



LANDBOOK OF CONSTRUCTED WETLANDS

a guide to creating wetlands for:

AGRICULTURAL WASTEWATER

DOMESTIC WASTEWATER

COAL MINE DRAINAGE

STORMWATER

in the Mid-Atlantic Region

VOLUME 4: COAL MINE DRAINAGE

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The findings, conclusions, and recommendations contained in the Handbook do not necessarily represent the policy of the USDA - NRCS, EPA - Region III, the Commonwealth of Pennsylvania, or any other state in the northeastern United States concerning the use of constructed wetlands for the treatment and control of nonpoint sources of pollutants. Each state agency should be consulted to determine specific programs and restrictions in this regard.

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CHAPTER 1 INTRODUCTION

This volume is a general guide to the use of constructed wetlands in treating coal mine drainage in northern Appalachia. It is to be used in conjunction with Volume 1: General Considerations, which provides information on wetland hydrology, soils, and vegetation, and on the siting, design, construction, operation, and maintenance of constructed wetland systems.

If the treatment of mine drainage by constructed wetlands is to be effective, the appropriate type of wetland treatment must be chosen and the wetland must be made large enough to provide the necessary retention time. This volume discusses the various types of mine drainage and the passive treatment the options available, and presents guidance on how to design systems correctly.

The use of passive treatment systems for treating coal mine drainage a developing technology. Much is not yet understood and questions remain regarding the optimal design of systems and their longevity. As our experience with passive technologies increases, the information offered here will probably be replaced by more refined information. The Handbook should be used with this clearly in mind.

DEVELOPMENT OF PASSIVE TREATMENT TECHNOLOGY

The mining of coal can result in drainage that is contaminated with high concentrations of dissolved iron, manganese, aluminum, and sulfate. The 1977 Surface Mining Control and Reclamation Act requires that mine drainages from all active and many inactive mines comply with effluent quality criteria. Chemical treatment to meet these criteria imposes a serious financial burden on coal companies. The high costs of chemical treatment also limit efforts to treat contaminated water from abandoned mine lands (AML). Passive treatment offers a low-cost alternative to conventional chemical treatment for active mines.

For some mine drainages, constructed wetlands provide the treatment needed to produce water that meets effluent limitations. For other drainages, constructed wetlands are used to pretreat mine water before conventional treatment, thereby lowering treatment costs, or they are used as a part-time alternative to full-time treatment. At pre-1977 sites and abandoned sites, constructed wetlands offer a low-cost means of improving the quality of mine water before it is released to receiving streams, although treatment may not achieve effluent standards. For abandoned mine lands, passive treatment may be the only economically feasible option for treating persistent drainages.

The treatment of mine drainage by wetlands has evolved from simple surface flow wetlands to sequential treatment in a variety of wet environments. Early constructed wetlands were built to mimic the peat (*Sphagnum*) wetlands that first showed that the quality of mine water was improved as it passed through these wetlands. However, *Sphagnum* wetlands proved to be difficult to establish and maintain and the design was replaced by one in which emergent plants, most often cattails, are the dominant vegetation.

Recently, passive treatment options have been expanded to include anoxic limestone drains (ALD), which add alkalinity to the drainage before wetland treatment, and successive alkalinity-producing systems (SAPS), which reduce the amount of surface area needed to generate alkalinity. Often, several treatment options are used sequentially.

Opinions vary on the merits of passive treatment of mine waters by constructed wetlands. Analyses by the Office of Surface Mining and others (for instance, Wieder 1989, Wieder et al. 1990) question the feasibility of the constructed wetland concept. On the other hand, many constructed wetland systems have worked quite well for a number of years (Brodie et al. 1993, Taylor et al. 1993, Hedin et al. 1994, Stark et al. 1994). Hundreds of constructed wetlands are now

being used to decrease concentrations of contaminants from active, reclaimed, and abandoned mines before the water is released, although not all of the systems may consistently treat water to effluent standards.

COSTS AND LIMITATIONS OF PASSIVE TREATMENT

Compared with conventional chemical treatment, passive methods usually require more land area but use less costly reagents and require less operational attention and maintenance.

The costs of wetland treatment are usually measured in terms of the land required. Constructed wetlands take advantage of natural chemical, physical, and biological processes to decrease metal concentrations and to neutralize acidity. Since some of these removal processes are slower than those used in conventional treatment, particularly if adequate alkalinity is not provided, the required retention times are longer and the area needed for treatment is larger for wetland treatment than for conventional treatment. If wetlands are to produce water that meets effluent criteria, the wetland must be large enough to provide the necessary retention times. The land available for treatment is often limited on mine sites and the sizing of constructed wetlands becomes crucial factor. Undersized wetlands cannot be expected treat water to compliance levels. At present, however, there is no way to predict precisely the effects of wetland treatment on a particular mine drainage and therefore to size the wetland precisely.

In sizing constructed wetlands at active mines, the available space and the costs of construction must be balanced against influent water quality and chemical treatment costs. Wetland treatment may be advantageous for alkaline to moderately acidic mine water; for highly acidic mine water the wetland may have to be so large that conventional treatment becomes cheaper. If the decision is made not to use year-round wetland treatment, wetlands can still play an important role in treat-

ing mine water. Treating mine water passively by wetlands before the water enters a chemical treatment system can reduce the costs for conventional treatment. Constructed wetlands can also be used as a part-time alternative to full-time chemical treatment.

INFORMATION SOURCES FOR HANDBOOK

Much of the material presented here has been summarized from the US Bureau of Mines Information Circular 9389, "The Passive Treatment of Coal Mine Drainage" (Hedin et al. 1994). The Circular is an important reference for those interested in designing passive treatment systems. It includes detailed information on the systems that provided the data base for the Circular and presents an in-depth discussion of mine drainage chemistry and the theoretical basis for passive treatment.

The SAPS concept is not covered by the Circular. The discussion of SAPS is based largely on the recent work of D. A. Kepler and E. C. McCleary (Kepler and McCleary 1994, McCleary and Kepler 1994).

CHAPTER 2

BASIS FOR WETLAND TREATMENT OF MINE WATER

INTRODUCTION

A number of natural processes reduce the impacts of mine drainage on receiving waters. As water flows through streams, rivers, lakes, and wetlands, its toxic characteristics decrease through chemical and biological reactions and through dilution with uncontaminated water. Metals react with oxygen in aerated water and precipitate as oxides and hydroxides. Dissolved iron (Fe) precipitates as an orange oxyhydroxide, dissolved manganese (Mn) precipitates as a black oxide or oxyhydroxide, and dissolved aluminum (Al) as a white hydroxide. The low pH that is common to many mine drainages is raised either by mixing with alkaline or less acidic water, or through contact with carbonate rocks.

The goal of constructed wetland treatment is to have these processes occur in the wetland rather than in the receiving water. Passive treatment systems function by retaining contaminated mine water long enough for chemical, physical, and biological processes to lower contaminant concentrations to acceptable levels. Efficient passive systems create conditions that promote the processes that most rapidly remove contaminants. Thus, the design of efficient passive systems must be based on an understanding of mine drainage chemistry and how different passive technologies affect this chemistry.

The following is a summary of the discussion of mine drainage chemistry presented in the Bureau of Mines Information Circular 9389 (Hedin et al. 1994), which can be referred to for a more complete discussion.

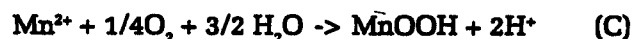
ACIDITY AND ALKALINITY

Acidity is a measure of the base neutralization capacity of a volume of water. There are three types of acidity:

- proton acidity (a measure of free H⁺ ions)
- mineral acidity associated with dissolved metals

- organic acidity associated with dissolved organic compounds.

Organic acidity is generally low in coal mine drainages because they contain little dissolved organic carbon. In mine drainage, acidity arises from free protons (low pH) and from the mineral acidity associated with dissolved Fe, Mn, and Al. Of the many possible reactions of metals with water, there are four major simplified reactions that remove dissolved metals from solution. All produce free protons:



These reactions can be used to calculate the total acidity of a sample of mine water and to partition the acidity into its various components. The expected acidity of a mine water is calculated from its pH and the sum of the milliequivalents of acidic metals:

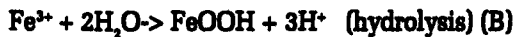
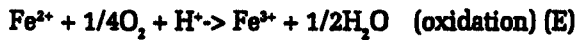
$$\text{Acid}_{\text{calc}} = 50 [2\text{Fe}^{2+}/56 + 3\text{Fe}^{3+}/56 + 3\text{Al}/27 + 2\text{Mn}/55 + 1000(10^{-\text{pH}})] \quad (1)$$

where all metal concentrations are in milligrams per liter (mg/L) and 50 is the equivalent weight of calcium carbonate (CaCO₃) and thus transforms milliequivalent per liter of acidity into mg/L CaCO₃ equivalent (Hedin et al. 1994).

The contributions to acidity from free protons and from dissolved metals vary from mine water to mine water, and are used as a guide to the type of passive treatment needed for effective treatment. The acidity of many mine waters arises largely from dissolved metals (mineral acidity) rather than from free protons (pH).

When the pH of a mine water is greater than 4.5, it has acid neutralizing capacity and is said to contain alkalinity. Alkalinity neutralizes mineral acidity and buffers changes in pH. Unless buffered by alkalinity, the reactions of metals with water (reactions A - D) will decrease pH.

