The highest rates of acidity removal, sulfate reduction, and limestone dissolution all occurred at the Howe-Lower site. This system differs from the others by its subsurface flow system. Drainage pipes, buried in the limestone that underlies the compost, cause the mine water to flow directly through the substrate. At the Somerset, Latrobe, REM, and FH systems, water flows surficially through the wetlands. Mixing of the acidic surface water and alkaline substrate waters presumably occurs by diffusion processes at the surface-flow sites. By directly contacting contaminated water and alkaline substrate, the Howe-Lower site is extracting alkalinity from the substrate at a significantly higher rate than occurs in surface flow systems. How long the Howe-Upper system can continue to generate alkalinity at the present rates is unknown. Monitoring of the system, currently in its third year of operation, is continuing.

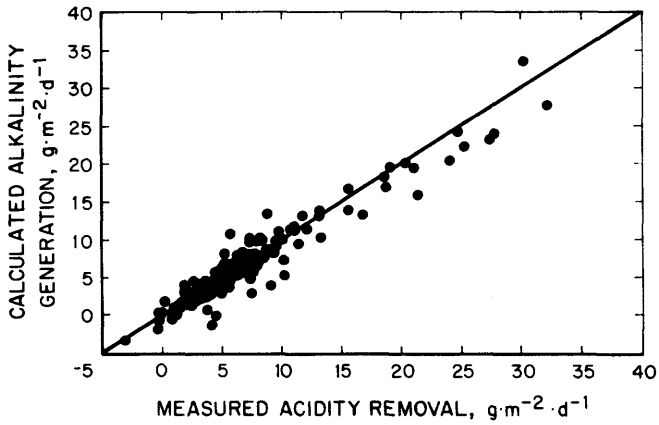

Figure 11.—Measured rates of alkalinity generation and acidity removal at the Friendship Hill wetland. Units are $g \cdot m^{-2} \cdot d^{-1}$ $CaCO_3$ equivalent.

CHAPTER 4. DESIGN AND SIZING OF PASSIVE TREATMENT SYSTEMS

Three principal types of passive technologies currently exist for the treatment of coal mine drainage: aerobic wetland systems, wetlands that contain an organic substrate, and anoxic limestone drains. In aerobic wetland systems, oxidation reactions occur and metals precipitate primarily as oxides and hydroxides. Most aerobic wetlands contain cattails growing in a clay or spoil substrate. However, plantless systems have also been constructed and at least in the case of alkaline influent water, function similarly to those containing plants (chapter 3).

Wetlands that contain an organic substrate are similar to aerobic wetlands in form, but also contain a thick layer of organic substrate. This substrate promotes chemical and microbial processes that generate alkalinity and neutralize acidic components of mine drainage. The term "compost wetland" is often used in this report to describe any constructed wetland that contains an organic substrate in which biological alkalinity-generating processes occur. Typical substrates used in these wetlands include spent mushroom compost, *Sphagnum* peat, haybales, and manure.

The ALD is a buried bed of limestone that is intended to add alkalinity to the mine water (*15, 33-34*). The limestone and mine water are kept anoxic so that dissolution can occur without armoring of limestone by ferric oxyhydroxides. ALD's are only intended to generate alkalinity, and must be followed by an aerobic system in which metals are removed through oxidation and hydrolysis reactions.

Each of the three passive technologies is most appropriate for a particular type of mine water problem. Often, they are most effectively used in combination with each other. In this chapter, a model is presented that is useful in deciding whether a mine water problem is suited to passive treatment, and also, in designing effective passive treatment systems.

Two sets of sizing criteria are provided (table 19). The "abandoned mined land (AML) criteria" are intended for groups that are attempting to cost-effectively decrease 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 (chapter 3). Most of the removal rates were measured for treatment systems (or parts of treatment systems) that did not consistently lower concentrations of contaminants to compliance with OSM effluent standards. In particular, the Fe sizing factor for alkaline mine water (20 $g \cdot m^{-2} \cdot d^{-1}$) is based on data from six sites, only one of which lowers Fe concentrations to compliance.

