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

contains an organic substrate in which alkalinity-generating microbial processes occur. If the substrate contains limestone, as spent mushroom compost does, then alkalinity will be generated by both calcite dissolution and bacterial sulfate reduction reactions. These alkalinity generating processes are slow relative to processes that remove Fe. Thus, the performance of the constructed wetlands that receive acidic water is usually limited by the rate at which alkalinity is generated within the substrate. While wetlands can significantly improve water quality, and have proven to be effective at moderately acidic sites, no wetland systems that consistently and completely transform highly acidic water to compliance quality are known. Inconsistent or partial treatment indicates undersizing. The authors believe this is because of a lack of awareness of how much larger wetlands constructed to treat acidic water must be than ones constructed to treat alkaline water. The Fe and acidity removal rates measured in this study indicate that the treatment of 5,000 $g \cdot d^{-1}$ of Fe in alkaline water requires ~250 m^2 of aerobic wetland. The treatment of the same Fe load in acidic water (where treatment requires both precipitation of the Fe and neutralization of the associated acidity) requires ~1,300 m^2 of compost wetland. Thus wetlands constructed to treat acidic water need to be six times larger than ones constructed to treat similarly contaminated alkaline water.

The recent development of limestone pretreatment systems, e.g., the anoxic limestone drain, is a significant advancement in passive treatment technology. When successful, ALD's can lower acidities or actually transform acidic water into alkaline water, and markedly decrease the sizing demands of the wetlands constructed to precipitate the metal contaminants. Because limestone is inexpensive, the cost of an ALD-aerobic wetland passive treatment system is typically much less than the compost wetland alternative. Thus, when the influent water is appropriate, ALD's should be the preferred method for generating alkalinity in passive treatment systems.

Anoxic limestone drains have also been used to increase the performance of existing constructed wetlands. At many poorly performing wetlands that receive acidic water, the wetland was built too small to treat an acidic, metal-contaminated influent, but is large enough for an alkaline, metal-contaminated influent. One of the study sites, the Morrison wetland, was undersized for the highly acidic water that it received. As a result, the wetland effluent required supplemental treatment with chemicals. Since construction of an ALD, and its addition of 275 $mg \cdot L^{-1}$ of bicarbonate alkalinity to the water, the discharge of the wetland has been alkaline, low in dissolved metals, and does not require any supplemental chemical treatment. Similar enhancements in wetland performance through the addition of ALD's have been reported elsewhere in Appalachia (*15, 18*).

KINETICS OF CONTAMINANT REMOVAL PROCESSES

This report presents an intensive analysis of contaminant removal kinetics in passive treatment systems. The rates presented are generally in agreement with those reported by other investigators. For example, the average Mn-removal rate measured in this study for alkaline, Fe-free waters, 0.5 $g \cdot m^{-2} \cdot d^{-1}$, is consistent with rates reported by the TVA for aerobic wetlands in southern Appalachia (*18*) and by the Pennsylvania Department of Environmental Resources (DER) for constructed wetlands in Pennsylvania (*60*). The average Fe-removal rate reported in this study for alkaline waters, 20 $g \cdot m^{-2} \cdot d^{-1}$, is only slightly greater than has been reported in other studies. The rates of Fe removal for aerobic wetlands in southern Appalachia ranged from 6 to 20 $g \cdot m^{-2} \cdot d^{-1}$ (*18*). Some of the lower rates reported by TVA investigators, however, are from wetland systems that discharge water with <1 $mg \cdot L^{-1}$ Fe and thus are loading limited with respect to Fe. Such sites were intentionally avoided in this study. Stark (*17*), in their studies of a constructed wetland in Ohio, reported Fe removal rates over a range of loading conditions. When the wetland system discharged >15 $mg \cdot L^{-1}$ Fe, and thus was overloaded with Fe, the removal rate averaged 21 $g \cdot m^{-2} \cdot d^{-1}$. When the wetland effluent contained <15 $mg \cdot L^{-1}$ Fe, the removal rate averaged only 11 $g \cdot m^{-2} \cdot d^{-1}$.