In water, Fe and Mn undergo oxidation and hydrolysis reactions. Oxidation decreases acidity while hydrolysis increases acidity. For Fe, the reactions are:



Alkalinity and acidity are not mutually exclusive. When water contains both alkalinity and mineral acidity, a comparison of the two measurements determines whether the water is net alkaline (alkalinity is greater than acidity) or net acidic (acidity is greater than alkalinity). Net alkaline water contains enough alkalinity to neutralize the mineral acidity represented by dissolved Mn and ferrous Fe.

In mine water, the principal source of alkalinity is dissolved carbonate, which can be present as undissociated H_2CO_3 , bicarbonate (HCO_3^-), or carbonate (CO_3^{2-}). Bicarbonate and carbonate can neutralize proton acidity:



As Fe and Mn oxidize and hydrolyze, the resulting proton acidity can be neutralized by bicarbonate. For waters contaminated with Fe^{2+} , the net reaction is:



This reaction indicates that net alkaline waters contain at least 1.8 mg/L alkalinity for each 1.0 mg/L of dissolved Fe. Waters that contain a lesser ratio are net acidic, since oxidation and hydrolysis of the total dissolved Fe content results in a net release of protons and a decrease in pH. The presence of bicarbonate alkalinity in mine waters that contain elevated levels of metals is not unusual. While all mine drainages are commonly referred to as "acid mine drainages", in fact many discharges are net alkaline.

When pH drops below 6, the rate of Fe oxidation slows dramatically and the removal of Mn by oxidation and hydrolysis virtually stops (Nairn et al. 1991). Alkalinity is therefore important for three reasons: it neutralizes mineral acidity, it

buffers against changes in pH, and it enables the removal of Mn.

Passive treatment systems can be expected to perform more effectively when the raw mine water has a pH greater than 6.0 and contains net alkalinity.

IRON AND MANGANESE REACTIONS IN AEROBIC ENVIRONMENTS

When mine water flows through aerobic environments, the oxidation and hydrolysis reactions discussed above cause concentrations of ferrous Fe (Fe^{2+}), ferric Fe (Fe^{3+}), and Mn to decrease. Whether these reactions occur quickly enough to lower metal concentrations to an acceptable level depends on:

- the availability of oxygen for oxidation reactions
- the pH of the water
- the activity of microbes that catalyze reactions
- the retention time of the water in the treatment system.

The pH is an especially important parameter because it influences both the solubility of metal hydroxide precipitates and the kinetics of the oxidation and hydrolysis processes. The relationship between pH and metal removal processes in passive treatment systems is complex because it differs among metals and also between biotic (biological) and abiotic (physical/chemical) processes.

In general, Fe and Mn precipitate sequentially, not simultaneously. Fe oxidizes and precipitates much more rapidly than Mn because oxidized Mn solids are unstable in the presence of Fe^{2+} . Concentrations of Fe^{2+} must be reduced to low levels before Mn can be converted to a stable solid precipitate.

IRON OXIDATION AND HYDROLYSIS

The most common contaminant in mine drainage is ferrous iron (Fe^{2+}). Fe^{2+} is removed from mine water largely through oxidation and hydrolysis (reactions E and B). In the oxidizing environments common to most surface waters, Fe^{2+} is converted to Fe^{3+} through ferrous oxidation. The conversion occurs both abiotically and as a result of bacterial activity.

The pH of the mine water affects the kinetics of the abiotic and biotic processes. For oxygenated waters with pH less than 4, Fe removal is limited by the oxidation process (reaction E). When oxygen is not limiting, the rate of abiotic Fe oxidation slows 100-fold for every unit decrease in pH. At pH values greater than 8, the abiotic process is fast (rates are measured in seconds) while at pH values less than 5 the abiotic process is slow (rates are measured in days). In contrast, bacterial oxidation of Fe^{2+} peaks at pH values between 2 and 3, and diminishes at pH less than 5. Abiotic oxidation processes dominate over bacterial oxidation processes at pH values above 6 while bacterial processes dominate at pH values below 5.

At pHs between 6 and 7, a range in which abiotic iron oxidation processes should dominate, the presence of bicarbonate alkalinity buffers mine water. Waters that contain no alkalinity have pHs less than 4.5 and the removal of Fe under oxidizing conditions occurs primarily by bacterial oxidation accompanied by hydrolysis and precipitation.

As Fe^{2+} is converted to Fe^{3+} , it is subject to hydrolysis reactions that can precipitate it as an oxyhydroxide (FeOOH) or hydroxide [$\text{Fe}(\text{OH})_3$]. The hydrolysis reaction occurs abiotically. The solubility of iron oxyhydroxide solids is such that, under equilibrium conditions, the amount of dissolved Fe^{3+} is negligible (<1 mg/L) if the pH is greater than 2.5. In actuality, the rate of the hydrolysis reaction is also pH-dependent, and significant Fe^{3+} can be found in mine water with a pH less than 2.5. A fourth-order relationship with pH has been suggested, in which Fe^{3+} hydrolysis

processes shift from a very rapid rate at pH above 3 to a very slow rate at pH below 2.5 (Singer and Stumm, cited in Hedin et al. 1994).

The tendency for dissolved Fe to oxidize and hydrolyze in aerobic environments with pH greater than 3 results in the precipitation of FeOOH and $\text{Fe}(\text{OH})_3$. Because the net result of the oxidation and hydrolysis process is the production of protons, the process can decrease pH. Thus, the passage of circumneutral net acidic water through wetlands commonly decreases both Fe concentration and pH.

MANGANESE OXIDATION AND HYDROLYSIS

Manganese oxidation and hydrolysis reactions result in the precipitation of manganese oxyhydroxides (MnOOH) and manganese oxides (Mn_2O_3 and MnO_2). If the environment is alkaline, manganese carbonate (MnCO_3) can also form. The specific mechanism(s) by which Mn^{2+} precipitates from aerobic mine water in the absence of chemical additions is uncertain. The processes generally result in the formation of MnO_2 , which precipitates.

While the reactions that remove Mn are mechanistically similar to those that remove Fe, Mn removal rates are 20 to 40 times slower than Fe removal rates under similar pH and Eh conditions. The kinetics of Mn^{2+} oxidation are strongly affected by pH. Abiotic reactions are very slow at pH less than 8. Microbes can catalyze Mn^{2+} oxidation, but do so only in oxygenated waters with pH greater than 6.

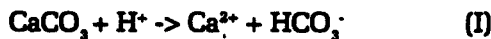
Although the hydrolysis of Mn produces protons, the precipitation of MnOOH does not result in large declines in pH, as can happen when FeOOH precipitates, because there is no natural mechanism that rapidly oxidizes Mn^{2+} under acidic conditions. If the pH falls below 6, Mn^{2+} oxidation virtually stops, the proton-producing hydrolysis reaction stops, and pH stabilizes.

IRON AND MANGANESE REACTIONS IN ANAEROBIC ENVIRONMENTS

Chemical and microbial processes in anaerobic environments differ from those in aerobic environments. Because O_2 is absent, Fe^{2+} and Mn^{2+} do not oxidize and their oxyhydroxide precipitates do not form. Hydroxides of reduced Fe and Mn ions [$Fe(OH)_2$ and $Mn(OH)_2$] do not form because they are highly soluble under acidic or circumneutral conditions. The chemistry of mine water in anaerobic environments is influenced by the chemical and biological processes that generate bicarbonate (HCO_3^-) and hydrogen sulfide (H_2S).

LIMESTONE DISSOLUTION

A major source of HCO_3^- in many anaerobic environments is the dissolution of carbonate materials, such as calcite:



Carbonate dissolution in anaerobic mine water environments can result in higher concentrations of HCO_3^- than in aerobic environments for two reasons:

- the absence of Fe^{3+} in most anaerobic environments limits the formation of $FeOOH$ coatings that armor (coat) carbonate surfaces and inhibit further carbonate dissolution in aerobic environments
- decomposition of organic matter and the neutralization of proton acidity in anaerobic environments leads to high CO_2 partial pressures which foster the dissolution of carbonate compounds.

The observation that limestone dissolution is enhanced by contact with mine water has led to the construction of anaerobic limestone treatment systems, known as anoxic limestone drains (ALD). In an ALD, mine water flows through a bed of limestone gravel that has been buried to limit inputs of atmospheric oxygen. The containment caused by burial also traps CO_2 within the ALD, fostering the development of high CO_2 partial pressures.

The amount of alkalinity that can be generated by an ALD is limited to a maximum value that is a function of the CO_2 partial pressure within the ALD. When the water in an ALD reaches equilibrium with the $CaCO_3$, no further alkalinity can be generated. The maximum amount of alkalinity that can be generated is still open to question. Hedin and Watzlaf (1994) estimate that the maximum amount of alkalinity that can be generated in a properly-functioning ALD is about 300 mg/L.

ALD treatment is discussed in Chapter 6.

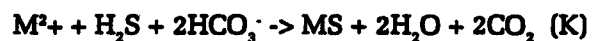
SULFATE REDUCTION

When mine water flows through an anaerobic environment that contains an organic substrate, bacterial sulfate reduction can occur. Sulfate-reducing bacteria use sulfate in their metabolism, releasing hydrogen sulfide and bicarbonate in the process:



where CH_2O represents organic matter.

