

A REVIEW OF SUBSTRATES AND VEGETATION IN WETLANDS
CONSTRUCTED TO TREAT ACID MINE DRAINAGE

by

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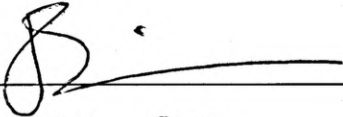
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ABSTRACT

Contamination of streams and rivers by acidic mine water is one of the most persistent industrial pollution problems in the United States. According to a recent survey, over 6450 kilometers (4000 miles) of rivers and streams in the United States are adversely affected by acid mine drainage (Kleinmann 1990). Acid mine drainage (AMD) is generated when pyrite (FeS_2) and other sulfide minerals are exposed to oxygen and water. Dissolved metals (particularly manganese and iron), sulfate (SO_4^{2-}), and acidity (H^+) are the products of the reaction. Federal regulations mandate the treatment of acid mine drainage to meet the following acceptable limits of water quality: a pH between 6 and 9, total iron concentration of less than 3.5 mg/L on a monthly average, and manganese less than 2.0 mg/L on a monthly average (U.S. Code of Federal Regulations 1981). Chemical treatment of mine drainage is often the mode of complying with the law although research suggests that constructed wetlands may be appropriate, less expensive, and very effective in treating weak to moderately polluted acid drainage on a long-term basis (Brodie 1990, Brodie 1991, Schafer 1991).

Treatment mechanisms operating in constructed wetlands are reviewed, as well as the roles of vegetation and substrates in treating AMD. Two treatment methods are recommended for a specific wetland construction project in Challis, Idaho: (1) an anoxic limestone drain combined with wetland cells designed to maximize sulfate reduction processes; and (2) an oxidation pond where precipitation of metals will occur, followed by an aerobic wetland cell for sulfate reduction, and finally a deep, polishing pond primarily for removal of manganese via oxidation processes. Mushroom compost is the recommended substrate for treatment of AMD by sulfate reduction. The selection of wetland vegetation to be planted in constructed wetlands is not of primary importance as long as the species are able to tolerate the local climate and metal concentrations in the AMD.

INTRODUCTION

Acid mine drainage (AMD) is a common problem associated with coal and metal mining. Contamination of streams and rivers by acidic mine water is a major economic and environmental problem in the United States. According to a recent survey, over 6452 km (4000 miles) of rivers and streams in the United States are adversely affected by acid mine drainage (Kleinmann 1990). Constructed wetlands may provide an inexpensive, low maintenance, and potentially long-term alternative to treating AMD with chemical means (Brodie 1990, Brodie et al. 1989b, Perry and Kleinmann 1991, Schafer and Associates 1991, Wieder 1991, Wieder et al. 1985). Over 400 constructed wetland water treatment systems have been built on mined lands (Perry and Kleinmann 1991). While much success has been documented, some constructed wetlands have failed to achieve desired results (Pesavento 1984, Wieder et al. 1985). Only about 20% of the constructed wetland systems in the United States discharge water that completely meet effluent standards (Kastning-Culp and Lockwood 1993, Kleinmann and Hedin 1989, Perry and Kleinmann 1991).

The objective of this paper is to review the principal processes operating in wetlands. The various substrates that have been employed in constructed wetlands will also be reviewed. Substrates will be evaluated as to their success in treating metal contaminants from AMD. Wetland plant materials will be investigated, as well, in terms of treatment of and/or tolerance to mine drainage waters. Finally, a plan for reclamation of a series of constructed wetland cells will be presented for a specific site

in Challis, Idaho.

While both natural and constructed wetlands have been shown to improve water quality, the exact mechanisms of metal removal are not always clear. Previous reviews of the removal processes operating in wetlands suggested that removal by humic material adsorption or through uptake by plants were the primary modes of metal removal (Henrot and Wieder 1989, Tarleton et al. 1984, Wildeman et al. 1993a). Recently, however, it has been suggested that removal through microbial activity, both aerobic and anaerobic, may be the dominant removal mechanism (Perry and Kleinmann 1991, Pesavento 1984, Stone 1984, Wieder et al. 1982, Wildeman et al. 1993a).

Wetland soils or substrates are very important to successful metal removal processes (Kastning-Culp and Lockwood 1993) and particularly in enhancing sulfate reduction in AMD treatment (Nawrot and Klimstra 1990). Inherent neutralization potential within the substrate often allows an initial increase in the pH of the effluent (Schafer and Associates 1991). An initial decrease in metals is also common in wetlands due to sorption mechanisms (Hedin 1991, Kleinmann and Hedin 1989, Wieder et al. 1985, Unz and Dietz 1986). Organic substrates provide sites for bacterial attachment and colonization (Hedin 1989). Wetland substrates provide a source of readily decomposable organic carbon and nitrogen (Nawrot and Klimstra 1990, Schafer and Associates 1991). Finally, the substrate in a wetland supports the growth of aquatic plants, which in turn provide a long-term source of carbon and nutrients for the bacteria essential to treatment processes (Nawrot and Klimstra 1990, Perry and Kleinmann 1991, Schafer and Associates 1991). Introducing alkalinity into the system

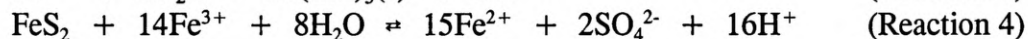
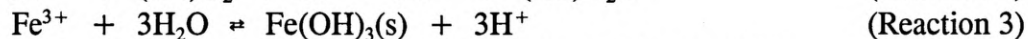
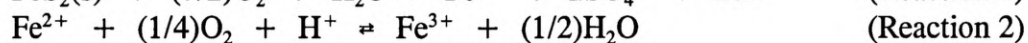
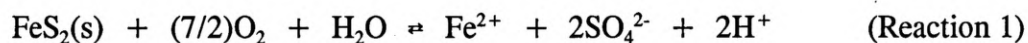
before AMD enters the wetland may provide an environment where sulfate reduction can occur at an optimum rate given an adequate supply of carbon, nitrogen, and other nutrients are available (Brodie 1990, Kastning-Culp and Lockwood 1993, Schafer and Associates 1991). One suggested method of introduction is an anoxic limestone drain (Brodie et al. 1991, Perry and Kleinmann 1991, Turner and McCoy 1990).

Similarly, establishment of wetland plants is critical to successful metal removal. Accumulation of metals by plants varies considerably depending upon the plant and metal in question (Kleinmann and Hedin 1989). *Sphagnum* moss, for example, can accumulate metals but can only tolerate trace amounts over long periods while cattails (*Typha* spp.) can tolerate very high metal concentrations due to physiological mechanisms that precipitate metals in the root zone, preventing uptake and accumulation in plant tissues (Hedin 1991). Uptake of metals by stems and leaves can occur, however, Wildeman et al. (1993a) have estimated that this process only accounts for approximately 1 to 5% of metal accumulation. Similarly, Sencindiver and Bhumbra (1988) found that bioaccumulation of iron in plants accounted for just 0.3% of the annual inflow iron load in a *Typha*-dominated wetland receiving low level AMD,. As with substrates, plants provide sites for bacterial attachment and colonization (Hedin 1989). Metal removal by roots and rhizomes may be significant (Wildeman et al. 1993a). Roots themselves do not accumulate metals, but they may generate microenvironments that promote the reduction and oxidation processes. Emergent plants are also capable of aerating the sediments of a wetland by releasing oxygen from their roots (Hedin 1989). Wetland plant material may also provide the

biomass necessary for other treatment processes (Schafer and Associates 1991, Wildeman et al. 1993). For example, decayed material yields organic matter, which will then be capable of removing metals by adsorption and exchange. Another product of decay is cellulose that provides nutrients for sulfate reducing bacteria. Wetland vegetation is important for stabilizing the substrate and reducing channelization in the surface flow (Howard et al. 1989a, Schafer and Associates 1991).

PRODUCTION OF ACID MINE DRAINAGE

AMD is generated when pyrite (FeS_2) and other sulfide minerals are exposed to oxygen and water. Dissolved metals (particularly iron), sulfate (SO_4^{2-}), and acidity (H^+) are the products of the reaction. Stumm and Morgan (1981) review the chemistry of pyrite weathering as follows:



Iron-oxidizing bacteria such as *Thiobacillus ferrooxidans* catalyze the reaction (Unz and Dietz 1986), which is oxidized by chemical and microbiological processes (Henrot and Wieder 1989). The resultant solution then interacts with other mine waste components in secondary reactions so that AMD may contain high concentrations of calcium and magnesium, as well as more problematic solutes including aluminum, manganese, zinc, copper, and nickel (Barton 1978). The discharge water quality can range from a highly acidic solution with large concentrations of metals and sulfate to a neutralized solution with low metal and sulfate concentrations.

Since 1969, in Appalachia alone, over 12,000 kilometers of streams and rivers have been adversely affected by drainage from abandoned coal mines (Wieder et al. 1989), and this pollution will likely continue unabated for decades (Girts and Kleinmann 1986). Impacts to streams and rivers would be even greater if mines operating since 1978 were not required to chemically treat their effluent water. Water

quality standards in the United States require that mine discharges have a maximum allowable pH between 6 and 9, total iron concentration of less than 3.5 mg/L on a monthly average, and total manganese less than 2.0 mg/L on a monthly average as illustrated in Table 1 (U.S. Code of Federal Regulations 1981).