Table 19.—Recommended sizing for passive treatment systems

	AML criteria, $g \cdot m^{-2} \cdot d^{-1}$		Compliance criteria, $g \cdot m^{-2} \cdot d^{-1}$	
	Alkaline	Acid	Alkaline	Acid
Fe	20	NAp	10	NAp
Mn	1.0	NAp	0.5	NAp
Acidity ..	NAp	7	NAp	3.5

NAp Not applicable.

It is possible that Fe removal rates are a function of Fe concentration; i.e., as concentrations get lower, the size of

system necessary to remove a unit of Fe contamination (e.g., 1 $g \cdot d^{-1}$) gets larger. To account for this possibility, a more conservative sizing value for systems where the effluent must meet regulatory guidelines was provided (table 1). These are referred to as "compliance criteria." The sizing value for Fe, 10 $g \cdot m^{-2} \cdot d^{-1}$, is in agreement with the findings of Stark (*17*) for a constructed compost wetland in Ohio that receives marginally acidic water. This rate is larger, by a factor of 2, than the Fe removal rate reported by Brodie (*18*) for aerobic systems in southern Appalachia that are regularly in compliance.

The Mn removal rate used for compliance, 0.5 $g \cdot m^{-2} \cdot d^{-1}$, is based on the performance of five treatment systems, three of which consistently lower Mn concentrations to compliance levels. A higher removal value, 1 $g \cdot m^{-2} \cdot d^{-1}$, is suggested for AML sites. Because the toxic effects of Mn at moderate concentrations (<50 $mg \cdot L^{-1}$) are generally not significant, except in very soft water (*54*), and the size of wetland necessary to treat Mn-contaminated water is so large, AML sites with Fe problems should receive a higher priority than those with only Mn problems.

The acidity removal rate presented for compost wetlands is influenced by seasonal variations that cannot currently be corrected with wetland design (*55*). This is not a problem for mildly acidic water, where the wetland can be sized in accordance with winter performance, nor should it be a major problem in warmer climates. In northern Appalachia, however, no compost wetland that consistently transforms highly acidic water (>300 $mg \cdot L^{-1}$ acidity) into alkaline water is known. One of the study sites, which receives water with an average of 600 $mg \cdot L^{-1}$ acidity and does not need to meet a Mn standard, has discharged water that only required chemical treatment during winter months. While considerable cost savings are realized at the site because of the compost wetland, the passive system must be supported by conventional treatment during a portion of the year.

Because long-term metal-removal capabilities of passive treatment systems are currently uncertain, current Federal regulations require that the capability for chemical treatment exist at all bonded sites. This provision is usually met by placing a "polishing pond" after the passive treatment system. The design and sizing model does not currently account for such a polishing pond.

All passive treatment systems constructed at active sites need not be sized according to the compliance criteria provided in table 19. Sizing becomes a question of balancing available space and system construction costs versus influent water quality and chemical treatment costs. Mine water can be treated passively before the water enters a chemical treatment system to reduce water treatment costs or as a potential part-time alternative to full-time chemical treatment. In those cases where both passive and chemical treatment methodologies are utilized, many operators find that they recoup the cost of the passive treatment system in less than a year by using simpler, less expensive chemical treatment systems and/or by decreasing the amount of chemicals used.

A flow chart that summarizes the design and sizing model is shown in figure 12. The model uses mine drainage chemistry to determine system design, and contaminant loadings combined with the expected removal rates in table 19 to define system size. The following text details the use of this flow chart and also discusses aspects of the model that are currently under investigation.

CHARACTERIZATION OF MINE DRAINAGE DISCHARGES

To design and construct an effluent treatment system, the mine water must be characterized. An accurate measurement of the flow rate of the mine discharge or seep is required. Water samples should be collected at the discharge or seepage point for chemical analysis. Initial water analyses should include pH, alkalinity, Fe, Mn, and hot acidity (H_2O_2 method) measurements. If an anoxic limestone drain is being considered, the acidified sample should be analyzed for Fe^{3+} and Al, and a field measurement of dissolved oxygen should be made.