Bacterial sulfate reduction results in the precipitation of dissolved metals as metal sulfide solids:



where M represents metals. For Fe, pyrite formation is also possible:



The bacteria require the presence of sulfate, suitable concentrations of organic compounds, a pH greater than 4, and the absence of oxidizing agents such as O_2 , Fe^{3+} , and Mn^{4+} . These conditions are met in mine drainage wetlands that contain organic matter and anaerobic substrates.

The precipitation of metal sulfides in an organic substrate improves water quality by decreasing the mineral acidity without causing a parallel increase in proton acidity. Protons released by H_2S dissociation ($H_2S \rightarrow 2H^+ + S^{2-}$) are neutralized by an equal release of HCO_3^- during sulfate reduction. For coal mine drainage, where metal contamination is generally limited to Fe,

Mn, and Al, the H_2S produced by bacterial sulfate reduction primarily affects dissolved Fe. Aluminum does not form any sulfide compounds in wetland environments and the relatively high solubility of MnS makes its formation unlikely.

Spent mushroom compost, which consists of a combination of spoiled hay, horse manure, corn cobs, wood chips, and calcite, has been used as the organic substrate at many constructed wetlands in western Pennsylvania. In wetlands built with substrates of limestone and spent mushroom compost, the alkalinity of the pore water appears to result from a combination of limestone dissolution and sulfate reduction. In wetlands constructed with an organic substrate, the pore water (the water within the pores of the substrate) often has a pH of 6 to 8. These alkaline conditions result in part from microbial reactions involving H_2S that result in the net generation of HCO_3^- . In most wetlands (natural and constructed), surface waters are generally aerobic while the underlying pore waters are anaerobic. Because pore waters are circumneutral and strongly buffered by HCO_3^- , the removal of Fe by oxidation as the pore water diffuses into aerobic surface zones can occur rapidly.

ALUMINUM

Aluminum has only one oxidation state in aquatic systems: +3. Oxidation and reduction processes, which complicate Fe and Mn chemistry do not directly affect concentrations of dissolved Al. Instead, concentrations of Al are primarily influenced by the solubility of $Al(OH)_3$. The passage of mine water through highly oxidized or highly reduced environments has no effect on Al concentrations unless the pH changes. When pH decreases (due to Fe oxidation and hydrolysis), concentrations of Al increase because of the dissolution of alumino-silicate clays, aluminum oxides, or aluminum hydroxides by the acidic water. At pH levels less than 4, $Al(OH)_3$ is highly soluble and high concentrations of Al^{3+} and Al ion complexes are possible. At pH levels between 5 and 8, $Al(OH)_3$ is highly insoluble and free Al is not present in solution. If the pH of acidic mine water is raised during passage through anaerobic environments (because of carbonate dissolution or microbial activity), $Al(OH)_3$ precipitates.

CHAPTER 3

DESIGN OF PASSIVE TREATMENT SYSTEMS

The passive treatment of mine drainage by constructed wetlands uses chemical and biological mechanisms to improve the quality of the water. Whether or not a constructed wetland will perform well in treating a specific mine water depends primarily on two factors:

- selecting the correct treatment process, or sequence of processes
- sizing the wetland correctly, so that the mine water is retained long enough for treatment to occur.

MINE WATER CHARACTERISTICS

WATER QUANTITY

An accurate measurement of the flow rate of the discharge or seepage is needed. Flow rates can vary significantly throughout the year and in response to storm events. Intense storms in summer, and snowmelt and high groundwater tables in winter and early spring can increase flows to ten times average annual flows. Such large volumes of water can result in flushing events that can greatly increase the concentrations of Fe, Mn, and solids. If a wetland discharge must consistently meet effluent criteria, the wetland must be designed to ensure sufficiently long retention times.

Flow rates are best determined by actual flow measurements. Measurements of water volume per unit time can be made with buckets or with simple weirs and flumes. If accurate flow data cannot be obtained and the system must produce water that meets effluent criteria, the system must be over-designed to assure adequate retention.

WATER QUALITY AND CONTAMINANT LOADINGS

Water samples for chemical analyses should be collected at the mine discharge or seepage point. Water quality analyses should include pH, alkalinity, hot acidity (H_2O_2 , method 2310 4a, APHA 1992), Fe, Mn, and Al. Samples for metal analysis should be acidified as soon as they are collected. Samples containing visible particulates should be filtered before being acidified. If an ALD is being considered, the acidified sample should be analyzed for Fe^{2+} and Al, and a field measurement of the dissolved oxygen (DO) of the mine water should be made.

Contaminant loading rates must be determined. Loadings of contaminants (Fe, Mn, and acidity) are calculated by multiplying contaminant concentrations by the flow rate. If the concentrations are in mg/L and the flow rate is in gallons per minute, the calculation is:

$$\text{loading of (Fe, Mn, acidity)(grams/day)} = \text{flow (gpm)} \times \text{mg/L (Fe, Mn, acidity)} \times 5.45 \quad (2)$$

If the concentrations are in mg/L and the flow rate is in liters per minute, the calculation is:

$$\text{loading of (Fe, Mn, acidity)(grams/day)} = \text{flow (L/min)} \times \text{mg/L (Fe, Mn, acidity)} \times 1.44 \quad (3)$$

Concentrations may vary as hydrologic conditions change. Concentrations may be greatest during low flows if high flows dilute pollutants. However, if high flows flush materials or raise the levels of mine pools, concentrations may be highest during high flows. It is therefore important to determine loadings for average conditions and also for those times when flows and contaminant concentrations are high to ensure that the wetland is adequately sized to accommodate the range of conditions.

TREATMENT OPTIONS

Four options for passive treatment are available. Each is most appropriate for a particular type of mine water. They can be used in sequence. The options are:

- aerobic wetlands, which promote oxidation reactions to precipitate metals as oxides and hydroxides. These wetlands typically contain cattails growing in a soil or spoil substrate. Aerobic wetlands are surface flow (SF) wetlands.
- organic substrate wetlands, which are often called compost wetlands. In these wetlands, the water flows through thick layer of organic material. The anaerobic conditions in the organic layer promote chemical and microbial processes that generate alkalinity and neutralize acidity. Organic material includes spent mushroom compost, peat, hay bales, and manure.
- anoxic limestone drains (ALD), which are buried beds of limestone. The limestone adds alkalinity to the water, which is then fed to a settling pond and wetland where the metals are precipitated. The ALD is sealed to exclude oxygen so that limestone dissolution can occur without armoring (the deposition of metal oxyhydroxides on the limestone) which blocks further dissolution. ALDs are not wetlands, but a pretreatment to prepare acidic water for wetland treatment.
- successive alkalinity-producing systems (SAPS), which place an organic substrate wetland over a layer of limestone. Water is introduced at the top, flows down through the layers, and is discharged from the bottom. As the mine water moves down through the layers, microbial activity removes dissolved oxygen and reduces Fe^{3+} to Fe^{2+} . Alkalinity is then produced by bacterial sulfate reduction in the organic layer and by limestone dissolution in the limestone layer. The strongly reducing environment of the organic layer prevents the armoring of the limestone. The water discharges to a settling pond where the metals are precipitated. Mine water can be recycled through a SAPS or passed through several SAPSs as often as necessary to remove the acidity.

PROCESS SEQUENCE

A decision tree for selecting the appropriate treatment, or sequence of treatments, for a specific mine water is given in figure 1. The treatment to be used depends largely on whether the water is net acidic or net alkaline.

NET ALKALINE WATER

Net alkaline waters are treated with aerobic wetlands. Since additional alkalinity is not needed, an ALD or an organic substrate is not needed. The design of aerobic wetlands for net alkaline water is discussed in Chapter 5.

NET ACIDIC WATER

Net acidic waters require that alkalinity be added to remove metals and raise pH. There are three options for adding alkalinity: an ALD, a SAPS, and a compost wetland.

An ALD can be used if DO, Fe^{3+} , and Al concentrations are low (<1 mg/L). A properly-sized ALD can add 150 to 300 mg/L alkalinity to the mine water. The ALD is followed by a settling pond for removal of the solids. If the acidity of the mine drainage is <300 mg/L, the water can be treated with an aerobic wetland after the settling pond. If the acidity of the mine drainage is >300 mg/L, the water should be treated with a SAPS to add further alkalinity since the ALD will not add sufficient alkalinity.

If the mine water contains >1 mg/L DO or Fe^{3+} , an ALD should not be used since $\text{Fe}(\text{OH})_3$ will form and armor the limestone. Instead, alkalinity should be added by a compost wetland or a SAPS. Treatment by a compost wetland requires a larger wetland than does treatment by a SAPS. In northern Appalachia, the treatment of highly acidic waters by compost wetlands does not consistently transform these waters into alkaline waters, particularly during the winter.

Organic substrate wetlands are discussed in Chapter 5, ALD treatment in Chapter 6, and SAPS treatment in Chapter 7.