Table 1. Effluent limitations on mine water pollutants as mandated by the Surface Mining Control and Reclamation Act of 1977 (U.S. Code of Federal Regulations 1981).

Pollutant	Maximum allowable value	Monthly average value
pH	6.0-9.0	6.0-9.0
Total Fe (mg/L)	< 7.0	<3.5
Total Mn (mg/L)	< 4.0	<2.0

Most of the constructed wetlands built to date have been designed to treat acidic coal mine drainage versus the effluent water from metal mines (Howard et al. 1989c, Kleinmann and Hedin 1989, Perry and Kleinmann 1991). The principal metal contaminants in coal mine drainage tend to include iron, manganese, aluminum, (Kleinmann and Hedin 1989, Stark et al. 1990, Perry and Kleinmann 1991), and sulfate (Fennessy and Mitsch 1989, Wildeman and Laudon 1989). Drainages from metal mines are dominated by copper, zinc, cadmium, lead, and arsenic, as well as iron, manganese, and sulfate (Wildeman and Laudon 1989). Since copper, zinc, lead, cadmium, arsenic can be present in coal mine drainage, as well, Wildeman et al.

(1993a) suggests that there is reasonable cause to group all AMDs together including that from eastern and western coal mines and metal mines since the effluents tend to have similar pollutant qualities. Schafer and Associates (1994), however, note that AMD from eastern coal mines tends to contain relatively low acidity, total dissolved solids, and metal concentration due to high infiltration rates compared with conditions found at western mine sites.

COMPARISON OF WETLAND TYPES

There are two basic types of constructed wetlands: those which emphasize aerobic processes and those in which anaerobic processes dominate (Table 2).

Table 2. A comparison of constructed wetland types (adapted from Wildeman et al. 1993a).

AEROBIC	ANAEROBIC
1. Emphasize oxidation 2. Surface water flow 3. Oxide precipitates 4. Processes can lower pH 5. Operate best at pH > 5.5 6. Might freeze in winter 7. Removes Fe quite well	1. Emphasize reduction 2. Subsurface water flow 3. Sulfide precipitates 4. Processes can raise pH 5. Can be effective at pH 2.5 6. Can operate through winter 7. Removes other heavy metals quite well

From Table 2 it can be observed that for treatment of mine drainage, the two types of constructed wetlands compliment each other. Aerobic systems are most applicable to coal mine drainage where seeps have a pH of greater than 5.5 and primarily contain iron and manganese (Wildeman et al. 1993a). Anaerobic systems will work at lower pH and are better capable of removing heavy metals such as copper, zinc, cadmium, and lead; the conditions commonly found in metal-mining operations (Wildeman et al. 1993a). Additionally, aerobic systems will remove contaminants from metal mines when the pH is above 6, and anaerobic systems will remediate severely contaminated coal mine waters with a pH of less than 3 (Wildeman et al. 1993a). A series of

wetland cells that combine both aerobic and anaerobic treatments appears to offer the most chance of successful water quality improvement (Calabrese et al. 1991, Wildeman et al. 1993a). One advantage of a mixed system is the production of dense, relatively stable iron sulfides which provide a renewable metal retention mechanism unavailable in simple aerobic systems (Calabrese et al. 1991, Kleinmann and Hedin 1989, Perry and Kleinmann 1991). Also, sulfate reduction, a removal mechanism associated with anaerobic systems, generates alkalinity to raise the pH of the system (Hendricks 1991, Wieder et al. 1985, Wildeman and Laudon 1989). Additionally, a series type system allows for a failure in one cell without destroying the entire project.

REMOVAL PROCESSES OPERATING IN WETLANDS

As Figure 1 demonstrates, a fairly complex system of chemical reactions is responsible for treatment of acid mine drainage in wetlands. In most of the constructed wetlands, the primary metal removal mechanism appears to be oxidation and hydrolysis reactions that cause dissolved metals to precipitate (Hedin 1989, Kleinmann and Hedin 1989, Perry and Kleinmann 1991, Stone 1984). For example, in a study of metal removal mechanisms in a 10-month-old constructed peat wetland, Wieder et al. (1985) reported that 27% of the accumulated manganese and 93% of the iron accumulation were in oxidized forms.

Bacterial sulfate reduction is another naturally occurring reaction which may be important in the treatment of AMD (Kleinmann and Hedin 1989, Reynolds et al. 1991). Studies at the Big Five constructed wetland, for example, showed a decrease in sulfate concentration of approximately 10-20%, significant decreases in concentrations of iron, zinc, and copper, and an increase in pH due to sulfate reduction (Machemer and Wildeman 1991, Reynolds et al. 1991). The processes of adsorption, ion exchange, and complexation with organic matter can also play a role in metal removal. Algae and plants bioaccumulate metals to varying degrees through uptake and filtration processes.

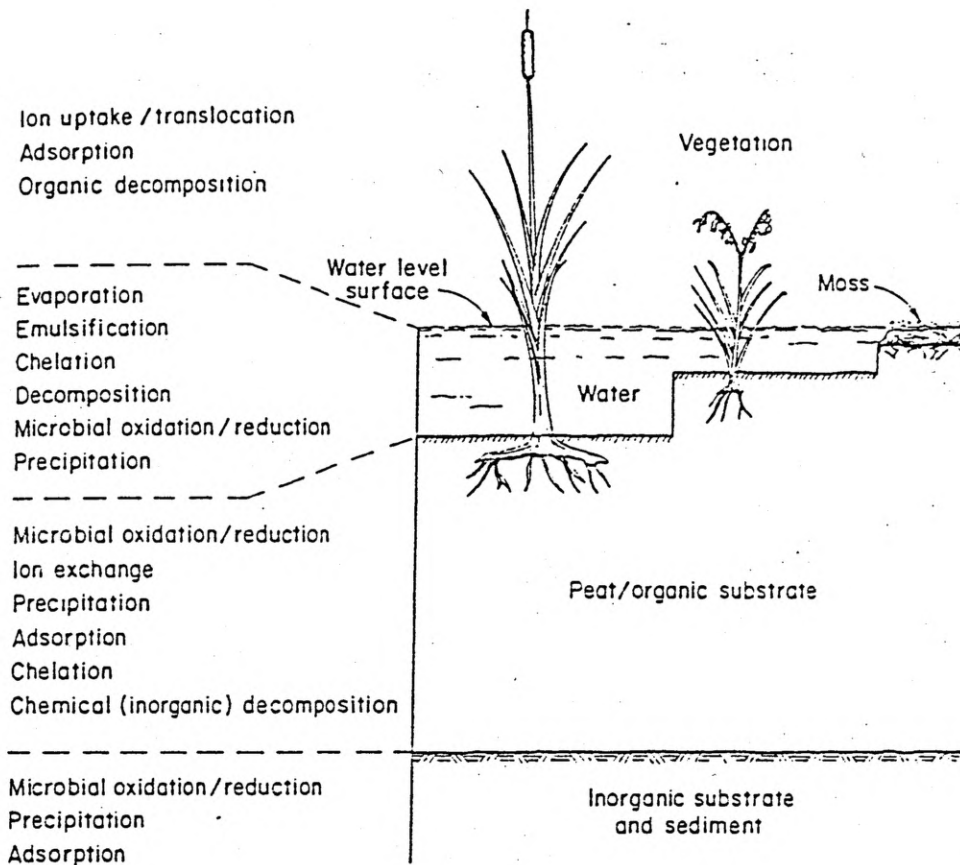


Figure 1. A complex and inter-related system of chemical transformations, much of which are biologically mediated, is responsible for the effective treatment of acid water in wetlands (Kleinmann and Girts 1986).

While researchers disagree on the relative importance of each of the removal processes operating in wetlands, Klusman and Machemer (1991) suggest the following sequence (in order from most to least important):

- (1) cation exchange of metals by an organic-rich substrate, which is usually peat in natural wetlands;
- (2) sulfate reduction with precipitation of iron and other sulfides;
- (3) precipitation of ferric and manganese hydroxides by oxidation;
- (4) adsorption of metals by ferric oxyhydroxides; and
- (5) metal uptake by living plants.

Others would add the following processes (Wildeman et al. 1993a):

- (6) filtering of suspended and colloidal material from the water;
- (7) neutralization and precipitation through the generation of NH_3 and HCO_3^- ; and
- (8) adsorption or exchange of metals onto algal material.

It is generally a combination of the above mentioned processes that results in an improvement in water quality when AMD flows through a constructed wetland.

Reactions with Organic Matter

Organic matter can remove metals from solution by adsorption, cation exchange (Watson et al. 1989), and complexation reactions (Hedin 1991). Adsorption can occur by covalent or electrostatic bonding and includes both organic complexation and cation exchange capacity (CEC) reactions. Organic complexation is pH dependent and includes covalent or charge-related reactions, while CEC measures electrostatic bonding.

The ability of organic matter to accumulate metals is limited by a finite number of reaction sites (Hedin 1991, Henrot and Wieder 1989, Watson et al. 1989). In fact, the capacity for organic substrates to remove metals is probably exhausted within several months of exposure to AMD (Wieder 1988). Organic matter is composed of humic and fulvic acids which are in turn composed of carboxyl and phenolic acid groups. The ion exchange properties of humus are attributed to carboxyl groups found in humic acids of peats and pectic compounds of plant cellular tissue (MacCarthy, P.

cited in Howard et al. 1989b). Exchange processes can be symbolized in the following reactions (Klusman and Machemer 1991).



