⁺ and Al. Mine waters with pH<5 can contain dissolved Al; mine waters with pH < 3.5 can contain Fe³⁺. In northern Appalachia, most mine drainages that have pH < 3 contain high concentrations of Fe³⁺ and AL.

When an ALD operates ideally, its only effect on water quality is to raise the pH to circumneutral levels and increase concentrations of calcium and alkalinity. Dissolved Fe and Mn should be unaffected by flow through the ALD. The ALD must be followed by a settling basin and a wetland system in which metal oxidation, hydrolysis and precipitation can occur. The type of post-ALD wetland system depends on the acidity of the mine water and the amount of alkalinity generated by the ALD. If the ALD generates enough alkalinity to transform the AMD to a net alkaline condition, then the ALD effluent can be treated with a settling basin and an aerobic wetland. If possible, the water should be aerated as soon as it exits the ALD and directed into a settling pond. An aerobic wetland should follow the settling pond. The total post-ALD system should be sized according to the criteria provided earlier for net alkaline mine water. At this time, it appears that mine waters with acidities less than 150 mg•L⁻¹ are readily treated with an ALD and aerobic wetland system.

When mine water contains DO, Fe^{3+} or Al, or contains concentrations of acidity > 300 mg·L⁻¹, construction of a compost wetland is recommended (16). Compost wetlands generate alkalinity through a combination of bacterial activity and limestone dissolution. The desired sulfate-reducing bacteria require a rich organic substrate in which anoxic conditions will develop. Limestone dissolution also occurs readily within this anoxic environment. A substance commonly used in these wetlands is spent mushroom compost, a substrate that is readily available in western Pennsylvania. However, any well-composted equivalent should serve

as a good bacterial substrate. Spent mushroom compost contains about 10%; CaCO₃; mixing in more limestone can increase the duration of the alkalinity generated by CaCO₃ dissolution. Compost substrates that do not have a high CaCO₃ content should be supplemented with limestone. The compost depth used in most wetlands is 30-45 cm. Typically, a ton of compost will cover about 3.5 square yards about 45 cm thick. Cattails or other emergent vegetation are planted in the substrate to stabilize it and to provide additional organic matter to "fuel" the sulfate reduction process. As a practical tip, cattail plant/rhizomes should be planted well into the substrate prior to flooding the wetland cell (16).

Compost wetlands in which water flows on the surface of the compost remove acidity (e.g. generate alkalinity) at rates of approximately 2-12 g·m²·d⁻¹. This range in performance is largely a result of seasonal variation: lower rates of acidity removal occur in winter than in summer (21). Such wetlands should be sized based on the removal rates in Table 2. For an AML site, the minimum wetland size (m²) = Acidity Loading (g·d⁻¹/7). (16)

In many wetland systems, the compost cells are preceded with a single aerobic pond in which Fe oxidation and precipitation occur. This feature is useful where the influent to the wetland is of circumneutral pH (either naturally or because of pretreatment with an ALD), and rapid, significant removal of Fe is expected as soon as the mine water is aerated. Aerobic ponds are not useful when the water entering the wetland system has a pH < 4. At such low pH, Fe oxidation and precipitation reactions are quite slow and significant removal of Fe in the aerobic pond would not be expected.

Recent research suggests that contaminant removal rates can be significantly improved in compost systemsdesigned to facilitate vertical flow. Limestone placed beneath the compost facilitates flow to a subsurface drainage effluent point, and neutralizes acidity as an ALD would, Hedin et al. (22) report an acidity removal rate of 15.4 g \cdot m² \cdot d¹ at one such system, although manganese was not removed. By maintaining at least a meter of free-standing water above the compost, and by periodically lowering the water level to flush the system, high aluminium concentrations can be flushed, and acidity removal rate of 30-50 g \cdot m² \cdot d⁻¹ are possible (23-25) An aerobic system, as previously described, can be used to remove manganese once the acidity is neutralized and iron is removed.

These systems can be constructed in series: subsurface flow compost wetlands neutralize acidity followed by iron precipitation in aerobic wetlands followed by subsurface flow compost wetlands, etc. These have been referred to as successive alkalinity-producing systems or SAPS (23). Once again, manganese, if a problem, would be removed in a final aerobic basin.

DISCUSSION/CONCLUSION

Constructing wetlands for the passive treatment of CMD has developed from an experimental concept into a well-demonstrated technology. During the last 10 years, over a thousand wetland systems have been created at or near mine sites worldwide. These systems provide relatively inexpensive water treatment and, if constructed and maintained properly, should continue to do so for 20-40 years.

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Wetlands constructed to treat CMD can have other beneficial aspects (i.e. wildlife use, water storage, etc.) but at least initially, most tend to be monocultures of *Typha* planted neatly in rows within rectilinear basins. Obviously, they do not have to be constructed in this manner. We strongly encourage wetland designers to build wetlands of varying depths and irregular boundaries, and to encourage the growth of volunteer wetland vegetation by incorporating transplanted plots of vegetation from other sites. However, it has been encouraging to see that even at sites that were well "engineered," volunteer vegetation has become established and wildlife use has been limited principally by water quality. Unfortunately, wildlife that would destroy the wetland must be discouraged; beavers and muskrats can be very troublesome, if not planned for (27).

It is also important to observe that not all efforts to passively treat mine water have been successful. In most cases, failures occur due to undersizing, either because published guidelines were not followed or the contaminant loading was underestimated (24).

It is also important to note that wetlands constructed to improve water quality should not be considered a replacement for natural wetlands. Replacement wetland construction is a rapidly advancing field, with its own criteria for success and failure. Passive treatment systems can complement such activities but should not substitute for them.

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ECOLOGY OF WETLANDS AND ASSOCIATED SYSTEMS

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