Both the flow rate and chemical composition of a discharge can vary seasonally and in response to storm

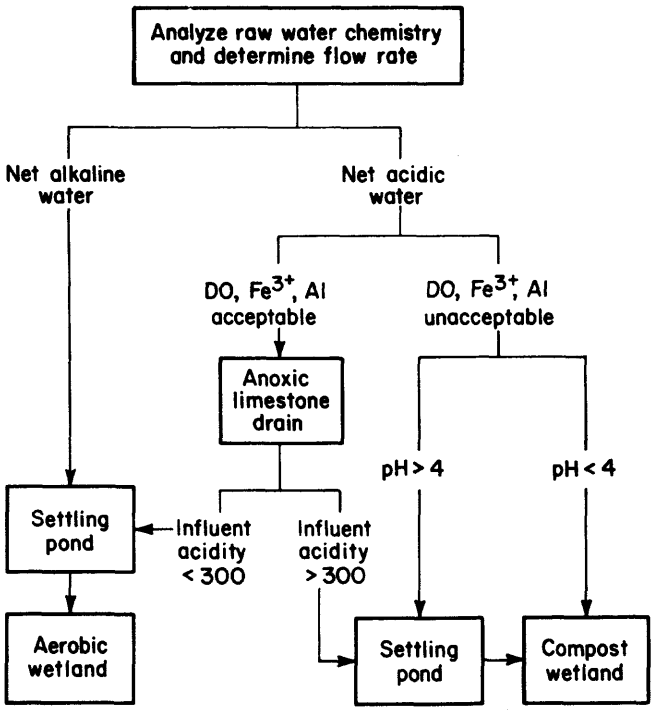

Figure 12.—Flow chart showing chemical determinations necessary for the design of passive treatment systems.

events. If the passive treatment system is expected to be operative during all weather conditions, then the discharge flow rates and water quality should be measured in different seasons and under representative weather conditions.

CALCULATIONS OF CONTAMINANT LOADINGS

The size of the passive treatment system depends on the loading rate of contaminants. Calculate contaminant (Fe, Mn, acidity) loads by multiplying contaminant concentrations by the flow rate. If the concentrations are milligrams per liter and flow rates are liters per minute, the calculation is

$$[\text{Fe,Mn,Acidity}]\ \text{g} \cdot \text{d}^{-1} = \text{flow} \times [\text{Fe,Mn,Acidity}] \times 1.44 \quad (11)$$

If the concentrations are milligrams per liter and flow rates are gallons per minute, the calculation is

$$[\text{Fe,Mn,Acidity}]\ \text{g} \cdot \text{d}^{-1} = \text{flow} \times [\text{Fe,Mn,Acidity}] \times 5.45 \quad (12)$$

Calculate loadings for average data and for those days when flows and contaminant concentrations are highest.

CLASSIFICATION OF DISCHARGES

The design of the passive treatment system depends largely on whether the mine water is acidic or alkaline. One can classify the water by comparing concentrations of acidity and alkalinity.

Net Alkaline Water: alkalinity > acidity
Net Acidic Water: acidity > alkalinity

The successful treatment of mine waters with net acidities of 0 to 100 $mg \cdot L^{-1}$ using aerobic wetlands has been documented in this report and elsewhere (*14, 18*). In these systems, alkalinity either enters the treatment system with diluting water or alkalinity is generated within the system by undetermined processes. Currently, there is no method to predict which of these marginally acidic waters can be treated successfully with an aerobic system only. For waters with a net acidity >0, the incorporation of alkalinity-generating features (either an ALD or a compost wetland) is appropriate.