Where: R = the organic portion of humic or fulvic acid
 COOH = a carboxylic group
 COO⁻ = a carboxyl ion
 M²⁺ = a metal ion in solution

Efficient complexation begins between pH 4 and 6 depending on the metal ion (Wildeman et al. 1993a). Manganese, zinc, and cadmium are difficult to remove by organic adsorption (Wildeman et al. 1993a) conceivably due to the preferential retention of other cations such as Fe²⁺. Iron adsorption onto peat occurs rapidly and the adsorbed iron is not water soluble. Until iron adsorption capacity is reached, iron adsorption will most likely be the principal process involved in iron removal from solution and accumulation in peat (Henrot and Wieder 1989, Tarleton et al. 1984). However, once iron adsorption capacity has been reached, iron oxide formation may become progressively more important (Henrot and Wieder 1989, Wieder and Lang 1986, Wieder et al. 1985). Wieder and Lang (1986) have estimated that adsorption and organic complexation can remove 33 mg of iron per gram of *Sphagnum* (dry weight). During the first few months that a new wetland system is in operation, adsorption and complexation by the substrate helps compensate for limited initial biological activity (Kleinmann and Hedin 1989, Perry and Kleinmann 1991). Once these capabilities are exhausted by saturation of the exchange sites, only the addition of new organic materials supplements this mechanism of metal removal (Kleinmann and Hedin, Perry

and Kleinmann 1991). This level of productivity is insufficient to remove a significant amount of metals in currently sized wetlands (Perry and Kleinmann 1991). However, organic exchange sites may serve as temporary retention of metal cations on the substrate of a wetland, thereby increasing the residence time for microbially-mediated removal processes (Wildeman et al. 1993a).

Sulfate Reduction Catalyzed by Bacteria

Bacterial sulfate reduction is a naturally occurring process that transpires only in the absence of oxygen and in the presence of sufficient organic carbon and sulfate (McIntire and Edenborn 1990, Perry and Kleinmann 1991, Rice and Rabolini 1972). Organic matter decomposition promotes the establishment of anaerobic conditions conducive to heterotrophic bacteria such as *Desulfovibrio* and *Clostridium* which convert sulfuric acid to hydrogen sulfide (Guertin et al. 1985, Rice and Rabolini 1972). Organic carbon resulting from decomposing wetland plants provides an electron source, or energy, for the sulfate-reducing bacteria (Nawrot and Klimstra 1990). The net by-products of sulfate reduction are hydrogen sulfide and bicarbonate, which can precipitate metal sulfides and neutralize acid water, respectively (McIntire and Edenborn 1990, Wildeman and Laudon 1989). Carbonate, a product of sulfate reduction, causes the sediments to have a pH of between 5 and 8 (Hedin et al. 1988a). By catalyzing the formation of iron sulfides, the sulfate-reducing bacteria cause iron in mine drainage to revert back to its original form: pyrite (Hedin 1989). This process

may be the most efficient of all iron removal mechanisms; in fact, Hedin et al. (1988b) documented that for every 48 g of carbon oxidized by sulfate-reducing bacteria, enough sulfide is generated to precipitate 56 g of iron as pyrite.

The application of microbiological processes for acid water treatment through sulfate reduction has been recognized for almost twenty years, yet it is only recently that these principles have been applied to treatment systems in the eastern coal fields (Nawrot and Klimstra 1990). Currently, at over 300 mine sites in the bituminous coal region of the eastern United States, AMD is being treated biologically in constructed wetlands (Hedin 1989, McIntire et al. 1990).

Results from microbiological tests of the substrate at the Big Five constructed wetland in Idaho Springs, Colorado indicate that sulfate-reducing bacteria are present in large numbers (Batal et al. 1989). Stone (1984) has correlated numbers of sulfate reducing bacteria with removal efficiency of metals in wetlands. Kuyucak et al. (1991) found that acclimatization was necessary for sulfate-reducing bacteria to tolerate high concentrations of metals and low pH; however, once the bacteria were established in a substrate they were very resistant to system perturbations such as pH changes. Likewise, Kleinmann and Hedin (1989) have reported that sulfate-reducing bacteria are tolerant of high metal concentrations including nickel, lead, and zinc concentrations as high as 60 mg/L.

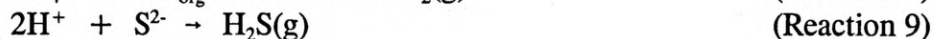
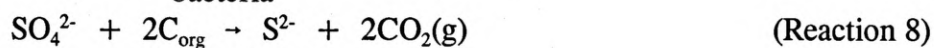
Biological sulfate reduction and the subsequent precipitation of insoluble iron sulfides and/or the evolution of reduced sulfur gases from the wetland soil may be the most important processes involved in modifying the AMD (Kleinmann and Hedin

1989, Wieder and Lang 1982) although the relative importance was uncertain for some time due to conflicting observations in *Sphagnum* wetlands and constructed *Typha* wetlands (Kleinmann and Hedin 1989, Perry and Kleinmann 1991). In reducing sulfate to sulfide, insoluble iron sulfide precipitates in the forms of FeS and FeS₂ are generated and hydrogen ions are removed from the solution in the formation of H₂S (Wieder and Lang 1982). The generation of H₂S consumes hydrogen ions, thus sulfate reduction results in an increase in pH (Hendricks 1991, Wieder et al. 1985). Rice and Rabolini (1972) suggest that sulfate reduction is most effective when a pH of > 5.5 is maintained and temperatures are above 10°C, although sulfate reduction has been demonstrated in solutions with a pH as low as 3.5 (Batal et al. 1989, Tuttle et al. 1969). Sulfate reduction requires pH greater than 5 according to Silver (1989) and is most efficient at a pH of about 7 (Reynolds et al. 1991). Kuyucak et al. (1991) found the best reaction rate was obtained at 35°C and that conducting sulfate reduction experiments at 4°C caused up to 60% decrease in the activity of sulfate reducing bacteria.

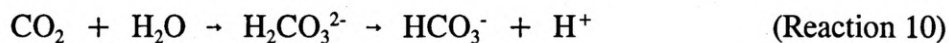
The long-term implications of bacterial sulfate reduction are potentially significant (Fennessy and Mitsch 1989, Hedin 1988, Kleinmann and Hedin 1989, Perry and Kleinmann 1991). Iron sulfide precipitates resulting from sulfate reduction are dense and form within the substrate unlike the large amounts of sludge which blankets the substrate as a result of oxidation and hydrolysis of iron (Kleinmann and Hedin 1989, Perry and Kleinmann 1991). Again, unlike bacterial oxidation, sulfate reduction actually consumes acidity (Wieder and Lang 1982). Sulfate reducers were discovered

throughout the wetland substrate while metal-oxidizing bacteria were found only very near the surface (Batal et al. 1989). Additionally, hydrogen sulfide, a product of sulfate reduction, reacts with many metals to form virtually insoluble sulfide compounds, where bacterial oxidation processes are ineffective in removing most heavy metals (Perry and Kleinmann 1991). Work by McIntire et al. (1990) indicates that metals such as cadmium, lead, copper, mercury, nickel, and zinc can be biologically precipitated as insoluble sulfides. Similarly, using pilot-scale biological reactor systems, Dvorak et al. (1991) lowered concentrations of iron, zinc, manganese, nickel, and cadmium by over 95% as the metals were precipitated as insoluble sulfides by reactions with bacterially generated H_2S . In addition to those metals mentioned above, Kuyucak et al. (1991) have found that arsenic can also be precipitated at low pH (pH 3 to 5).

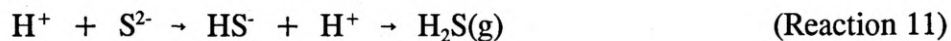
The processes involved in sulfate reduction are summarized below (Postgate 1984).



The CO_2 is converted in an aquatic environment into bicarbonates when $7 < \text{pH} < 9$ and into carbonate when $\text{pH} > 11$, as shown below (Kuyucak et al. 1991).



Hydrogen ions released are utilized in the system to form HS^- and H_2S , according to the following equation (King et al. 1974).



The following equations illustrates how hydrogen sulfide then reacts with many contaminant metals to form insoluble metal sulfides (Dvorak et al. 1991).



where M^{2+} represents metal cations in solution such as iron, zinc, manganese, nickel, cadmium, copper, and lead; and MS represents the insoluble metal-sulfide complexes.

Precipitation of sulfide metals will only continue if there is a sufficient supply of organic carbon. Thus, if seasonal decay of emergent plant species and the excretion of organic compounds by plants is inadequate, periodic additions of organic carbon may be necessary (Dollhopf et al. 1988, Hedin 1989, Kleinmann and Hedin 1989, Perry and Kleinmann 1991, Pesavento 1984, Reynolds et al. 1991). In a study comparing a natural and constructed wetland, Dollhopf et al. (1988) found that while sulfate-reducing bacteria were present in the man-made wetland, microbial biomass and densities were low, probably due to insufficient levels of carbon and other nutrients. The addition of sodium lactate, a preferred carbon source of many sulfate-reducing bacteria, greatly increased the rate of sulfate reduction activity and nickel precipitation in experiments conducted to test the potential for nickel removal from metal mine wastewater (McIntire et al. 1990). Hay extract was found to enhance the activity of sulfate-reducing bacteria approximately five times, and sodium lactate increased the rate by 2.5 times, the rate of sulfide production when no nutrient amendment was added in the Big Five constructed wetland in Idaho Springs, Colorado (Reynolds et al. 1991).