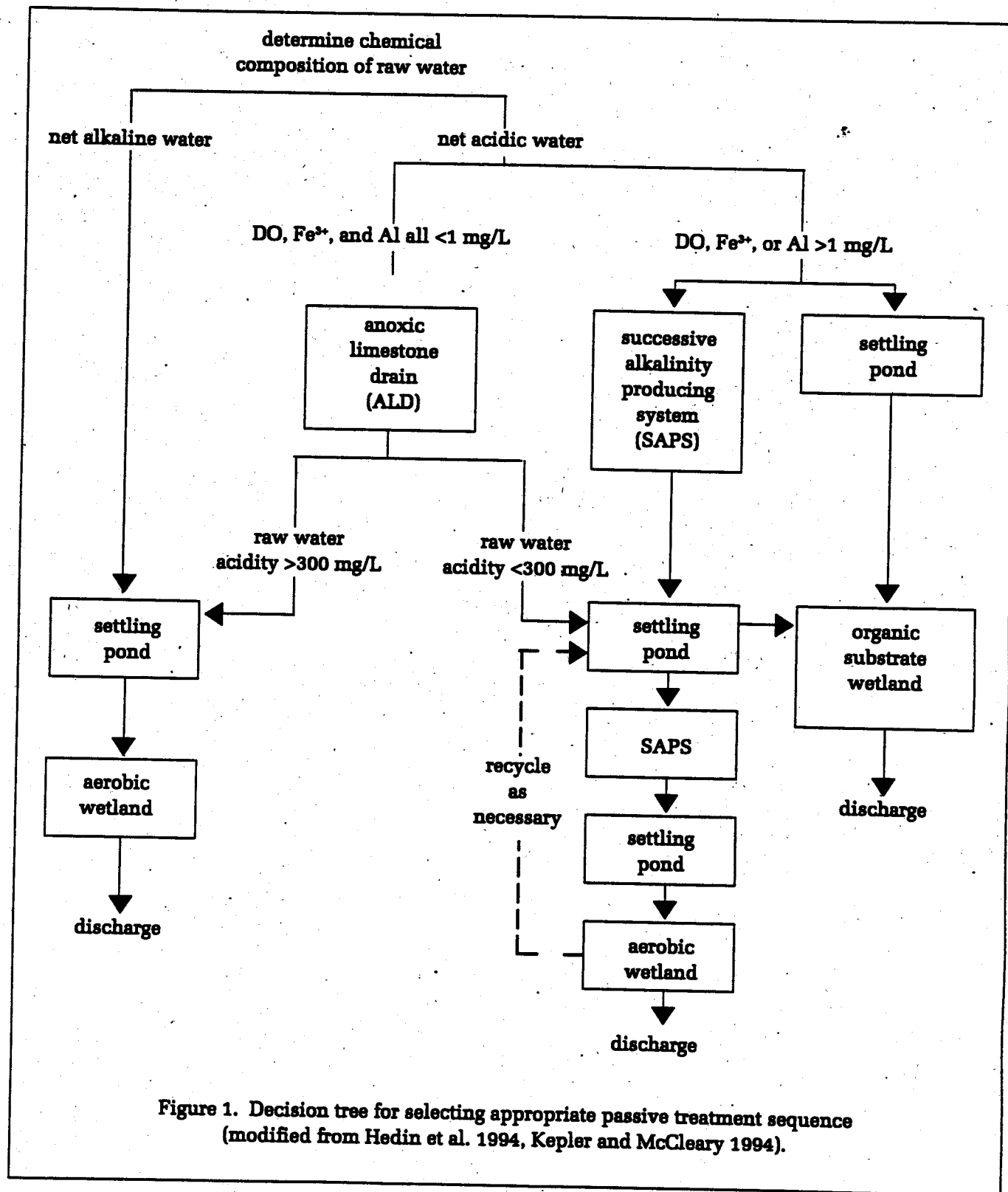


Figure 1. Decision tree for selecting appropriate passive treatment sequence (modified from Hedin et al. 1994, Kepler and McCleary 1994).

Marginally acidic waters (net acidities of 0 - 100 mg/L) have sometimes been treated successfully with aerobic wetlands. However, at present there is no way to predict which marginally acidic waters can be successfully treated with aerobic systems and which cannot. For marginally acidic waters, incorporating an alkalinity-generating component into the design is recommended.

SETTLING BASINS AND PONDS

A settling basin or sediment pond is usually installed after an ALD or SAPS to remove metal precipitates (Brodie et al. 1993, Kepler and McCleary 1994). The advantage of a pond is that it can be easily dredged. The precipitation of Fe from newly aerated and highly alkaline water occurs physico-chemically and does not require wetland treatment. Hedin et al. (1994) suggest that about 50 mg/L Fe can be removed by a pond before additional aeration is needed. For an acidic drainage with low to moderate Fe concentrations, and depending on site conditions, the wetland and pond can be combined into marsh-pond cell with a large portion devoted to deep water (3 - 6 ft) where precipitates can accumulate.

During rainstorms, flows through a wetland can increase by an order of magnitude or more with a drop in treatment efficiency to near zero (Stark et al. 1994). A polishing pond placed between the wetland and the discharge to the receiving water can be valuable in preventing discharges of resuspended materials during high flows. At a site that included three polishing ponds after a wetland system, discharges remained in compliance during intense summer storms that tripled inflow rates, an outcome attributed to the ponds (Stark et al. 1994).

SIZING

A method for calculating the optimal size of passive treatment systems for various types of mine drainage has not yet been developed. Some

systems have been greatly oversized while others have been greatly undersized. Sizing is affected by the composition of the mine drainage, the quantity of water to be treated, and the specifics of the site. The seasonally-variable nature of mine discharges and the varying effectiveness of passive treatment during different times of the year further complicate the assessment of the size needed to achieve a certain level of water quality. The size of passive treatment systems has often been determined by the space available for such treatment rather than by optimal treatment requirements.

Based on the performance of 13 constructed wetland systems in Pennsylvania, the Bureau of Mines (Hedin et al. 1994) suggests two sets of sizing criteria :

- abandoned mined land (AML) criteria. In many AML situations, the goal is cost-effective improvement in water quality rather than compliance with effluent standards. The criteria are based on removals that have been observed at existing sites. Wetland treatment at these sites significantly improves water quality, although in many cases the wetlands may not consistently lower contaminant concentrations to NPDES effluent standards.
- compliance criteria. These criteria are suggested for wetlands that must produce effluents that comply with NPDES effluent standards. The criteria are conservative and result in wetlands that are twice as large as AML wetlands.

Table 1 gives recommended wetland sizes for compliance and AML criteria. For example, to size a wetland according to the Fe AML criterion in table 1:

$$\text{minimum wetland size (ac)} = \text{Fe loading (lb/day)} \\ + 180 \text{ (lb/ac/day)}$$

To remove both Fe and Mn, the size needed to remove Fe must be added to the size needed to remove Mn. This is necessary because Fe and Mn are removed sequentially in constructed wetlands.

Fe removal rates may be a function of Fe concentration, with removal rates decreasing as Fe concentrations become smaller, and the suggested compliance criteria sizing value for Fe (90 lb/ac/day, 10 g/m²/day) is conservative to account for this.

The criteria for Mn removal reflect the large size of wetland needed to remove Mn. Because the toxic effects of Mn at moderate concentrations are generally not significant and the size of the wetland needed to treat Mn-contaminated water is so large, Hedin et al. (1994) suggest that AML sites with Fe problems should receive a higher priority than those with only Mn problems.

Treatment systems that incorporate more than one treatment methodology are sized by summing the treatment areas needed for each of the components. For instance, to remove Fe and acidity, a system should be sized for the removal of Fe by an aerobic wetland plus the removal of acidity by a compost wetland.

DESIGN CONSIDERATIONS

SYSTEM LAYOUT

The characteristics of the mine drainage and the site determine the layout of the components of the treatment system. The source, volume, and probable variability of the drainage should be

understood before the system is designed. The mine maps and/or core borings should be reviewed to determine the source of the water (shaft tunnel, or waste pile), and the area drained by this source. Information on groundwater and underlying soils is also needed. Site topography affects cut and fill requirements, drainage and erosion characteristics, and slope stability. The system should be designed for gravity flow.

Since precipitation is a major Fe removal pathway, the configuration should create long flow paths to provide long retention times. Long flow paths can be created by building long narrow channels or by using baffles to create serpentine flow paths in short, wide cells. Very large cells are subject to short-circuiting and should be hydrologically chambered with simple low or subsurface finger dikes, logs, riprap baffles, or other structures. Hay bales can be used but they decompose readily and must be replaced periodically. Alternatively, irregular or serpentine cells can be built. The cells should be designed to maximize the contact of the mine water with the surfaces in the cell and to avoid channeling and short-circuiting.

The geometry of the site, as well as flow and treatment considerations, may dictate the use of multiple cells. The intercell connections can be ditches, riprapped channels, V-notch weirs, or lined railroad tie steps. The intercell connections should dissipate enough kinetic energy so that the water entering a cell will not cause erosion or disturb precipitated solids. The

Table 1. Bureau of Mines compliance and abandoned mine land (AML) sizing criteria for alkaline and acidic mine waters (from Hedin et al. 1994).