PASSIVE TREATMENT OF NET ALKALINE WATER

Net alkaline water contains enough alkalinity to buffer the acidity produced by metal hydrolysis reactions. The metal contaminants (Fe and Mn) will precipitate given enough time. The generation of additional alkalinity is unnecessary so incorporation of limestone or an organic substrate into the passive treatment system is also unnecessary. The goal of the treatment system is to aerate the water and promote metal oxidation processes. In many existing treatment systems where the water is net alkaline, the removal of Fe appears to be limited by dissolved O_2 concentrations. Standard features that can aerate the drainage, such as waterfalls or steps, should be followed by quiescent areas. Aeration only provides enough dissolved O_2 to oxidize about 50 $mg \cdot L^{-1}$ Fe^{2+}. Mine drainage with higher concentrations of Fe^{2+} will require a series of aeration structures and wetland basins. The wetland cells allow time for Fe oxidation and hydrolysis to occur and space in which the Fe floc can settle out of suspension. The entire system can be sized based on the Fe removal rates shown in table 19. For example, a system being designed to improve water quality on an AML site should be sized by the following calculation:

$$\text{Minimum wetland size } (\text{m}^2) = \text{Fe loading } (\text{g} \cdot \text{d}^{-1}) / 20\ (\text{g} \cdot \text{m}^{-2} \cdot \text{d}^{-1}). \quad (13)$$

If Mn removal is desired, size the system based on the Mn removal rates in table 19. Removal of Fe and Mn occurs sequentially in passive systems. If both Fe and Mn removal are necessary, add the two wetland sizes together.

A typical aerobic wetland is constructed by planting cattail rhizomes in soil or alkaline spoil obtained on-site. Some systems have been planted by simply spreading cattail seeds, with good plant growth attained after 2 years. The depth of the water in a typical aerobic system is 10 to 50 cm. Ideally, a cell should not be of uniform depth, but should include shallow and deep marsh areas and a few deep (1 to 2 m) spots. Most readily available aquatic vegetation cannot tolerate water depths greater than 50 cm.

Often, several wetland cells are connected by flow through a V-notch weir, lined railroad tie steps, or down a ditch. Spillways should be designed to pass the maximum probable flow. Spillways should consist of wide cuts in the dike with side slopes no steeper than 2H:1V, lined with nonbiodegradable erosion control fabric, and coarse rip rap if high flows are expected (*18*). Proper spillway design can preclude future maintenance costs because of erosion and/or failed dikes. If pipes are used, small diameter (<30 cm) pipes should be avoided because they can plug with litter and FeOOH deposits. Pipes should be made of polyvinyl chloride (PVC). More details on the construction of aerobic wetland systems can be found in a text by Hammer (*56*).

The geometry of the wetland site as well as flow control and water treatment considerations may dictate the

use of multiple wetland cells. The intercell connections may also serve as aeration devices. If there are elevation differences between the cells, the interconnection should dissipate kinetic energy and be designed to avoid erosion and/or the mobilization of precipitates.

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.

The floor of the wetland cell may be sloped up to about 3% grade. If a level cell floor is used, then the water level and flow are controlled by the downstream dam spillway and/or adjustable riser pipes.

As discussed in chapter 3, some of the aerobic systems that have been constructed to treat alkaline mine water have little emergent plant growth. Metal removal rates in these plantless, aerobic systems appears to be similar to what is observed in aerobic systems containing plants. However, plants may provide values that are not reflected in measurements of contaminant removal rates. For example, plants can facilitate the filtration of particulates, prevent flow channelization and provide wildlife benefits that are valued by regulatory and environmental groups.

PASSIVE TREATMENT OF NET ACID WATER

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: construction of a compost wetland or pretreatment of acidic drainage by use of an ALD. In some cases, the combination of an ALD and a compost wetland may be necessary to treat the mine water.