Three factors are critical for sulfate reduction to occur and for the products to remain stable: (1) drainage must flow through the anaerobic portion of the wetland; (2) the wetland must be continuously flooded so the products of reduction are not oxidized (Wildeman et al. 1993a); and (3) a continuous supply of sufficient organic matter must be available to maintain an anaerobic environment and to provide the necessary nutrients for sulfate-reducing bacteria (McIntire et al. 1990, Nawrot and Klimstra 1990, Perry and Kleinmann 1991). Because sulfate reduction usually proceeds more slowly than iron oxidation, compost-based wetlands must be larger (Perry and Kleinmann 1991).

Most constructed wetlands containing an organic substrate and receiving sulfate-enriched mine drainage develop an anaerobic zone in which sulfate reduction becomes established. However, since the major flow path of water in these wetlands generally bypasses the anaerobic zone, the mine drainage is minimally affected by sulfate reduction (Dvorak et al. 1991, Hedin et al. 1989, McIntire et al. 1990, Watson et al. 1989). Designing wetlands to increase flow through the substrate may result in more effective and consistent levels of sulfate reduction and sulfide retention.

Oxidation and Oxyhydroxide Precipitation Catalyzed by Bacteria

In most constructed wetlands, the major mechanism responsible for metal removal appears to be oxidation of iron by bacterial catalysis (Brodie et al. 1991, Kleinmann and Hedin 1989, McIntire et al. 1990, Perry and Kleinmann 1991). In a

study of chemistry within a 10-month-old peat constructed wetland, Wieder et al. (1985) found that 93% of the accumulated iron and 27% of the accumulated manganese was in oxidized forms. Most commonly, Fe^{2+} makes up the vast majority of the total iron load at the source of mine drainage (Kepler 1990). Exposure to the atmosphere, retention time, and a moderately high pH (greater than 6.0) allows iron to rapidly oxidize and precipitate as an oxyhydroxide ($\text{Fe}(\text{OH})_3$) (Kepler 1990) by both abiotic and biotic processes (Hedin 1989).

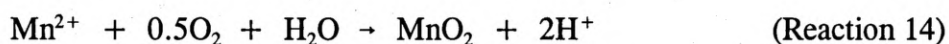
The oxidation of ferrous iron to ferric iron and the subsequent precipitation of iron oxyhydroxide floc is depicted in the following reaction (Hedin 1989):



Ferrous iron tends to autoxidize in aerated solutions when the pH is greater than 6; in more acidic water naturally-occurring bacteria catalyze the reaction (Kleinmann and Crerar 1979, Singer and Stumm 1970). These bacteria increase the ferrous iron oxidation rate by several orders of magnitude and enhance the formation of iron precipitates in acid waters (Hedin 1989). Iron-oxidizing bacteria can be quite important in treating acid mine drainage since most mine discharges have pH between 3.0 and 5.0 (Hedin 1989). Stone (1984) has positively correlated numbers of iron- and manganese-oxidizing bacteria from experimental wetlands with iron and manganese removal, respectively. Generally, iron-oxidizing bacteria are capable of meeting their nutritional needs in acid mine drainage (Hedin et al. 1989) and are able to oxidize iron at very low oxygen concentrations (Nealson 1983).

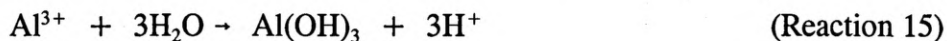
Both autotrophic (e.g., *Thiobacillus ferrooxidans* and *Gallionella* spp.) and

heterotrophic (e.g., *Sphaerotilus* spp. and *Leptothrix* spp.) bacteria can oxidize iron (Henrot and Wieder 1989). Iron-oxidizing bacteria were present on the surface and at depth in all substrates tested at the Big Five constructed wetland after two months of mine drainage flow (Batal et al. 1989). Bacteria which are capable of oxidizing manganese are also present but generally have negligible effect on manganese concentrations unless the pH is above 6 (Perry and Kleinmann 1991). The oxidation of manganese by *Pseudomonas* spp. and *Citrobacter freundii* occurs as follows (Batal et al. 1989):



Studies by Gregory and Staley (1972) suggest that the ability to oxidize manganese is widespread among freshwater heterotrophic bacteria. Marshall (1979) reported that manganese autoxidation does not take place at measurable rates at pH below 8.5, and Owens (1963) documented oxidation rates are very slow in water with a pH of less than 9.4. Manganese oxidation may also be inhibited by high concentrations of manganese; for example, *Arthrobacter* spp. is inhibited at manganese concentrations above 54 mg/L (Bromfield and David 1976).

In general, aluminum is only removed when the pH is raised sufficiently to initiate hydrolysis (Kleinmann and Hedin 1989, Perry and Kleinmann 1991). Hedin (1991) provides the following reaction responsible for aluminum hydrolysis.



Kleinmann and Hedin (1989) have found that oxidation through bacterial activity will not remove metal mine drainage contaminants such as zinc, nickel, lead and cadmium.

Efficient removal of iron and manganese by hydroxide precipitation requires oxidation which is most readily achieved in wetlands by microbial catalysis in the aerobic zone (Wildeman et al. 1993a). The *Ferrooxidan* family of bacteria are responsible for most oxidation of iron (Unz and Dietz 1986). Iron oxidation occurs at an acceptable rate at pH values near 7, while manganese oxidation requires a pH of above 9.0 for an acceptable rate (Wildeman et al. 1993a). It should be noted, however, that wetlands constructed by the Tennessee Valley Authority (TVA) have successfully removed manganese when the pH was below 9.0. Design of successful TVA wetlands require that the loading factor for Mn be 7 m²/mg/min if the pH is less than 5.5 and 2 m²/mg/min when the pH is greater than 5.5 (Brodie et al. 1988). Oxidation of manganese is catalyzed by such heterotrophic genera as *Arthrobacter*, *Pseudomonas*, *Citrobacter*, *Leptothrix pseudoochraceae*, and *Metallogenium* (Silver 1989). The higher the percentage of manganese oxidizers, the more manganese removed (Pesavento 1984). Stone (1984) has found a correlation between numbers of bacteria that can oxidize iron and manganese and removal of these metals. Pesavento (1984) has been able to transplant manganese oxidizers from a natural wetland and induce them to remove 150% more manganese in a man-made wetland that contained 350% more manganese than the wetland they were removed from.

Although iron oxidation and hydrolysis processes are effective at removing much of the iron from AMD, these processes do nothing to help raise the pH of the water (McIntire and Edenborn 1990). In fact, the pH can be lowered by oxidation and hydrolysis; many constructed wetlands with circumneutral pH and iron contaminated

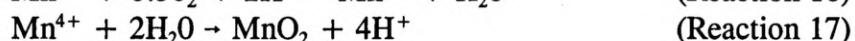
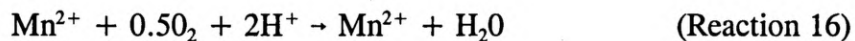
inflow water actually produce water with a lower pH due to the generation of hydrogen ions (Brodie et al. 1988, Kepler 1990). Brodie et al. (1991) and others (Perry and Kleinmann 1991, Turner and McCoy 1990) recommend pretreating AMD before it enters the wetland by installing an anoxic limestone drain (ALD) to add alkalinity into the system. The ALD consists of an open, unlined trench or excavation backfilled with .61 to 1.52 m (2 to 5 feet) of gravel-sized (1.9 to 3.81 cm diameter crushed rock), crushed, high-calcium limestone (i.e., $\geq 90\%$ available CaCO_3) and covered with 20-mil plastic and .61 m (2 feet) of soil to prevent oxygen infiltration (Brodie et al. 1991). Diverting the drainage next through a deep, aerobic wetland cell will allow precipitation of iron (Brodie et al. 1991). Since dissolution of limestone commonly results in pH of 6 to 8, a significant secondary effect of its contact with AMD can be increased rates of abiotic iron oxidation and hydrolysis (Hedin 1989).

While they can be successful, ALD failures have been noted in the literature (Schafer and Associates 1994). If AMD is highly oxidized before entering the ALD, the drain may become plugged due to aluminum and ferric iron precipitation (Schafer and Associates 1994).

It should be noted that sulfate reduction and metal oxidation are mutually exclusive processes where one requires aerobic and the other anaerobic conditions. Perry and Kleinmann (1991) recommend that aerobic wetland flow be staged first to remove as much iron as possible through oxidation; the flow can then be directed through cells in which the water level remains within the substrate (compost) to raise the pH to near neutral. A final aerobic stage can then be utilized, if necessary, to

remove manganese (Brodie 1990, Perry and Kleinmann 1991), although much controversy surrounds the success of this treatment. Some researchers maintain that a pH of 9.0 is required for successful oxidation of Mn (Wildeman et al. 1993a), while others (Brodie et al. 1988) report that by adjusting the loading factor, treatment of Mn is possible even when the pH is below 5.5. A second option for introducing alkalinity is to pretreat AMD by flow through an anoxic limestone-filled drain (as described above) before it enters the wetland (Brodie et al. 1991, Perry and Kleinmann 1991, Turner and McCoy 1990); up to 100 to 200 mg/L of alkalinity can be added to mine drainage with this treatment technique (Perry and Kleinmann 1991).