	<u>lb (Fe, Mn, acidity) / ac wetland surface / day</u>				<u>g (Fe, Mn, acidity) / m² wetland surface / day</u>			
	<u>Compliance</u>		<u>AML</u>		<u>Compliance</u>		<u>AML</u>	
	<u>Alkaline</u>	<u>Acidic</u>	<u>Alkaline</u>	<u>Acidic</u>	<u>Alkaline</u>	<u>Acidic</u>	<u>Alkaline</u>	<u>Acidic</u>
Fe	90	na	180	na	10	na	20	na
Mn	4.5	na	9	na	0.5	na	1	na
Acidity	na	30	na	60	na	3.5	na	7

na: not applicable

connections between the cells can serve as aeration devices. Pipes should be avoided because they tend to clog with Fe precipitates. However, pipes must be used to move water from an ALD since precipitates will form on exposure to air, clogging an open channel.

A settling pond should be placed before the first wetland cell to remove particulate material that does not need wetland treatment. Effluent from the wetland system should pass through another settling pond to settle Fe solids before the water is discharged to the receiving water.

WATER CONTROL

Mine drainage can come from a variety of sources, including existing surface drainage ditches, seeps from backfill or spoil, and openings such as mine portals or boreholes. These sources must be plumbed into the treatment system. The plumbing must be able to handle variations in flow without leaking.

Seeps and surface drainage can be collected either in strategically-located collection ponds or in contour ditches that discharge to a collection pond or the first wetland cell. These structures can also serve as pretreatment or primary treatment units to settle precipitates. The advantages of collection ponds are that they control flow surges and provide a relatively constant flow to the system. Inflow surge control is particularly important if the drainage is a surface flow or is heavily influenced by runoff since these flows fluctuate in response to changes in the weather. A relatively steady flow to the wetland simplifies design and minimizes hydraulic stresses during operation.

To collect seepage from spoil, French drains in the toe area can be used. The drain system should be designed to minimize mixing with uncontaminated water to avoid flow surges to the system. Pipes can be chronic maintenance problems because of the clogging that results when Fe precipitates collect in pipes. If pipes are necessary, as they are for moving water from an ALD,

steps must be taken to exclude oxygen. A gas trap should be placed at the exit and clean-out plugs should be provided.

The mine drainage can be collected to yield a single flow to the wetland. If several widely different drainages are to be treated, combining them may or may not be desirable. It may be advantageous to combine a low flow, high strength acid mine drainage (AMD) with a high flow, low strength AMD to dilute the strong AMD. On the other hand, adding a low flow, low strength AMD to a high flow, high strength AMD will only increase the amount of strong AMD to be treated and separate systems may be preferable.

Water levels can be controlled with a spillway or water control structure (weir or swiveling standpipe). Spillways are simple to construct but are not adjustable and problems arising from incorrect water levels can be difficult to remedy. Spillways should consist of wide cuts in the dike with sides no steeper than 2H:1V and lined with erosion-resistant material. Spillways must be able to pass the expected high and low flows.

In areas where vandalism could be a problem, the use of removable boards in weirs or flexible hoses (see Volume 1) should be avoided. Enclosing pipes and valves in protective enclosures may be advisable.

CHAPTER 4

AEROBIC WETLANDS FOR NET ALKALINE WATER

APPROPRIATE APPLICATIONS

Net alkaline mine waters can be treated with aerobic wetlands since net alkaline water contains enough alkalinity to buffer the acidity produced by the metal hydrolysis reactions. The metals - Fe and Mn - will precipitate, given enough time. The goal of wetland treatment of net alkaline water is to aerate the water and to promote the oxidation and settling of the metals.

SIZING

Aerobic systems for the removal of Fe or Mn can be sized with the criteria in table 1. Aerobic, alkaline wetlands function largely through chemical reactions and performance is not strongly affected by season. The limiting factor is the slow settling of metal oxyhydroxides.

CONFIGURATION

A typical aerobic wetland consists of a basin with cattails growing in soil or in alkaline spoil. Water depths are generally 6 - 18 inches (10 - 50 cm). These depths are appropriate for most emergent wetland vegetation. Most wetland plants cannot tolerate water deeper than about 18 inches (50 cm)(see Volume 1).

The depth of the water may vary, depending on the needs of the operator. Shallow water (<6 inches) enhances oxygen levels and oxidizing conditions but freezes more quickly in winter. Shallow wetlands fill more quickly with deposits. Brooks found that saturated sediments rather than standing water enhanced treatment and recommends depths of 0 (saturated soil) to 6 inches (R. P. Brooks, Penn State University, pers. comm.). Deeper water (8 - 24 inches) decreases vegetation diversity and retards oxygenation near the substrate, but can increase the life span of the cell since it provides more space for the accumulation of deposits. Deeper cells may be appropriate for

moderate mine water or as the first stage in a wetland system to accommodate the rapid precipitation of FeOOH. Hedin et al. (1994) suggest that designing wetland with shallow and deep marsh areas plus a few areas of deeper (3 - 6 ft) open water will accommodate seasonal and year-to-year variations in weather and flow.

In many wetlands that treat alkaline water, the removal of Fe appears to be limited by the availability of dissolved oxygen. To promote Fe removal, aeration of the water should be followed by passage through quiet areas where the iron can react with DO and the iron floc can settle out. Aeration can be provided by waterfalls or V-notch weirs. Aeration provides only enough DO to oxidize about 50 - 70 mg/L Fe²⁺. Mine waters with higher concentrations of dissolved Fe can be treated with a series of aeration structures and wetland cells. The wetland cells allow time for Fe oxidation and hydrolysis, and provide space in which the Fe floc can settle out. Many systems add a sedimentation pond after the SF wetland to polish the water before discharge.

Some SF wetlands have been built that do not have typical wetland features. These wetlands consist of open water ditches or shallow, rock-filled ponds with few plants. These wetlands have achieved removal rates similar to wetlands filled with plants. Although plants may not be necessary for Fe and Mn removal, plants may increase the filtration of particulates, prevent flow channeling, and reduce the resuspension of sediments during storms.

LONG-TERM PERFORMANCE

In systems treating alkaline water, Fe and Mn are precipitated by oxidative processes. The rapid removal of Fe means that alkaline systems can be expected to fill up with deposits of metal precipitates. At a site in Ohio, Fe sludge is accumulating at a rate of about 1.25 - 1.5 inches (3 - 4 cm) per year (Stark 1992). Measurements at a number of

sites in Pennsylvania show sludge accumulating at 0.75 - 1.25 inches (2 - 3 cm) per year (Hellier and Hedin 1992). These data suggest that dikes that provide 3 ft of freeboard should provide sufficient volume for 25 to 50 years of treatment.

Stark et al. (1994) found that water quality at some surface mines improved within a decade after regrading and reclamation were completed. At these sites, 25 to 50 years of passive treatment may be enough to mitigate the contaminant problem. At

surface mine sites with continual contaminant production, or at systems constructed to treat drainage from underground mines or coal refuse disposal areas, systems can either be built with greater freeboard or rebuilt when they fill up. Site conditions will decide whether it is more economical simply to bury the wetland in place and construct a new one, or to excavate and haul away the deposits and reestablish the wetland.

CHAPTER 5

ORGANIC SUBSTRATE WETLANDS

APPROPRIATE APPLICATIONS

For mine waters containing dissolved oxygen, Fe^{2+} , Al, or concentrations of >300 mg/L acidity, a organic substrate wetland can be used. Organic substrate 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. The anoxic environment within organic substrate also promotes the dissolution of the limestone.

Organic substrate wetlands in which water flows on the surface of the substrate remove acidity (that is, generate alkalinity) at rates of approximately $2 - 12$ g/m²/d (18 - 107 lb/ac/d). The wide range in performance is largely the result of seasonal variation, with lower rates in winter than in summer. Supplementing the compost with limestone and designing the wetland so that water flows through, rather than over, the organic matter has been shown to improve winter performance.

The three important factors in an organic substrate wetland are:

- a substrate rich in organic matter (to support sulfate reduction)
- vegetation (to replace consumed organic matter)
- flow contained largely within the substrate (to promote reducing reactions).

A material often used in organic substrate wetlands is spent mushroom compost, which is readily available in Pennsylvania, although any well-composted equivalent can be used. Spent mushroom compost has a high CaCO_3 content (about 10% dry weight). Mixing in more limestone may increase the amount of alkalinity that can be generated by CaCO_3 dissolution. A compost substrate that does not have a high CaCO_3 content should be mixed with supplemental limestone. Materials that have been used where spent mushroom compost is not available include sawdust, wood chips, peat moss, composted straw bales, and composted chicken litter.

Compost often contains large amounts of loose organic matter. As this material is flushed out during the first few weeks of operation, shock loading of a stream by high levels of biochemical oxygen demand (BOD) can kill the aquatic life in the stream (D. Seibert, Soil Conservation Service, Somerset, PA, pers. comm.). During system start-up, it may be prudent to store the wetland effluent for several weeks in temporary sediment ponds to reduce BOD concentrations before the water is discharged.

If the pH of the mine water is greater than 4, a pond in which Fe can be oxidized and precipitated should be placed before the wetland. A pond is useful when the influent to the wetland is circumneutral and rapid removal of Fe can be expected as soon as the water is aerated. If the pH is less than 4, iron oxidation and precipitation reactions are too slow for significant removal of Fe by a pond. A sedimentation pond can be placed after the compost wetland to polish the water before discharge.