ALD's produce alkalinity at a lower cost than do compost wetlands. However, not all water is suitable for pretreatment with ALD's. The primary chemical factors believed to limit the utility of ALD's are the presence of ferric iron (Fe^{3+}), aluminum (Al) and dissolved oxygen (DO). When acidic water containing any Fe^{3+} or Al contacts limestone, metal hydroxide particulates (FeOOH or $Al(OH)_3$) will form. No oxygen is necessary. Ferric hydroxide can armor the limestone, limiting its further dissolution. Whether aluminum hydroxides armor limestone has not been determined. The buildup of both precipitates within the ALD can 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 ALD's that receive water containing elevated levels of Fe^{3+}, Al, or DO can be spectacular (total removal of the metals within the ALD) (*34*), the long-term performance of these ALD's 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 ALD's constructed to treat mine water that contained 20 mg•L^{-1} Al became plugged after 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, Fe^{3+}, and Al that are all <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 Fe^{3+} and Al. Mine waters with pH <5 can contain dissolved Al; mine waters with pH <3.5 can contain Fe^{3+}. In northern Appalachia, most mine drainages that have pH <3 contain high concentrations of Fe^{3+} and Al.

PRETREATMENT OF ACIDIC WATER WITH ALD

In an ALD, alkalinity is produced when the acidic water contacts the limestone in an anoxic, closed environment. It is important to use limestone with a high $CaCO_3$ content because of its higher reactivity compared with a limestone with a high $MgCO_3$ or $CaMg(CO_3)_2$ content. The limestones used in most successful ALD's have 80% to 95% $CaCO_3$ content. Most effective systems have used number 3 or 4 (baseball-size) limestone. Some systems constructed with limestone fines and small gravel have failed, apparently because of plugging problems. The ALD must be sealed so that inputs of atmospheric oxygen are minimized and the accumulation of CO_2 within the ALD is maximized. This is usually accomplished by burying the ALD under several feet of clay. Plastic is commonly placed between the limestone and clay as an additional gas barrier. In some cases, the ALD has been completely wrapped in plastic before burial (*35*). The ALD should be designed so that the limestone is inundated with water at all times. Clay dikes within the ALD or riser pipes at the outflow of the ALD will help ensure inundation.

The dimensions of existing ALD's vary considerably. Most older ALD's were constructed as long narrow drains, approximately 0.6 to 1.0 m wide. A longitudinal section and cross section of such an ALD is shown in figure 13. The ALD shown was constructed in October 1990, and is 1 m wide, 46 m long and contains about 1 m depth of number 4 limestone. The limestone was covered with two layers of 5 mil plastic, which in turn was covered with