LONG-TERM PERFORMANCE

Passive treatment systems cannot be expected to perform indefinitely. In the long term, wetland systems will fill up with metal precipitates or the conditions that facilitate contaminant removal may be compromised. None of the treatment systems considered in this study demonstrated any downward trends in contaminant removal performance. Therefore, estimates of the long-term performance of passive systems must be made by extrapolating available data. Like the design and sizing of passive treatment systems, estimates of long-term performance vary with the chemistry of the mine water. Systems receiving alkaline water precipitate Fe and Mn contaminants by oxidative processes. The rapid removal of Fe that occurs in alkaline treatment systems means that such systems will inevitably fill up. Stark (*61*) reports that the Fe sludge in a constructed wetland in Ohio is increasing by 3 to 4 cm per year. Similar measurements at Pennsylvania wetlands indicate an increase in sludge depth of 2 to 3 cm per year (*62*). These measurements suggest that dikes that provide 1 m of freeboard should provide sufficient volume for 25 to 50 years of performance.

At some surface mines, water quality tends to improve within a decade after regrading and reclamation are

completed (*63-64*). At these surface minesites, 25 to 50 years of passive treatment may be adequate to mitigate the contaminant problem. At surface mine sites where contaminant production is continual, or at systems constructed to treat drainage from underground mines or coal refuse disposal areas, the system can either be built with greater freeboard or rebuilt when it eventually fills up. Site conditions will determine whether it is more economical to simply bury the wetland system in place and construct a new one, or to excavate and haul away the accumulated solids for proper disposal. Disposal of these excavated sludges is not difficult or unduly expensive because the material is not considered a hazardous waste.

Wetlands that receive acidic water, and function through the alkalinity-generating processes associated with an organic substrate, may decline in performance as the components of the organic substrate that generate alkalinity are exhausted. The compost wetlands described in this report neutralize acidity through the dissolution of limestone and the bacterial reduction of sulfate. Limestone dissolution is limited by the amount of limestone present in the substrate. The limestone content of spent mushroom compost is ~30 $kg \cdot m^{-3}$ (*65*). If a wetland containing a 40 cm depth of compost generates $CaCO_3$-derived alkalinity at a mean rate of 3 $g \cdot m^{-2} \cdot d^{-1}$ (the average rate measured in this study), then the limestone content of the compost will be exhausted in 11 years. The same volume of compost contains ~40 kg of organic carbon. If bacterial sulfate reduction mineralizes 100% of this carbon to bicarbonate at a rate of 5 $g \cdot m^{-2} \cdot d^{-1}$, then the carbon will be exhausted in 91 years. This estimate is increased by the carbon input of the net primary production of the wetland system, but decreased by the fact that some of the carbon is mineralized by reactions other than sulfate reduction. Studies of a salt marsh on Cape Cod, MA, indicated that 75% of the carbon was eventually mineralized by sulfate reduction processes (*66*). Another significant factor that decreases the available carbon is that a portion of the carbon pool is recalcitrant.

A realistic scenario for the long-term performance of a compost wetland is that sulfate reduction is linked, in a dependent manner, to limestone dissolution. Sulfate-reducing bacteria are inactive at pH less than 5 (*37*). Their activity in a wetland receiving lower pH water may depend, in part, on the presence of pH-buffering supplied by limestone dissolution. Thus, limestone dissolution may create alkaline zones in which sulfate reduction can proceed and produce further alkalinity. If this scenario is accurate, then the long-term performance of a compost wetland may be limited by the amount of limestone in the substrate, or according to the above calculations, about 11 years of performance. Under these conditions it would be advisable to increase the chemical buffering capability of the wetland substrate by adding additional limestone during wetland construction. In fact, this procedure is commonly practiced at many constructed compost wetland sites.

The performance of anoxic limestone drains has many aspects that make long-term expectations uncertain. Anoxic limestone drains function through the dissolution, and thus removal, of limestone. Eventually, this chemical reaction will exhaust the limestone. Long-term scenarios about ALD performance fail to consider the hydrologic implications of the gradual structural failure of the systems. In large ALD's, most of the limestone dissolution occurs in the upgradient portion of the limestone bed. It is unknown whether this preferential dissolution will produce partial failure of the integrity of the system or whether the permeability will be adversely affected. Another aspect that affects long-term ALD performance is the fact that ALD's retain ferric iron and aluminum (*34-35*). This retention has raised concerns about the armoring of limestone or the plugging of flow paths long before the limestone is exhausted by dissolution reactions (*34*). No methods are currently available to predict exactly how the retention of these metals affects the performance of ALD's.