SIZING

Compost wetlands can be sized according to the Bureau of Mines guidelines (table 1). For example, for an AML site the calculation is:

$$\text{minimum wetland size (ft}^2\text{)} = \frac{\text{acidity loading (g/day)}}{0.7}$$

The acidity removal rate for compost wetlands is influenced by seasonal effects that currently cannot be corrected with wetland design. This is not a problem for mildly acidic water nor should it be a problem in warmer climates. However, in northern Appalachia, compost wetlands do not consistently transform highly acidic water (>300 mg/L acidity) into alkaline water. While considerable cost savings can be realized by using a compost wetland during warm or mild weather, such treatment must usually be supported by conventional treatment during the winter.

CONFIGURATION

The depth of the compost is usually 12 -18 inches (30 - 45 cm). To encourage water flow through the compost, it should be laid gently into the wetland (not compacted). Spent mushroom compost has a bulk density of about 1,100 lb/yd³. A ton of compost usually covers about 3.5 yd³ to a depth of 18 inches (45 cm). Flow within the substrate can be encouraged by sloping or piling the compost a little higher than the free water surface so that the water must flow through the substrate.

Cattails or other vegetation are usually planted in the compost to stabilize it and to provide additional organic matter to fuel the sulfate reduction process.

LONG-TERM PERFORMANCE

At wetlands that treat acidic water by alkalinity-generating processes associated with an organic substrate, performance may decline over time as the alkalinity-generating components become exhausted. Limestone dissolution is limited by the amount of limestone present in the substrate. The limestone content of spent mushroom compost is approximately 30 kg/m³ (1.9 lb/ft³). If a wetland containing a 40 cm (16 inch) depth of compost generates CaCO₃-derived alkalinity at an average rate of 3 g/m²/day (27 lb/ac/d) (the average rate measured by Hedin et al. 1994), then the limestone in the compost will be exhausted in 11 years.

The same volume of compost contains about 40 kg (88 lb) of organic carbon. If bacterial sulfate reduction converts 100% of the carbon to bicarbonate at a rate of 5 g/m²/day (45 lb/ac/d), the carbon will be exhausted in 91 years. This estimate is lengthened by the carbon captured by plants through photosynthesis and shortened by non-sulfate mineralization reactions. Also, part of the carbon is recalcitrant (it does not readily enter into reactions).

A realistic scenario for 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. Their activity in a wetland receiving lower pH water may depend, in part, on the 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 (about 11 years, according to the above calculation). Under these conditions, the chemical buffering capacity of the substrate could be increased by adding additional limestone during wetland construction. In fact, this is done at many sites.

CHAPTER 6

ANOXIC LIMESTONE DRAINS (ALD)

APPROPRIATE APPLICATIONS

In an ALD, alkalinity is produced when acidic mine water dissolves limestone (CaCO_3). The ALD is sealed to prevent contact with atmospheric oxygen, which can result in the limestone becoming armored with Fe precipitates, and to promote the production of carbon dioxide, which increases the dissolution of the limestone. An ALD is simply a pretreatment unit to increase alkalinity and to raise pH before the mine water enters the wetland. Dissolved Fe and Mn are largely unaffected by flow through the drain and an ALD must be followed by a settling basin or wetland in which metals can be oxidized and precipitated.

The amount of alkalinity generated by the ALD is not easy to predict. ALDs that treat acidic mine waters with low concentrations of Fe^{2+} , Al^{3+} , and DO have generally discharged water with alkalinities of 120 - 350 mg/L.

The most important factors limiting the usefulness of ALDs are the presence of Fe^{2+} , Al, and DO. When acidic water containing any Fe^{2+} or Al^{3+} comes in contact with limestone, metal hydroxide particulates, such as FeOOH , $\text{Fe}(\text{OH})_3$, or $\text{Al}(\text{OH})_3$, will form. No oxygen is necessary. The buildup of Fe hydroxides armors the limestone and limits its further dissolution. The buildup of Al hydroxide particulates within the ALD eventually decreases the permeability of the ALD and causes it to plug. The presence of DO in the mine water promotes the oxidation of Fe^{2+} to Fe^{3+} within the ALD and thus potentially leads to armoring and plugging. The short-term performance of ALDs in treating water high in Fe^{2+} , Al^{3+} , and DO can be spectacular (total removal of metals within the ALD). Long-term performance is questionable because of armoring and plugging.

Mine water that contains very low concentrations of DO, Fe^{2+} , and Al (that is, each <1 mg/L) is ideally suited for pretreatment with an ALD. As concentrations of these parameters increase, the risk that the ALD will fail prematurely also increases. Two ALDs fed water containing 20 mg/L

Al became plugged within 6 to 8 months (Hedin et al. 1994).

The suitability of a mine water for ALD treatment can sometimes be made by observing the mine discharge and measuring the field pH. Mine waters that seep from spoils and flooded underground mines and that have field pHs above 5 characteristically have concentrations of DO, Fe^{2+} , and Al^{3+} that are all <1 mg/L. Such sites are a good candidates for pretreatment with an ALD. Mine waters that discharge from open drift mines or have pHs below 5 must be analyzed for Fe^{2+} and Al. Mine waters with pH below 5 can contain dissolved Al and mine waters with pH below 3.5 can contain dissolved Fe^{2+} . In northern Appalachia, most mine drainages with pH below 3 also contain high concentrations of Fe^{2+} and Al.

SIZING

As yet, there is no method for calculating the exact size of an ALD needed to treat a specific mine water discharge. Theoretical calculations can estimate the mass of limestone that will be needed to neutralize a certain discharge for a specified period of time. An important factor in the calculations is the concentration of alkalinity expected to be produced by the ALD.

A maximum value of approximately 275 - 300 mg/L alkalinity has been observed at a number of ALDs that have recently been constructed. The minimum mass of limestone needed to treat a year's flow of mine water can be calculated from the flow rate and the assumption that the ALD will produce the maximum amount of alkalinity (300 mg/L):

$$\begin{aligned} \text{yearly CaCO}_3 \text{ consumption (tons)} &= \\ & \text{flow (gpm)} \times 0.6565 \\ \text{yearly CaCO}_3 \text{ consumption (kg)} &= \\ & \text{flow (L/min)} \times 158 \end{aligned}$$

To determine the total mass of limestone needed in the drain, the above calculation must be

adjusted for the CaCO_3 content and the projected life of the ALD. Masses of limestone are converted to volumes by assuming a density for aggregate limestone of 1.2 - 1.5 ton/yd³.

The amount of alkalinity that can be generated by an ALD is limited by the solubility of the calcite. Hedin and Watzlaf (1994) found that a residence time of about 14 to 23 hours was necessary for mine waters to reach maximum concentration of alkalinity. Retention times longer than 23 hours did not appreciably increase alkalinity concentrations. Hedin and Watzlaf (1994) recommend a residence time of approximately 15 hours. To achieve a 15 hour residence time, the required mass of limestone (m) can be calculated as:

$$M = Q \rho_b t_d V_v$$

where Q is the flow volume of the mine water (in L/min), ρ_b is the bulk density of the limestone (kg/m³), t_d is the retention time and is set to 15 hours, and V_v is the bulk void volume expressed in decimal form. To this mass of limestone, enough must be added to satisfy dissolution losses during the expected life of the ALD:

$$M = Q C T / x$$

where Q is the volume flow of the mine water, C is the predicted concentration of alkalinity in the

ALD effluent (mg/L), T is the design life of the ALD, and x is the CaCO_3 content of the limestone in decimal form. The sum of the two masses is the total limestone required.

TYPE OF LIMESTONE

It is important to use limestone with a high CaCO_3 content because of its higher reactivity compared to limestones with high MgCO_3 or $\text{CaMg}(\text{CO})_3$ content. The limestones used in most of the successful ALDs have had a CaCO_3 content of 85 - 95%.

Most of the successful ALDs have used #3 or #4 limestone. Larger stone provides settling space for grit and other particulate matter that may be generated during construction, but offers less surface area for contact with the mine drainage (Faulkner and Skousen 1994).

CONFIGURATION

An ALD is simply a buried bed of limestone (figure 2). The dimensions of existing ALDs vary considerably. Many older drains were built as long, narrow drains that were approximately 1.5 - 3 ft

Table 2. Sample calculation for ALD.

A sample calculation for determining the mass of limestone (M) in metric tons needed to achieve a concentration (C) of 300 mg/L alkalinity to a flow (Q) for 25 L/min of AMD for 20 years (T) is calculated as follows:

- Given:
1. bulk density of limestone (ρ_b) = 1600 kg/m³
 2. bulk void volume of limestone (V_v) = 50%
 3. CaCO_3 content of limestone (x) = 90%
 4. detention time (t_d) = 15 hr

Solution: $M = (Q \rho_b t_d / V_v) + (Q C T / x)$

$$= \frac{(25 \text{ L/min} \times 60 \text{ min/hr})(1600 \text{ kg/m}^3 \times \text{m}^3/1000 \text{ L} \times \text{mt}/1000 \text{ kg})(15 \text{ hr})}{0.50}$$

$$+ \frac{(25 \text{ L/min} \times 60 \text{ min/hr})(300 \text{ mg/L} \times \text{mt}/10^6 \text{ mg})(20 \text{ yr} \times 8766 \text{ hr/yr})}{0.90}$$

$$= 72.0 \text{ mt} + 87.7 \text{ mt} = 159.7 \text{ mt}$$

This is equivalent to about 27 tons of limestone for each gallon per minute of flow.