Dissolved manganese in mine drainage exists in its reduced 2^+ state; however, in highly alkaline aerobic environments, manganese is oxidized to the 4^+ state, which is insoluble and precipitates rapidly as an oxide such as MnO_2 (Hedin 1991). The following equations depict the manganese reactions:



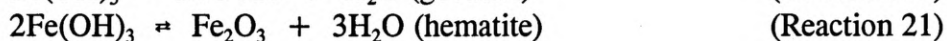
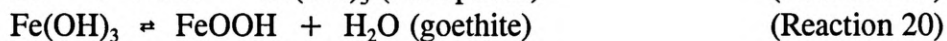
For simple aerobic wetlands it has been recommended that approximately 100 m² of wetland is necessary to remove each kg/day of iron; about 500 m² of wetland is required to remove each kg/day of manganese (Hedin and Nairn 1990, Perry and Kleinmann 1991).

Adsorption of Metals by Oxyhydroxides

When Fe (III) and aluminum precipitate as hydroxides and Mn (IV) precipitates as an oxyhydroxide, they have a strong capability of adsorbing other metal ions onto their surfaces (Stumm and Morgan 1981, Wildeman et al. 1993a). Aluminum hydroxides form as follows (Dvorak et al. 1991):



For iron, the reaction sequence is as follows (Lindsay 1979):



The amorphous hydroxide is the first form to precipitate. In moist environments it turns into goethite. Both hematite and goethite have better adsorbing capabilities for trace metals than does the amorphous hydroxides (Stumm and Morgan 1981). Stumm and Morgan (1981) theorize that oxyhydroxides adsorb metals to their surface due to their surface charge. The surface operates like a weak acid, attracting hydroxide ions, thus creating a negative surface charge. The negative surface of the particles then attracts positively charged metal ions. Wetlands containing high concentrations of organic matter and high iron loads, encourage large aggregates of iron flocculate to form and precipitate (Kepler 1990). Thus, if oxidation and precipitation occur, a side benefit is further removal by the precipitates (Wildeman et al. 1993a).

Bioaccumulation of Metals by Plants

Accumulation of metals by plants varies considerably depending upon the metal and the plant in consideration (Kleinmann and Hedin 1989), and researchers report varying degrees of success. Steiner and Freeman (1989) state that nutrient assimilation by plants is insignificant and that the nutrients are released when the aboveground vegetation dies seasonally (Brix 1987, Gersberg et al. 1986). Kleinmann and Hedin (1989) maintain that although potentially significant in slightly contaminated waters, bioaccumulation of metals is biomass-limited. Recent studies have shown that uptake by stems and leaves of plants accounts for only 1 to 5% of metal accumulation (Howard et al. 1989, Wildeman 1993).

Various wetland plant species have differing capabilities for accumulating metals (Kastning-Culp and Lockwood 1993). *Sphagnum* moss, for example, can accumulate metals but can tolerate only trace amounts over long periods (Hedin 1991). Spratt and Wieder (1988) found that while *Sphagnum* has a tremendous ability to accumulate iron, if exposed to even moderate concentrations, it absorbs so much metal that the moss petrifies. Chan et al. (1982) discovered several plants capable of bioaccumulating metals: *Salicornia pacifica* (pickleweed) removed 0.016 kg copper/ha, *Phalaris arundinacea* (reed canary grass) removed 0.69 kg copper/ha and *Carex stricta* removed up to 0.67 kg nickel/ha. At the Big Five Tunnel constructed wetland in Colorado, cadmium and lead concentrations in both the leaves and roots increased in *Typha latifolia*, *Carex aquatilis*, and *C. utriculata* over the course of two growing

seasons (Wildeman et al. 1993a). Martin (1976) suggests that *Eichhornia crassipes* (water hyacinths) are effective in removing lead, mercury, and other metals, and are capable of concentrating metals as much as 10,000 times that of the ambient solution concentration.

The best metal accumulators for copper were found to be *Pohlia annotina* (a moss) accumulating 3,032 times the level of Cu in water, and 48 times the amount in soil; *Deschampsia cespitosa* (a grass) accumulated 1,979 times the amount in water and 31 times the amount in soil; *Pohlia wahlenbergii* (a moss) accumulated 14,813 times the amount of copper in water and 7 times the level in soil; and *Senecio fremontii* (a forb) accumulated 31 times the amount in water and 5 times the soil level (Kastning-Culp and Lockwood 1993).

The greatest zinc accumulators were found to be *Pohlia wahlenbergii* (a moss), which accumulated 3,814 times the level in water and 10 times the soil level; *Equisetum arvense* (a horsetail) accumulated 1,120 times the level in water and 8 times the amount present in the soil; *Poa interior* (a grass) accumulated 2,128 times the amount in water and 6 times the background soil level; and *Agrostis exarata* (a grass) accumulated 261 times the level of zinc in water and 4 times that in the soil (Kastning-Culp and Lockwood 1993).

For lead, the best accumulators were *Carex microptera* (a sedge) which accumulated 5,954 times the amount in water and 2,977 times the soil level and *Poa*

interior (a grass) which accumulated 5,347 times the amount in water and 2,674 times the amount in soil (Kastning-Culp and Lockwood 1993). The most successful cadmium accumulator was *Pohlia wahlenbergii* (a moss), which contained 6,393 times the level of cadmium in the water and 2 times the soil level (Kastning-Culp and Lockwood 1993).

Arsenic accumulators include *Pohlia wahlenbergii* (a moss), accumulating 3,221 times the amount of As in water and 1,073 times the amount in soil; *Bryum lisae* (a moss) accumulated 6,443 times the amount in water and 1,073 times the soil level; *Epilobium glaberrimum* (a forb) accumulated 5,814 times the amount in water and 969 times the soil level; and *Carex scopulorum* (a sedge) contained 26,432 times the amount of arsenic in the water and 113 times that in the soil (Kastning-Culp and Lockwood 1993). Overall, one of the best metal accumulators is *Pohlia wahlenbergii* (Kastning-Culp and Lockwood 1993).

One of the difficulties of relying heavily on plant uptake of metals is the relatively brief turnover time resulting in short-term storage, particularly with floating aquatic macrophytes (Reddy and DeBusk 1987). If metal accumulation by wetland vegetation is significant, plant harvest may be required to provide an ultimate means of metal removal from the system (Blake et al. 1987, Dierberg et al. 1987, Reddy and DeBusk 1987).

Algal Accumulation of Metals

Algae are capable of accumulating iron, manganese (Kleinmann and Hedin 1989, Perry and Kleinmann 1991), copper, lead, and zinc (Gale 1985 as cited in Kleinmann and Hedin 1989, Kalin 1988). Kepler (1986), for example, found concentrations as high as 56,000 mg/kg of manganese (dry weight) in samples of *Oscillatoria* algae. The Bureau of Mines has documented *Microspora* algae samples from a constructed wetland containing up to 85,000 $\mu\text{g/g}$ manganese (Hedin 1989). The most commonly encrusted genera of algae growing in acid mine drainage include *Oedogonium*, *Mougeotia*, and *Microspora* (Stevens et al. 1989). The encrustations generally contain mostly iron if the iron concentration in the AMD is moderate to high regardless of the manganese concentration; however, if the iron concentration is low, encrustations contain manganese (Stevens et al. 1989). Algae are also capable of concentrating uranium, zinc, copper, nickel, and radium-226 under alkaline conditions (Wildeman and Laudon 1989, Van Everdingen and Krouse 1988). Since algal biomass in wetlands is very limited, algal accumulation of metals will rarely reach significant levels (Hedin 1989, Kleinmann and Hedin 1989, Perry and Kleinmann 1991), thus cannot be considered a solution to treating AMD.

VEGETATION

While some researchers claim that metal uptake by wetland vegetation can be significant (Blake et al. 1987, Chan et al. 1982, Kastning-Culp and Lockwood 1993, Martin 1976), others maintain this mode of AMD treatment is minimal (Fennessy and Mitsch 1989, Howard et al. 1989, Sencindiver and Bhumbra 1988, Wildeman 1993). Nonetheless, vegetation is an important component of wetlands. Removal of iron and manganese has been directly linked with the density of vegetation in a wetland (Brooks 1984). Some of the roles plants play in AMD treatment in wetlands include providing sites for bacterial attachment and colonization; providing the biomass necessary for the other treatment processes (i.e., providing a carbon source for microbes); stabilizing the substrate; reducing channelization in the surface flow; and plant roots generate microenvironments, which promote the reduction and oxidation processes. Emergent plants are also capable of aerating the sediments of a wetland by releasing oxygen from their roots.