CONTINUALLY EVOLVING PASSIVE TECHNOLOGIES

This document reports the current state of passive mine water treatment technologies. The design and sizing recommendations presented herein represent current methodologies that will subsequently be replaced with more efficient techniques. For example, important experiments are underway in Pennsylvania, Virginia, and West Virginia testing "hybrid" ALD-compost wetland systems. In these experimental systems, organic substrates are used to reduce ferric iron to ferrous iron and strip dissolved oxygen from the water so that the mine water is suitable for flow through an anoxic limestone drain. If these systems prove successful, it may be possible to treat highly acidic water by cycling it between anoxic alkalinity-generating environments and aerobic, metal-removal environments. Experimental systems using this design have recently been constructed in western Pennsylvania (*67*).

While the specific tools of passive treatment are likely to evolve in the coming years, the fundamental mechanisms of passive treatment that have been identified in this report will probably not change markedly. Research has shown that the treatment of contaminated coal mine drainage by constructed wetlands can be explained by well-known chemical and biological processes. Passive treatment, like active treatment with chemicals, requires that the metal contaminants be precipitated and that the acidity associated with these ions be neutralized. By recognizing that these treatment goals need not be accomplished

simultaneously, one can focus on optimization of the individual objectives. As a result, the performance and cost effectiveness of passive treatment systems is rapidly improving. Today, most mine operators who install properly designed passive treatment systems rapidly recoup the cost of their investment through decreased water treatment costs. There is no reason to doubt that this technology will continue to improve and that, over time, passive treatment will be used in applications that are not possible today.

REFERENCES

1. Kleinmann, R. L. P. Acid Mine Drainage. Eng. Min. J., v. 190, 1989, pp. 16I-16N.

2. Kleinmann, R. L. P., J. R. Jones, and P. M. Erickson. An Assessment of the Coal Mine Drainage Problem. Paper in the Proceedings of the 10th Annual Conference of the Association of Abandoned Mine Land Programs, PA Bur. AML Reclam., 1988, pp. 1-9.

3. Herlihy, A. T., P. R. Kaufman, M. E. Mitch, and D. Brown. Regional Estimates of Acid Mine Drainage Impact on Streams in the Mid-Atlantic and Southeastern United States. Water, Air, and Soil Pollution, v. 50, 1990, pp. 91-107.

4. Huntsman, B. E., J. G. Solch, and M. D. Porter. Utilization of *Sphagnum* Species Dominated Bog for Coal Acid Mine Drainage Abatement. Geological Society of America (91st Annual Meeting) Abstracts, Toronto, Ontario, Canada, 1978, p. 322.

5. Wieder, R. K., and G. E. Lang. Modification of Acid Mine Drainage in a Freshwater Wetland. Paper in Proceedings of the Symposium on Wetlands of the Unglaciated Appalachian Region, WV Univ., Morgantown, WV, ed. by B. R. McDonald, 1982, pp. 43-53.

6. Kleinmann, R. L. P., T. O. Tiernan, J. G. Solch, and R. L. Harris. A Low-Cost, Low-Maintenance Treatment System for Acid Mine Drainage Using *Sphagnum* Moss and Limestone. Paper in Proceedings of the 1983 Symposium on Surface Mining, Hydrology, Sedimentology and Reclamation, Univ. KY, Lexington, KY, ed. by S. B. Carpenter and R. W. DeVore, 1983, pp. 241-245.

7. Gerber, D. W., J. E. Burris, and R. W. Stone. Removal of Dissolved Iron and Manganese Ions From Water by a *Sphagnum* Moss System. Paper in Proceedings of Wetlands and Water Management on Mined Lands, Oct. 23-24, 1985, PA State Univ., University Park, PA, 1985, pp. 365-372.

8. Stone, R. W., and B. J. Pesavento. Micro- and Macro-Biological Characteristics of Wetlands Removing Iron and Manganese. Abstract in Proceedings of Wetlands and Water Management on Mined Lands, Oct. 23-24, 1985, PA State Univ., University Park, PA, 1985, p. 373.

9. Weider, R. K., G. E. Lang, and A. E. Whitehouse. Metal Removal in *Sphagnum*-Dominated Wetlands: Experience With a Man-Made Wetland System. Wetlands and Water Management on Mined Lands, PA State Univ., University Park, PA, 1985, pp. 353-364.

10. Spratt, A. K., and R. K. Wieder. Growth Responses and Iron Uptake in *Sphagnum* Plants and Their Relations to Acid Mine Drainage (AMD) Treatment. Paper in 1988 Mine Drainage and Surface Mine Reclamation, Volume I: Mine Water and Mine Waste. BuMines IC 9183, 1988, pp. 279-286.