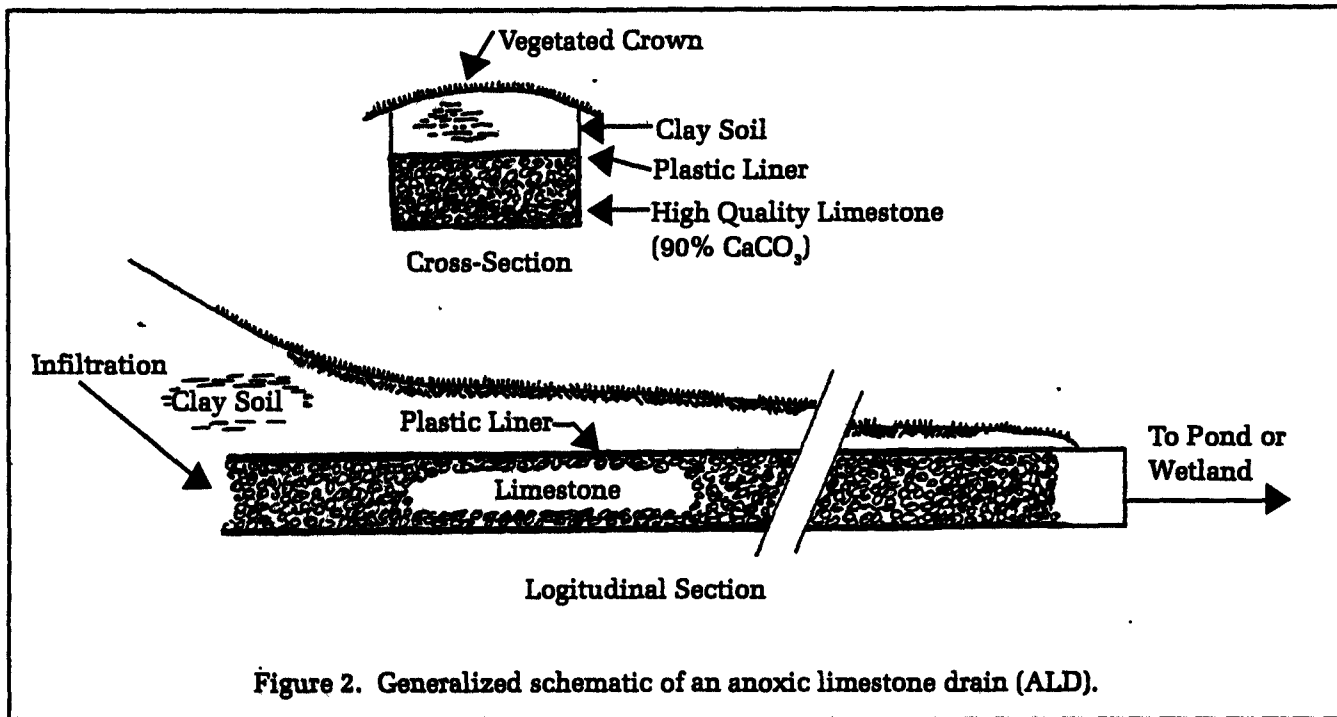


Figure 2. Generalized schematic of an anoxic limestone drain (ALD).

wide. On sites where a linear drain was not feasible, ALDs have been made as wide as 30 - 60 ft; these ALDs have produced alkalinities similar to those produced by narrow, conventional beds.

A typical limestone drain is about 3 ft deep and is capped and covered with soil. The soil surface should be crowned to encourage surface runoff and to allow for subsidence as the limestone is consumed over time. The side slopes of the drain are not critical and are usually made near vertical to simplify construction.

The ALD must be sealed to minimize inputs of atmospheric oxygen to the drain and to maximize the accumulation of carbon dioxide within the drain. Most ALDs are sealed by burying the ALD under several feet of clay. One or two layers of 5 - 10 mil plastic are often placed between the limestone and the clay as an additional gas barrier. Some ALDs have been completely wrapped in plastic before burial. The ALD should be designed so that the limestone will be inundated with water at all times. Clay dikes within the drain, or riser pipes at the outflow of the drain will help to ensure inundation.

A collection system to bring the mine water to the ALD must be devised. Underground mine openings can be sealed and flooded, and the drainage routed to the ALD via a pipe. Alternatively, openings can be backfilled with limestone to create the ALD. Discrete seeps or springs are good starting points for excavation into backfill. Non-point seeps may require more innovative means of collection, such as specialized rock drains or the construction of an embankment to contain the ALD (Brodie et al. 1993).

If the water seeps through backfill or spoil, flow paths should be thoroughly investigated before designing the system. At some sites, preferential flow paths have developed within backfill and the water has bypassed the ALD. A number test pits should be dug several months before the ALD is to be built to determine where the water will go so that the collection system can be designed properly.

The collection system should avoid tapping into sources of uncontaminated water. Some ALDs have unintentionally collected non-target water, thereby increasing the volume of water passing

through the systems. Incorporating water that was more contaminated than the target water led to the failure of an ALD (Hedin and Watzlaf 1994).

LONG-TERM PERFORMANCE

Most ALDs have been built since the late 1980's and there are no data on the long-term performance of ALDs. Many aspects of ALDs make long-term expectations uncertain. ALDs function through the dissolution, and therefore removal, of limestone. Questions remain about the ability of ALDs to maintain unchannelized flow for long periods of time, whether 100% of the limestone can be expected to dissolve, and

whether the drains will collapse after a portion of the limestone has dissolved. In large ALDs, most of the limestone dissolution occurs in the upgradient portion of the limestone bed. The effects of this preferential dissolution on the permeability of the system or its structural integrity are unknown. It may be prudent to provide contingency for failure, for instance by including structural supports.

Another aspect that affects long-term performance is the fact that ALDs retain Fe^{3+} and Al. This retention has raised concerns about the armoring of limestone or the plugging of flow paths long before the limestone is exhausted by dissolution. At present, there is no way to predict exactly how the retention of metals affects ALD performance.

CHAPTER 7

SUCCESSIVE ALKALINITY-PRODUCING SYSTEMS (SAPS)

APPROPRIATE APPLICATIONS

Successive alkalinity-producing systems (SAPS) have been suggested as a means of overcoming the alkalinity-producing limitations of ALDs and the large area required for organic substrate wetlands (Kepler and McCleary 1994, McCleary and Kepler 1994). In a SAPS, the mine water flows down through an organic layer into limestone beds below the organic layer; the pore waters are discharged. A SAPS adds alkalinity through bacterial sulfate reduction and by limestone dissolution. A SAPS is followed by a settling pond where the metals are precipitated. Because DO is removed within the organic layer of the SAPS before the water comes in contact with the limestone, the use of SAPS is not limited by the concentrations of DO, Fe²⁺, or Al in the raw mine water. Three SAPSs have been installed in Pennsylvania since 1990 and all have performed well (Kepler and McCleary 1994, McCleary and Kepler 1994).

Water containing high acidity levels can be treated by passing it through a series of SAPSs. In this case, a SAPS is followed by a settling pond where the metals are precipitated, and then an aerobic wetland where oxidizable organic matter is added, before entering another SAPS. The sequence can be repeated as often as necessary. SAPS treatment offers two advantages over other treatment options:

- since a SAPS relies on volume rather than surface area for treatment contact, the required surface area can be smaller than for an aerobic or organic substrate wetland
- the water can be passed through a SAPS as often as necessary.

Kepler and McCleary (1994) did not encounter any hydraulic problems due to physical plugging of the limestone. They suggest that the pressure exerted by the downward force of the free-standing pool above the substrate in a SAPS reduces the risk of physical plugging compared to the lateral

flow in an ALD (Kepler and McCleary 1994, McCleary and Kepler 1994). The SAPS design, with adequate freeboard, allows for the buildup of the static head required to move water down and through the substrates. This, in combination with the bottom discharge, maintains vertical flow.

SIZING

A SAPS should be sized to provide the retention time required to produce the needed alkalinity. The limestone dissolution in ALDs is viewed as a rate function, with 12 to 15 hours regarded as a minimum retention time for maximum alkalinity production (Hedin et al. 1994, Kepler and McCleary 1994). The detention time in a SAPS should be similar and the calculation of the amount of limestone needed should be calculated similarly to that for an ALD.

CONFIGURATION

The configuration of a SAPS is shown in figure 3. At the three SAPSs in Pennsylvania, the depths of the free-standing water are 5 - 6 ft, the depth of the organic layer (spent mushroom compost) is 18 inches, and the depths of the limestone are 18 - 24 inches (Kepler and McCleary 1994). Vertical flow is created by placing the discharge pipes at the bottom of the SAPS.