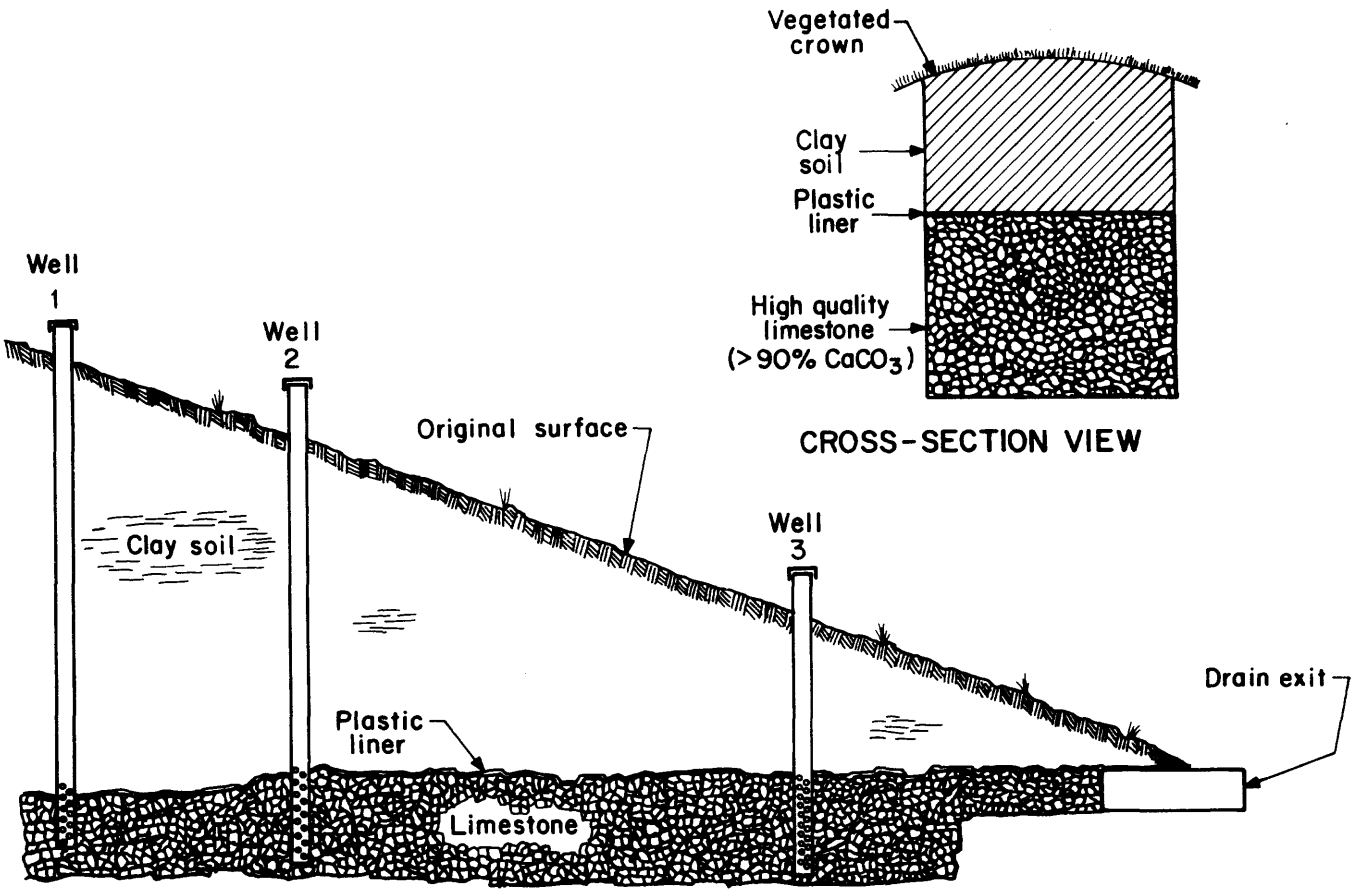

Figure 13.—Longitudinal-section and cross-section of the Morrison ALD. Wells are for sampling purposes and have no importance to drain's functioning.

0.3 to 3 m of on-site clay to restore the original surface topography (*34, 36*).

At sites where linear ALD's are not possible, anoxic limestone beds have been constructed that are 10 to 20 m wide. These bed systems have produced alkalinity concentrations similar to those produced by the more conventional drain systems.

The mass of limestone required to neutralize a certain discharge for a specified period can be readily calculated from the mine water flow rate and assumptions about the ALD's alkalinity-generating performance. Recent USBM research indicates that approximately 14 h of contact time between mine water and limestone in an ALD is necessary to achieve a maximum concentration of alkalinity (*57*). To achieve 14 h of contact time within an ALD, ~3,000 kg of limestone rock is required for each liter per minute of mine water flow. An ALD that produces 275 $mg \cdot L^{-1}$ of alkalinity (the maximum sustained concentration thus far observed for an ALD), dissolves ~1,600 kg of limestone a decade per each liter per minute of mine water flow. To construct an ALD that contains sufficient limestone to insure a 14-h retention time throughout a 30-yr period, the limestone bed should contain ~7,800 kg of limestone for each liter per minute of flow. This is equivalent to 30 tons of limestone for each gallon per minute of flow. The calculation assumes that the ALD is constructed with 90% $CaCO_3$ limestone rock that has a porosity of 50%. The calculation also assumes that the original mine water does not contain ferric iron or aluminum. The presence of these ions would result in potential problems with armoring and plugging, as previously discussed.