11. Girts, M. A., R. L. P. Kleinmann, and P. M. Erickson. Performance Data on *Typha* and *Sphagnum* Wetlands Constructed To Treat Coal Mine Drainage. Paper in Proceedings of the 8th Annual Surface Mine Drainage Task Force Symposium, WV Univ., 1987.

12. Wieder, R. K. A Survey of Constructed Wetlands for Acid Coal Mine Drainage Treatment in the Eastern United States. Wetlands, v. 9, No. 2, 1989, pp. 299-315.

13. Brodie, G. A. Treatment of Acid Drainage Using Constructed Wetlands Experiences of the Tennessee Valley Authority. Paper in Proceedings of the 1990 National Symposium on Mining, Lexington, KY, ed. by D. H. Graves and R. W. DeVore, OES Publ., 1990, pp. 77-83.

14. Hedin, R. S., and R. W. Nairn. Contaminant Removal Capabilities of Wetlands Constructed To Treat Coal Mine Drainage. Chapter in Constructed Wetlands for Water Quality Improvement, ed. by G. A. Moshiri. Lewis Publ., 1993, pp. 187-195.

15. Turner, D., and D. McCoy. Anoxic Alkaline Drain Treatment System, a Low Cost Acid Mine Drainage Treatment Alternative. Paper in Proceedings of the 1990 National Symposium on Mining, Lexington, KY, ed. by D. H. Graves and R. W. DeVore, OES Publ., 1990, pp. 73-75.

16. Wieder, R. K., M. N. Linton, and K. P. Heston. Laboratory Mesorosm Studies of Fe, Al, Mn, Ca, and Mg Dynamics in Wetlands. Water, Air and Soil Pollution, v. 51, 1990, pp. 181-196.

17. Stark, L., E. Stevens, H. Webster, and W. Wenerick. Iron Loading, Efficiency, and Sizing in a Constructed Wetland Receiving Mine Drainage. Paper in Proceedings of the 1990 Mining and Reclamation Conference and Exhibition: Volume II, ed. by J. Skousen, J. Sencindiver and D. Samuel, WV Univ. Publ., Serv., 1990, pp. 393-401.

18. Brodie, G. A. Achieving Compliance With Staged Aerobic Constructed Wetlands to Treat Acid Drainage. Paper in Proceedings of the 1991 National Meeting of the American Society of Surface Mining and Reclamation, ed. by W. Oaks and J. Bowden, ASSMR, Princeton, WV, 1991, pp. 151-174.

19. Wheeler, B. D., and K. E. Giller. The Use of Dialysis Cells for Investigating Pore Water Composition in Wetland Substrata, With Particular Reference to Dissolved Iron and Sulphide. Communications in Soil Sci. Plant Anal., v. 15, 1984, pp. 707-716.

20. Fales, H. A., and F. Kenny. Inorganic Quantitative Analysis. D. Appleton Century Co., New York, 1940, 713 pp.

21. American Public Health Association. Standard Methods for the Examination of Water and Wastewater. Washington, DC, 1985, 1,268 pp.

22. Grant, D. M. Open Channel Flow Measurement Handbook. Isco, Inc., 1988, 227 pp.

23. Hem, J. D. Study and Interpretation of the Chemical Characteristics of Natural Water, 3rd ed. U.S. Geol. Surv. Water Supply Paper 2254, 1985, 263 pp.

24. Stumm, W., and J. Morgan. Aquatic Chemistry, 2d ed. Wiley, NY, 1981, 583 pp.

25. Singer, P. C., and W. Stumm. Oxygenation of Ferrous Iron. Water Poll. Cont. Res. Ser., 14010-06169, 1970, 183 pp.

26. ____. Acid Mine Drainage: The Rate-Determining Step. Science 167, 1970, pp. 1121-1123.

27. Nealson, K. H. The Microbial Iron Cycle. Microbial Geochemistry, ed. by W. E. Krumbein, Blackwell Scientific, Boston, MA, 1983, pp. 159-190.

28. Diehl, D., and W. Stumm. Is Dissolved Mn^{2+} Being Oxidized by O_2 in Absence of Mn-Bacteria or Surface Catalysts. Geochem. Cosmochim., v. 48, 1984, pp. 1571-1573.

29. Nealson, K. H. The Microbial Manganese Cycle. Microbial Geochemistry, ed. by W. E. Krumbein, Blackwell Scientific, Boston, MA, 1983, pp. 159-190.

30. Davies, S. H., and J. J. Morgan. Manganese (II) Oxidation Kinetics on Metal Oxide Surfaces. J. Colloid and Interface Sci., v. 129, 1989, pp. 63-77.