The concept of constructing wetlands to treat AMD has its origin in observations that natural *Sphagnum* peat wetlands receiving AMD appear to clean the polluted water (Hedin 1989, Lapakko and Eger 1988, Wieder and Lang 1986). For example, through adsorption and complexation processes, Wieder and Lang (1986) reported that *Sphagnum* removed 33 mg of iron per gram (dry weight) of moss. Since the discharge of AMD into natural wetlands is prohibited, it has been necessary to construct wetlands that act as water treatment systems. Historically, most wetland

construction projects mimicked the original *Sphagnum* peat wetlands (Burris et al. 1984, Hedin 1989, Kleinmann et al. 1983). In spite of promising laboratory results (Burris et al. 1984, Gerber et al. 1985, Kleinmann et al. 1983), virtually all field tests of the *Sphagnum*-dominated constructed wetlands failed to provide sufficient water treatment for more than a few months (Hedin 1989, Perry and Kleinmann 1991). It was found that *Sphagnum* was quite sensitive to the stresses associated with transplanting, abrupt changes in water chemistry, excessive or insufficient water depth, and excessive accumulations of iron (Hedin 1989, Perry and Kleinmann 1991).

It was also observed that *Typha* was present in natural, acidic wetlands, seeps, and ponds; subsequently, *Typha*-dominated wetlands were constructed to treat coal mine drainage. Currently, over 400 *Typha*-dominated wetlands have been constructed in the bituminous coal region of the eastern United States (Perry and Kleinmann 1991).

Girts and Kleinmann (1986) and Kleinmann and Girts (1986) suggest wetland performance increases in the second year after construction as vegetation becomes better established. Since most experimental wetlands constructed to evaluate water treatment performance have been observed for the short term, the findings by Girts and Kleinmann (1986) and Kleinmann and Girts (1986) may in part explain why many constructed wetlands have failed to improve water quality to a significant extent.

In a survey of the effectiveness of constructed wetlands built prior to 1986, Girts and Kleinmann (1986) found that constructed wetlands consistently neutralize acidity and remove iron (50 to 96% removal at most sites), but moderation of manganese and sulfate concentration is less effective than in some natural systems. In

a survey of water chemistry in constructed wetlands, Girts and Kleinmann (1986) found hydrogen and iron ion removal ranged from 80 to 96%, acidity decreased from 68 to 76%, and manganese and sulfate were reduced 22 to 50%. While regulation of pH is variable, the application of crushed limestone can help reduce the hydrogen ion concentration by as much as 90% (Girts and Kleinmann 1986). Preliminary results from an evaluation of wetlands constructed to treat AMD indicate that the wetlands dominated by emergent species are out-performing the *Sphagnum*-dominated wetlands (Kleinmann and Girts 1986). A review of constructed wetlands conducted by the Bureau of Mines revealed that in general the wetlands were not effectively lowering the manganese concentrations of AMD (Hedin 1989). Pesavento (1984) suggests that manganese removal from *Sphagnum*-dominated wetlands may not be appreciable. Stillings et al. (1988) found that even a *Typha*-dominated system was unable to successfully remove Mn from AMD. There is no record of any type of wetland removing more than 40 mg/L manganese with any rate of flow (Pesavento 1984).

Sphagnum

While opinions vary, some researchers maintain that *Sphagnum* moss has a well-established capability to accumulate iron (Burris et al. 1984, Gerber et al. 1985, Guertin et al. 1985, Wenerick et al. 1988). However, *Sphagnum* has proven to be difficult to establish in constructed wetlands (Hedin 1991, Hellier 1989, Girts and Kleinmann 1986, Kleinmann and Girts 1987). With the exception of low flow, low

concentration environments (iron < 10 mg/L), toxic levels of metal accumulation in plants are reached within a single growing season (Spratt and Wieder 1988) or even just several months (Hedin 1989, Perry and Kleinmann 1991). Kleinmann and Girts (1987) reported that high levels of iron (> 100 mg/L) kill *Sphagnum* due to the high adsorption rate of the moss. Since *Sphagnum* is an acid-producing genus, it has no significant effect on raising the pH of mine drainage (Fennessy and Mitsch 1989) and in some cases actually lowers effluent pH due to the generation of H⁺ (Perry and Kleinmann 1991, Turner and McCoy 1990). In addition, the legal discharge requirements for a pH between 6 and 9 is not consistent with the preservation of the naturally acidic *Sphagnum*-dominated wetlands (Wieder et al. 1984).

Sphagnum species vary in their tolerance to pH but *Sphagnum recurvum* can lower effluent concentration of iron, manganese, calcium, magnesium, and sulfate even at a pH of 2.5 (Girts and Kleinmann 1986, Unz and Dietz 1986). In disagreement, Harris et al. (1984) have shown that while *Sphagnum* may be tolerant of low pH, iron adsorption diminishes rapidly below pH 2.5. Pesavento (1984) and others (Gerber et al. 1985, Guertin et al. 1985, Harris et al. 1984, Kleinmann et al. 1985, Wieder and Lang 1986) have been unable to remove manganese from wetlands planted primarily with *Sphagnum* moss. Aluminum concentrations are also not significantly affected by *Sphagnum* moss (Kleinmann 1985).

Kleinmann et al. (1983) reported up to 97% of the ferrous iron and up to 72% of the total iron were removed in a constructed wetland loosely packed with transplanted *Sphagnum* moss; cation exchange processes were attributed to the

successful treatment. Successful treatment of acid mine drainage has also been witnessed in the Tub Run Bog (a natural *Sphagnum*-dominated wetland) via biological sulfate reduction as evidenced by reductions in hydrogen ions, iron, and sulfate and by the production of H_2S , FeS , and FeS_2 ; however, it should be noted that this is a natural wetland (Wieder et al. 1982b). For constructed wetlands, it has been very difficult, if not impossible, to achieve significant sulfate reduction in man-made *Sphagnum*-dominated systems. It is likely that this failure can be attributed to the acidic conditions present in *Sphagnum* systems; Schafer and Associates (1991) found in a laboratory experiment that very limited sulfate reduction occurred in acidic substrates (wood waste/sewage sludges and peats) since alkaline conditions are required for sulfate-reducing bacteria.

It may be that *Sphagnum*-dominated wetlands used for the treatment of mine drainage will be most effective at sites where drainage waters have a pH of 4 or less (Wieder et al. 1985), although sulfate reduction processes cannot be expected to contribute significantly to water treatment (Schafer and Associates 1991). In addition, Kleinmann (1985) recommends that *Sphagnum*-dominated wetlands may be most appropriate for relatively small flows (less than 10 gallons per minute) allowing for the large surface area requirement (200 ft² of wetted area per gallon per minute of flow).

Since *Sphagnum* metal ion accumulation generally occurs by cation exchange (Burris et al. 1984, Kleinmann 1985, Watson et al. 1989) and is five times greater in favor of iron over manganese, Unz and Dietz (1986) recommend that a two stage system be employed, where *Sphagnum* spp. is planted in the first stage to reduce iron

content and a marsh-type planting (i.e., *Typha*) be employed to remove manganese in the second phase of treatment. Similarly, Burris et al. (1984) demonstrated that where manganese and iron are both present in the drainage, iron is preferentially removed over manganese; iron can also displace manganese bound to the *Sphagnum* (Fennessy and Mitsch 1989).

Sphagnum recurvum is one of the most tolerant species, growing well in low pH and high sulfate conditions (Wieder et al. 1984). In the field, *Sphagnum recurvum* can tolerate AMD with iron concentrations of less than 150 mg/L for the long term (Kleinmann 1985). Kleinmann and Hedin (1989) found that cadmium concentrations of 11 mg/L retarded the growth of *Sphagnum fimbriatum* but no retardation was evidenced at 1.1 mg/L cadmium. Lead at 207 mg/L strongly retarded the growth of *Sphagnum fimbriatum* (Gignac and Beckett 1986).

Typha and Other Vascular Plants

Almost all wetlands constructed today are planted with *Typha latifolia* (Hedin 1989, Nawrot and Klimstra 1990, Perry and Kleinmann 1991). *Typha* is readily available, transplants well, and has proven to be tolerant to a wide range of water conditions (Fennessy and Mitsch 1989, Samuel et al. 1988, Sencindiver and Bhumbla 1988). *Typha* is much more tolerant of mine drainage than *Sphagnum*, most likely because it does not accumulate metals at high enough concentrations to reach toxicity (Sencindiver and Bhumbla 1988). Kastning-Culp and Lockwood (1993) advise that

Typha is not amenable to use in high elevation wetlands as survival would not be assured above 2400 meters (8000 feet).

The Tennessee Valley Authority (TVA) has generally been successful in the use of oxidative mechanisms within constructed wetlands to at least partially treat AMD. Almost all of the TVA wetlands have been planted with *Typha* and other vascular plants. Only one wetland has successfully met compliance requirements (Table 1) when influent total iron concentrations exceeded 50 mg/L (Brodie 1990). Other wetlands receiving AMD with a total iron concentration of greater than 50 mg/L possess low pH, high acidity, and the resultant inability to remove iron and manganese effectively (Brodie 1990). The successful wetland contained influent alkalinity of approximately 180 mg/L while the alkalinities of other wetlands ranged from 0 to 26 mg/L (Brodie 1990). Similarly, after studying 48 constructed wetlands in Pennsylvania, Hellier (1989) concludes that adjustment of influent alkalinity through the addition of calcium carbonate to not less than 200 mg/L CaCO_3 may improve the effectiveness of wetlands treatment of iron removal. Mine drainage with a net alkalinity ($\text{pH} > 7$) can be treated with a simple aerobic system (Brodie 1990, Perry and Kleinmann 1991, Stark et al. 1990).