Because the oldest ALD's are only 3 to 4 yr old, it is difficult to assess how realistic these theoretical calculations are. Questions about the ability of ALD's to maintain unchannelized flow for a prolonged period, whether 100% of the $CaCO_3$ content of the limestone can be expected to dissolve, whether the ALD's will collapse after significant dissolution of the limestone, and whether inputs of DO that are not generally detectable with standard field equipment (0 to 1 $mg \cdot L^{-1}$) might eventually result in armoring of the limestone with ferric hydroxides, have not yet been addressed.

The anoxic limestone drain is <u>one</u> component of a passive treatment system. When the ALD operates ideally, its only effect on mine water chemistry is to raise pH to

circumneutral levels and increase concentrations of calcium and alkalinity. Dissolved Fe^{2+} and Mn should be unaffected by flow through the ALD. The ALD must be followed by a settling basin or wetland system in which metal oxidation, hydrolysis and precipitation can occur. The type of post-ALD treatment 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 acid mine drainage to a net alkaline condition, then the ALD effluent can then 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 <150 mg•L^{-1} are readily treated with an ALD and aerobic wetland system.

If the mine water is contaminated with only Fe^{2+} and Mn, and the acidity exceeds 300 mg•L^{-1}, it is unlikely that an ALD constructed using current practices will dis-charge net alkaline water. When this partially neutralized water is treated aerobically, the Fe will precipitate rapidly, but the absence of sufficient bufferring can result in a discharge with low pH. Building a second ALD, to recharge the mine water with additional alkalinity after it flows out of the aerobic system, is currently not feasible because of the high DO content of water flowing out of aerobic systems. If the treatment goal is to neutralize all of the acidity passively, then a compost wetland should be built so that additional alkalinity can be generated. Such a treatment system thus contains all three passive technologies. The mine water flows through an ALD, into a settling pond and an aerobic system, and then into a compost wetland.

If the mine water is contaminated with ferric iron (Fe^{3+}) or Al, higher concentrations of acidity can be treated with an ALD than when the water is contaminated with only Fe^{2+} and Mn. This enhanced performance results from a decrease in mineral acidity because of the hydrolysis and precipitation of Fe^{3+} and Al within the ALD. These metal-removing reactions decrease the mineral acidity of the water. ALD's constructed to treat mine water contaminated with Fe^{3+} and Al and having acidities greater than 1,000 mg•L^{-1} have discharged net alkaline water. The long-term prognosis for these metal-retaining systems has been questioned (*34*). However, even if calculations of system longevity (as described above) are inaccurate for waters contaminated with Fe^{3+} and Al, their treatment with an ALD may turn out to be cost-effective when compared with chemical alternatives (*35*).

When a mine water is contaminated with Fe^{2+} and Mn and has an acidity betweem 150 and 300 mg•L^{-1}, the ability of an ALD to discharge net alkaline water will depend on the concentration of alkalinity produced by the limestone system. The amount of alkalinity generated by a properly constructed and sized ALD is dependent on chemical characteristics of the acid mine water. An experimental method has been developed that results in an accurate assessment of the amount of alkalinity that will be generated when a particular mine water contacts a particular limestone (*58*). The method involves the anoxic incubation of the mine water in a container filled with limestone gravel. In experiments at two sites, the concentration of alkalinity that developed in these containers after 48 h correlated well with the concentrations of alkalinity measured in the ALD effluents at both sites.

TREATING MINE WATER WITH COMPOST WETLAND

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. 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 has a high $CaCO_3$ content (about 10% dry weight), but mixing in more limestone may increase the alkalinity generated by $CaCO_3$ dissolution. Compost substrates that do not have a high $CaCO_3$ content should be supplemented with limestone. The compost depth used in most wetlands is 30 to 45 cm. Typically, a metric ton of compost will cover about 3.5 m^2 to a depth of 45 cm thick. This is equivalent to one ton per 3.5 yd^2. 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.