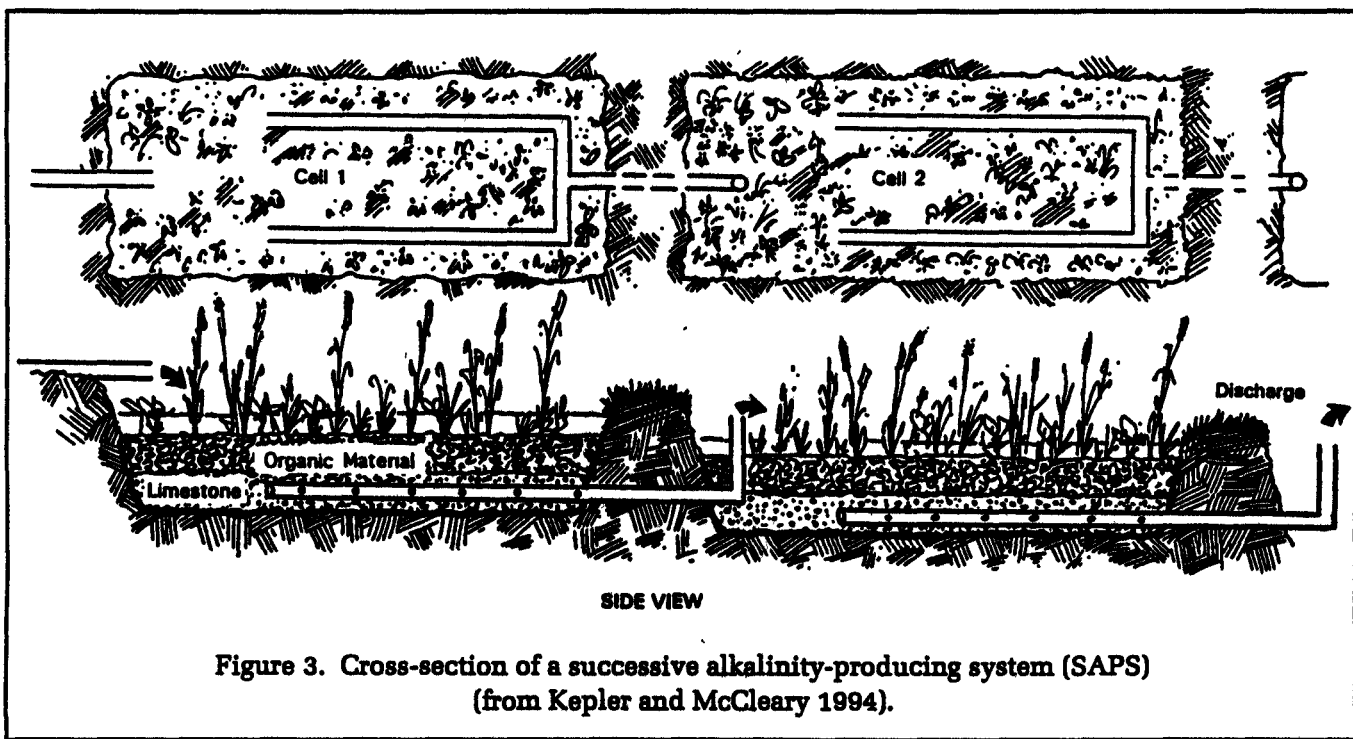
Clogging has not been a problem at any of the three sites. This is thought to be due to the downward pressure of the freestanding water plus the lack of free oxygen in the substrate (Kepler and McCleary 1994).

A SAPS must be followed by a settling pond where Fe precipitation can occur. A vegetated aerobic wetland should be placed after the settling pond to aid in removing suspended solids and to provide a sustainable supply of oxidizable organic matter to a subsequent SAPS.

LONG-TERM PERFORMANCE

SAPS treatment is a recent concept and there are no long-term data. The longevity of effective SAPS treatment will depend upon the amount of limestone available for dissolution and the amount

of alkalinity generated by bacterial activity in the organic layer. Kepler and McCleary (1994) suggest that additional organic material can be added to a SAPS by feeding the SAPS with water from an aerobic wetland.



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GLOSSARY

- abiotic not involving biological processes
- aerobic requiring free oxygen
- algae primitive green plants that live in wet environments
- ALD anoxic limestone drain
- AMD acidic mine drainage
- AML abandoned mine lands
- anaerobic a situation in which molecular oxygen is absent; lacking oxygen
- anoxic without free oxygen
- aquifer a permeable material through which groundwater moves
- aspect the ratio of length to width
- AWMS animal waste management system
- baseflow the portion of surface flow arising from groundwater; the between-storm flow
- biomass the mass comprising the biological components of a system
- biotic the living parts of a system; biological
- BMP Best Management Practice
- BOD biochemical oxygen demand, often measured as 5-day biochemical oxygen demand (BOD₅); the consumption of oxygen by biological and chemical reactions
- CEC cation exchange capacity
- community (plant) the assemblage of plants that occurs in an area at the same time
- denitrification the conversion of nitrate to nitrogen gas through the removal of oxygen
- detritus loose, dead material; in wetlands, largely the leaves and stems of plants
- emergent wetland a wetland dominated by emergent plants, also called a marsh
- EC electrical conductivity
- effluent the surface water flowing out of a system
- emergent plant a non-woody plant rooted in shallow water with most of the plant above the water surface
- ET evapotranspiration
- evapotranspiration loss of water to the atmosphere by evaporation from the water surface and by transpiration by plants
- exfiltration the movement of water from a surface water body to the ground
- exotic species not native; introduced
- HLR hydraulic loading rate; loading on a unit area basis
- HRT hydraulic residence time; average time that moving water remains in a system
- hydric soil a soil that is saturated, flooded, or ponded long enough during the growing season to develop anaerobic conditions in the upper part of the soil
- hydrolysis chemical decomposition by which a compound is resolved into other compounds by taking up the elements of water
- hydroperiod the conversion of ammonia to nitrate through the addition of oxygen
- infiltration the movement of water from the ground into a surface water body
- influent the surface water flowing into a system
- karst irregular, pitted topography characterized by caves, sinkholes, and disappearing streams and springs, and caused by dissolution of underlying limestone, dolomite, and marble
- marsh an emergent wetland
- microbe microscopic organism; includes protozoa, bacteria, yeasts, molds, and viruses

microorganism term often used interchangeably with microbe
native species one found naturally in an area; an indigenous species
nitrification the conversion of ammonia to nitrate through the addition of oxygen
non-persistent plant ... a plant that breaks down readily after the growing season
non-vascular plant a plant without differentiated tissue for the transport of fluids; for instance, algae
NPS nonpoint source
organic matter matter containing carbon
oxidation the process of changing an element from a lower to a higher oxidation state by the removal of an electron(s) or the addition of oxygen
pathogen a disease-producing microorganism
peat partially decomposed plant material, chiefly mosses
perennial plant a plant that lives for many years
permeability the capacity of a porous medium to conduct fluid
persistent plant a plant whose stems remain standing from one growing season to the beginning of the next
redox reduction/oxidation
reduction the process of changing an element from a higher to a lower oxidation state, by the addition of an electron(s)
rhizome a root-like stem that produces roots from the lower surface and leaves, and stems from the upper surface
riparian pertaining to the bank of a stream, river, or wetland
SAPS successive alkalinity-producing system
SF surface flow
SSF subsurface flow
stolon a runner that roots at the nodes
scarification abrasion of the seed coat
stratification treatment of seed by exposure to cold temperatures
succession the orderly and predictable progression of plant communities as they mature
transpiration the process by which plants lose water
tussock a hummock bound together by plant roots, especially those of grasses and sedges
tuber a short thickened underground stem having numerous buds or "eyes"
TSS total suspended solids
vascular plant a plant that possesses a well-developed system of conducting tissue to transport water, mineral salts, and foods within the plant
wrack plant debris carried by water

ABBREVIATIONS AND CONVERSION FACTORS

<i>MULTIPLY</i>	<i>BY</i>	<i>TO OBTAIN</i>
ac, acre	0.4047	ha, hectare
cfs, cubic foot per second	448.831	gpm, gallon per minute
cfs, cubic foot per second	2.8317×10^{-2}	m ³ /s, cubic meter per second
cm, centimeter	0.3937	inch
cm/sec, centimeter per second	3.28×10^{-2}	fps, foot per second
°F, degree Fahrenheit	$5/9 (°F - 32)$	°C, degree Celsius
ft, foot	0.305	m, meter
ft ² , square foot	9.29×10^{-2}	m ² , square meter
ft ³ , cubic foot	2.83×10^{-2}	m ³ , cubic meter
ft/mi, foot per mile	0.1895	m/km, meter per kilometer
fps, foot per second	18.29	m/min, meter per minute
g/m ² /day, gram per square meter per day	8.92	lb/ac/day, pound per acre per day
gal, gallon	3.785	L, liter
gal, gallon	3.785×10^{-3}	m ³ , cubic meter
gpm, gallon per minute	6.308×10^{-2}	L/s, liter per second
ha, hectare	2.47	ac, acre
inch	2.54	cm, centimeter
kg, kilogram	2.205	lb, pound
kg/ha/day, kilogram per hectare per day	0.892	lb/ac/day, pound per acre per day
kg/m ² , kilogram per square meter	0.2	lb/ft ² , pound per square foot
L, liter	3.531×10^{-2}	ft ³ , cubic foot
L, liter	0.2642	gal, gallon
lb, pound	0.4536	kg, kilogram
lb/ac, pound per acre	1.121	kg/ha, kilogram per hectare
m, meter	3.28	ft, foot
m ² , square meter	10.76	ft ² , square foot
m ³ , cubic meter	1.31	yd ³ , cubic yard
m ³ , cubic meter	264.2	gallon, gal
m ³ /ha/day, cubic meter per hectare per day	106.9	gallon per day per acre, gpd/ac
mm, millimeter	3.94×10^{-2}	inch
mi, mile	1.609	kilometer, km