Kleinmann and Girts (1986) have indicated that a typical *Typha*-dominated wetland reduces manganese by 20 to 50%. Taylor and Crowder (1983a) have found that *Typha latifolia* can tolerate copper concentrations up to 50 mg/L and nickel concentrations greater than 150 mg/L. *T. latifolia* shoots grown in a solution of 100 mg/L copper (as an EDTA complex) accumulated $127 \pm 28 \mu\text{g/g}$ copper in their

leaves and $2,364 \pm 209 \mu\text{g/g}$ copper in their roots, but displayed reduced leaf growth when the copper concentrations in the leaves reached $80 \mu\text{g/g}$ (Taylor and Crowder 1983b, Taylor and Crowder 1984). Taylor and Crowder (1983b) have reported *Typha* growing in marsh water where sediment contained $3,738 \mu\text{g/g}$ copper and $9,372 \mu\text{g/g}$ nickel. *Typha* has been successfully transplanted by Kalin and Van Everdingen (1987) into water with a pH range of 2.9 to 5.7, a mean zinc concentration of 221 mg/L and a mean copper concentration of 21 mg/L.

Typha-dominated wetlands appear more successful than *Sphagnum* wetlands in treating acid mine drainage but this may be due to difficulty in establishing the latter (Girts and Kleinmann 1986, Kleinmann and Girts 1987). In contrast, survival of cattails has been evaluated as high in mine water with a pH of 3 or above with little replanting necessary (Kleinmann and Girts 1986). Another factor affecting the greater success of *Typha* wetlands in the treatment of AMD may be the presence of reducing conditions which enhance sulfate reduction processes. Due to saturated soil conditions and the presence of decomposable organic matter, the biochemical oxygen demand of most cattail wetlands far exceeds the rate of oxygen diffusion into the sediments; without oxygen the sediments become highly reduced due to microbial respiration (Sencindiver and Bhumbla 1988). Microbial sulfate reduction is sensitive to the presence of oxygen; their efficiencies are directly related to lower redox conditions (Wildeman et al. 1990). Cattail-dominated wetlands also generally have the advantage of possessing a higher pH than *Sphagnum* wetlands, which is a required condition for sulfate reduction to occur.

Typha latifolia may provide higher radial oxygen loss (i.e., loss of oxygen to substrate) than other common species, thereby enhancing substrate oxidizing conditions to bind insoluble forms of metal precipitates in the substrate (Brodie 1990, Michaud and Richardson 1989). While cattails were found to remove less than 1% of the total iron in mine drainage entering a wetland, Sencindiver and Bhumbra (1988) have indicated that cattails may lower the redox potential of wetlands enough to precipitate iron and manganese. Under reducing conditions, Fe^{2+} is thought to be oxidized to the less soluble Fe^{3+} form by oxidative agents released from the *Typha* roots, thereby creating a coating of an insoluble Fe^{3+} compound on the root surface (Taylor et al. 1984). These iron oxide plaques are a survival mechanism for cattails growing in a contaminated environment, since the coatings allow no further uptake of iron (Taylor and Crowder 1984, Taylor et al. 1984).

Researchers working at the Big Five Tunnel site in Idaho Springs, Colorado have achieved successful AMD treatment with a constructed wetland planted with *Typha latifolia*, *Carex aquatilis*, *Carex utriculata*, and *Juncus arcticus* (Wildeman et al. 1993a). The cattails were found to be by far the most successful of the transplanted species, accounting for approximately 75 to 80% of the total plant biomass.

Combination Vegetation--*Sphagnum* and *Typha*

Experiments by Girts et al. (1987) showed that combined vegetation (*Typha* and *Sphagnum*) systems exhibited the largest average metal removal, followed by *Typha*

and *Sphagnum*-dominated systems. However, when turnover time is considered, *Typha* systems are associated with the greatest change per unit contact time, with mixed systems intermediate in effectiveness (Girts et al. 1987). Brooks (1984) has shown that tolerances of *Sphagnum* and *Typha* overlap in the 4.5 to 5.5 pH range. Pesavento (1984) has records of removal of 80 or more mg/L iron from solution within a *Typha*-bryophyte combination marsh. It is recommended that other species including *Typha*, sedges, or rushes be planted with *Sphagnum* to provide a substrate for growth of the moss (Brooks 1984).

Equisetum

When acid mine drainage (median pH values as low as 3.8, iron concentrations of over 55 mg/L, and manganese concentrations of almost 60 mg/L) was applied to a constructed wetland planted with *Equisetum arvense* (field horsetail) and other vegetation, effluent water quality improved (Kepler 1986). Since the more iron concentrated in *E. arvense*, the less manganese that is concentrated, the *Equisetum* was planted downgradient from the cattail, rush, and sedge portion of the wetland; the system was then able to remove the majority of the iron from the drainage before the flow reached the horsetail section of the wetland (Kepler 1986). Preliminary results show that when wetlands are designed to remove iron from the AMD before attempting to remove the manganese, *Equisetum arvense*, and possibly other wetland vegetation, has at least some capacity to remove manganese from AMD (Kepler 1986).

Plant Tolerances to Metals

In an experiment testing tolerance of three wetland plant species (*Typha latifolia*, *Sphagnum recurvum*, and *Pohlia nutans*) to AMD, Wenerick et al. (1989) found *T. latifolia* to be the most generally tolerant of acid (pH of 3.40-3.50) mine water heavily contaminated with iron (365-496 mg/L total Fe), manganese (67.0-92.1 mg/L total Mn), and sulfate (3830-4330 mg/L SO₄). When exposed to AMD water diluted 2:1, *T. latifolia* leaves exhibited 108% growth, about the same as the control plants (Wenerick et al. 1989). *Typha latifolia* has also been reported to be tolerant of high levels of zinc (McNaughton et al. 1974). Brodie (1991) reported the most tolerant wetland species to acid mine drainage were cattail (*Typha* spp.), rush (*Eleocharis* spp.), water sedge (*Carex* spp.), threesquare (*Scirpus americanus*), and wool grass (*Scirpus cyperinus*).

Equisetum arvense (field horsetail) was found by Kepler (1986) to be present in wetlands receiving and effectively treating AMD; *E. arvense* was noted to be absent from most of those wetlands associated with acid mine drainage but ineffective in treating the wastewater. When Kepler (1986) constructed a wetland planted with *E. arvense* and added AMD with median pH values as low as 3.8, iron concentrations of over 55 mg/L, and manganese concentrations of almost 60 mg/L, water quality improved. Further, preliminary ashing results revealed that over 4000 mg/kg of manganese (dry weight) was found in *E. arvense* in wetlands treating acid mine

drainage (Kepler 1986).

In a study of plant tolerances to heavy metals, Kastning-Culp and Lockwood (1993) found *Carex aquatilis* and *Pohlia nutans* to be tolerant of sites with high zinc and copper concentrations. *Carex microptera* was commonly found on sites high in zinc, and *Carex rostrata* was present on sites with high concentrations of zinc, lead, and arsenic. Webster (1985) found *Pohlia nutans*, a subaquatic, turf-forming moss, growing on exposed coal seams in water with low pH (pH 2.65) and high metal concentrations. While the best overall accumulator was found to be *Pohlia wahlenbergii*, all of the plants tested (*Deschampsia cespitosa*, *Pohlia wahlenbergii*, *P. nutans*, *P. annotina*, *Carex microptera*, *C. aquatilis*, *C. rostrata*, *C. scopulorum*, *Epilobium glaberrimum*, *Bryum lisae*, *Senecio fremontii*, *Equisetum arvense*, *Poa interior*, *Agrostis exarata*) were capable of accumulating multiple metals with uptake of arsenic being the most effective, followed by lead, copper, zinc, and then cadmium (Kastning-Culp and Lockwood 1993).

As mentioned earlier, other plants capable of bioaccumulating metals include *Salicornia pacifica* (pickleweed), *Phalaris arundinacea* (reed canary grass), and *Carex stricta* (Chan et al. 1982). Martin (1976) suggests that *Eichhornia crassipes* (water hyacinths) are very effective in removing metals such as lead and mercury from contaminated waters. Jamil et al. (1987) also reported that water hyacinths growing in either acid or alkaline environments had a tendency to change the pH towards neutrality. Iron solutions (Fe^{3+}) were not tolerated by water hyacinths and exposure

resulted in death of the plants; neutralization capacity was found to be more for Zn^{2+} , followed by Cd^{2+} and Cu^{2+} when the plant was exposed to mixtures of heavy metal solutions (Jamil et al. 1987).

Dollhopf et al. (1989) strongly suggest that *Carex rostrata*, *Betula glandulosa*, and *Salix boothii* be transplanted to constructed wetlands since they absorbed substantial quantities of aluminum, cadmium, copper, iron, manganese, nickel, and zinc without ill effect and are readily available in Montana and relatively easy to transplant.

One of the problems associated with treating AMD via plant uptake of metals is that the relatively brief turnover time of the plants results in short-term storage, particularly with floating aquatic macrophytes (Reddy and DeBusk 1987). Thus, if metal uptake by wetland vegetation is significant, it may be necessary to harvest plants in order to ultimately remove metals from the system (Blake et al. 1987, Dierberg et al. 1987, Reddy and DeBusk 1987).