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•m^{-2}•d^{-1}. This range in performance is largely a result of seasonal variation: lower rates of acidity removal occur in winter than in summer (*55*). Research in progress indicates that supplementing the compost with limestone and incorporating system designs that cause most of the water to flow through the

compost (as opposed to on the surface) may result in higher rates of limestone dissolution and better winter performance.

Compost wetlands should be sized based on the removal rates in table 19. For an AML site, the calculation is

$$\text{Minimum Wetland Size } (m^2) = \text{Acidity Loading } (g \cdot d^{-1}/7). \quad (14)$$

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.

OPERATION AND MAINTENANCE

Operational problems with passive treatment systems can be attributed to inadequate design, unrealistic expectations, pests, inadequate construction methods, or natural problems. If properly designed and constructed, a passive treatment system can be operated with a minimum amount of attention and money.

Probably the most common maintenance problem is dike and spillway stability. Reworking slopes, rebuilding spillways, and increasing freeboard can all be avoided by proper design and construction using existing guidelines for such construction.

Pests can plague wetlands with operational problems. Muskrats will burrow into dikes, causing leakage and potentially catastrophic failure problems, and will uproot significant amounts of cattails and other aquatic vegetation. Muskrats can be discouraged by lining dike inslopes with chainlink fence or riprap to prevent burrowing (*13*). Beavers cause water level disruptions because of damming and also seriously damage vegetation. They are very difficult to control once established. Small diameter pipes traversing wide spillways ("three-log structure") and trapping have had limited success in beaver control. Large pipes with 90° elbows on the upstream end have been used as discharge structures in beaver-prone areas (*18*). Otherwise, shallow ponds with dikes with shallow slopes toward wide, riprapped spillways may be the best design for a beaver-infested system.

Mosquitos can be a problem where mine water is alkaline. In southern Appalachia, mosquitofish (*Gambusia affinis*) have been introduced into alkaline-water wetlands. Other insects, such as the armyworm, have devastated monocultural wetlands with their appetite for cattails (*59*). The use of a variety of plants in a system will minimize such problems.

CHAPTER 5. SUMMARY AND CONCLUSIONS

The treatment of contaminated coal mine drainage requires the precipitation of metal contaminants and the neutralization of acidity. In conventional treatment systems, distinctions between these two treatment objectives are blurred by additions of highly basic chemicals that simultaneously cause the rapid precipitation of metal contaminants and the neutralization of acidity. Passive treatment differs from conventional treatment by its distinction between these two treatment objectives. It is possible to passively precipitate Fe contaminants from mine water, but have little effect on the mine water acidity. Alternatively, it is possible to passively add neutralizing capacity to acidic mine water without decreasing metal concentrations.

Waters that contain high concentrations of bicarbonate alkalinity are most amenable to treatment with constructed wetlands. Bicarbonate acts as a buffer that neutralizes the acidity produced when Fe and Mn precipitate and maintains a pH between 5.5 and 6.5. At this circumneutral pH, Fe and Mn precipitation processes are more rapid than under acidic pH conditions. Given the ability of bicarbonate alkalinity to positively impact both the metal precipitation and neutralization aspects of mine water treatment, it is not surprising that the most noteworthy applications of passive treatment have been at sites where the mine water was net alkaline. The most successful wetlands constructed in western Pennsylvania in the early 1980's treated mine waters that contained alkalinity. All of the early successes of the TVA were, likewise, with waters that were alkaline (*13*). Similarly, the Simco wetland in Ohio, which has discharged compliance water for several years (*17*), receives water containing ~160 $mg \cdot L^{-1}$ alkalinity. In this study, the two treatment systems that met all effluent discharge requirements (Donegal and Blair) both received alkaline, metal-contaminated water.

When mine water is acidic, enough alkalinity must be generated by the passive treatment system to neutralize the acidity. The most common method used to passively generate alkalinity is the construction of a wetland that