Choosing Wetland Vegetation for Planting

Some criteria to consider when choosing vascular plants for planting or seeding in constructed wetlands include: whether or not the species are common in wetlands with high concentrations of metal such as iron, manganese, and other metals; availability; transportability; how hardy the species is; and whether or not the species is capable of reproducing by rhizomes or seeds (Pesavento 1984).

Since acid mine drainage from each source varies, it is recommended that laboratory testing be performed to determine the specific contaminants present in the AMD. Research must then be completed to determine which wetland species tolerate or absorb these contaminants. Of the tolerant species, those which tend to be present in local wetlands should be given priority. In addition, where possible, small scale experiments may be conducted to evaluate which plant species are tolerant to the specific conditions (i.e., pH, iron, manganese, sulfate) of the AMD. These are important steps in increasing the survivability of the vegetation planted in the constructed wetland.

Wetland plants can be purchased from nurseries, collected in the wild, or grown for a specific project. Only a few commercial nurseries, however, specialize in plants for wetland construction. Most of the nurseries offering wetland plants are located in Florida, Maryland, New York, and Wisconsin with very few in the central and western United States (Allen et al. 1989). Wetland plants collected in the wild are more closely adapted to local environmental conditions than those plants acquired from a nursery (Allen et al. 1989, Cairns and Pratt 1985, Fennessy and Mitsch 1989). Warburton et al. (1985) found that survival of locally collected *Scirpus* spp. was 95% higher than survival of a commercial stock. Some of the disadvantages of collecting plants in the wild include: (1) undesirable weedy species may be inadvertently collected; (2) rare or endangered species may be damaged or decimated during collection; (3) cost may be prohibitive due to logistics or time expenditures; (4) plants may not be available due to limited supplies, local regulations, or difficult access to private land; and (5) plants

may be unavailable early or late in the season (Allen et al. 1989).

Most emergent wetland species transport well, while floating emergents are less transportable. Transporting emergents requires care so that plants are not subjected to extreme temperatures, drying or wind (Brodie 1990). Establishment of live plants will immediately prevent soil erosion (Kastning-Culp and Lockwood 1993). For some species, planting rhizomes instead of whole plants is most successful. For example, when *Equisetum arvense* was planted in a constructed wetland as mature plants, survival was poor; however, when planted as rhizomes, they established themselves as above water plants in less than one month (Kepler 1986).

Planting Methods

Planting of whole plants or dormant rhizomes or tubers has offered the most successful establishment of wetland vegetation (Allen et al. 1989, Brodie 1990, Kepler 1986). Allen et al. (1989) have found seeding to be unsuccessful due to stratification requirements of wetland seed and loss of seed by water action. However, Kastning-Culp and Lockwood (1993) recommend that both live plants and seeds be used to ensure successful establishment of plants. Establishing live plants will immediately prevent soil erosion (Kastning-Culp and Lockwood 1993). Tubers can be planted by placing them deep enough in the substrate to prevent them from floating out of the medium (Allen et al. 1989). Allen et al. (1989) also suggest that cores of wetland soil (eight to ten cm diameter) and associated propagules can be collected from existing

wetlands and transplanted to constructed wetlands. Brodie (1990) and Brodie et al. (1986) suggest that *Typha* be set into the substrate and stems broken over at the water level in order to stimulate new growth from the rhizomes and to prevent windfall. Allen et al. (1989) recommend that emergent species be planted in a wet but not flooded substrate and permitted to grow enough to generate a stem with leaves that protrude above the initial flooding height. The substrate for the young stalks should only be saturated, not flooded, and as the plants grow the water level may be raised proportionally. For species which grow in the shallow-to-transitional zone (wet meadow species), watering during the first year should be limited to shallow sheetflow with intermittent dry periods depending on the species. Spikerushes (*Eleocharis* spp.), fescues (*Festuca* spp.), bluegrasses (*Poa* spp.), bentgrasses (*Agrostis* spp.), perennial foxtails (*Alopecurus* spp.), and other wet meadow species will tolerate reasonably long dry periods, while manna grasses (*Glyceria* spp.) and other species should not become completely dry during the first growing season (Allen et al. 1989).

When *Equisetum arvense* was planted in a constructed wetland as mature plants, survival was poor; however, when planted as rhizomes, they established themselves as above water plants in less than one month (Kepler 1986). Kepler (1986) found the more iron concentrated in the *Equisetum*, the less manganese that is concentrated. Similarly, the less iron available to *E. arvense* in the water flow, the lower the manganese effluent values at the *Equisetum* planting; thus, it appears critical that iron be removed from the acid mine drainage before the flow passes through the *Equisetum*

planting (Kepler 1986).

Generally, *Typha* and other emergent species are planted in 15 to 45 cm of organic substrate such as topsoil, rotten animal manure, spoiled hay, compost, and/or spent mushroom compost (Hedin 1989). At acidic sites, 8 to 16 cm of crushed limestone is commonly spread beneath the substrate to provide some neutralization (Hedin 1989). Shallower basins may have volunteer *Typha* along the shoreline which probably contributes to erosion control and settling of sediment. Cattails also volunteer in linear or sinuous wetlands found along the toe of long slopes (Brooks 1984).

Brodie (1990) successfully established wetland vegetation when plants were hand-dug to obtain complete root balls and rhizomes, and planting was completed on the same day. Careful transport is important so that plants are not subjected to extreme temperatures, drying or wind (Brodie 1990). Successful transplanting of *Scirpus*, *Carex*, and *Juncus* was achieved by simply placing clumps in the desired transplant location (Brodie 1990). *Eleocharis* and *Equisetum* were carefully set into the substrate (Brodie 1990). Brodie et al. (1986) successfully transplanted wetland vegetation where larger plants (*Typha*) and clump-forming species (*Scirpus*) were spaced 1/m² and those with smaller habits (*Eleocharis* and *Juncus*) were planted 5 to 10/m².

The planting period generally begins in temperate climates after dormancy has begun in the fall and ends once the first third of the summer growing season ends (Allen et al. 1989). Allen et al. (1989) recommend that tubers and rootstock be planted during fall dormancy. Successful plantings in the fall include the following species from the genera of burreed (*Sparganium*), dock (*Rumex*), bulrush (*Scirpus*), rush

(*Juncus*), and arrowhead (*Sagittaria*). In contrast, some species of manna grasses (*Glyceria* spp.), sedges (*Carex* spp.), and cattail (*Typha* spp.) have been more successfully transplanted after dormancy has been broken in the spring (Allen et al. 1989). Kastning-Culp and Lockwood (1993) recommend that *Epilobium* be planted in the spring, mosses in midsummer, and sedges and grasses in the fall; *Epilobium* will act as a nurse crop as it germinates in three to six days. Sedges and grasses will receive stratification which will increase germination success with fall planting (Kastning-Culp and Lockwood 1993). Order of planting is suggested as follows: *Epilobium anagallidifolium* should be planted first, next *Pohlia nutans* and *P. wahlenbergii*, *Carex aquatilis*, *C. rostrata*, and *Deschampsia cespitosa* next.

Allen et al. (1989) recommend that plantings be permitted to become well established before introducing wastewater into the wetland since plants require time to overcome stress associated with planting before being introduced to additional stresses. Successful establishment may require one or two full growing seasons.

Brooks (1984) recommends that the optimum water depth for wetlands constructed to treat mine drainage is 0.5 m (18 inches) or less, with some deep holes and shallow areas to enhance plant survivorship. The average water depth in wetlands successfully constructed by the Tennessee Valley Authority (TVA) to treat mine drainage ranged from 15-30 cm with some deeper and shallower areas to provide for species diversification (Brodie 1990, 1991). Isolated deep pockets of up to 2 m were included to provide refuge in drought events (Brodie 1990, 1991). In most constructed wetland systems, water depth is 5 to 15 cm above the substrate and flow is

quite slow (Hedin 1989, Perry and Kleinmann 1991). Girts and Kleinmann (1986) suggest that *Sphagnum* moss should be planted in constructed wetlands which are shallow and handle low flow seepage, while wetlands constructed to treat higher flow should be planted with *Typha*, *Equisetum*, and sedges. Transplanting success has been achieved when *Typha* was transplanted in deep (<0.5 m) water and *Eleocharis*, *Carex*, and *Scirpus* were planted in shallow water (<0.2 m) (Brodie 1990).

SUBSTRATES

A critical component of wetland systems is the organic substrate (Kastning-Culp and Lockwood 1993, Spotts et al. 1992). It has been recommended that composted plant materials be used since the higher the percent of compost added, the higher the effective rate of mitigation (Kastning-Culp and Lockwood 1993). In most constructed wetlands, the major flow path of water is generally over the substrate (Dvorak et al. 1991, Hedin et al. 1989, McIntire et al. 1990); metals are removed primarily by oxidation and the formation of metal oxyhydroxides (Hedin 1989, Spotts et al. 1992, Stone 1984). Commonly there is no increase in pH or an actual decrease in pH (Brodie et al. 1988, Kepler 1990), particularly in *Sphagnum*-dominated systems (Fennessy and Mitsch 1989). By maximizing contact of the AMD with the substrate to optimize microbial sulfate reduction, however, notable increases in pH and metal removal have been observed (Kleinmann and Hedin 1989, Perry and Kleinmann 1991). It should be noted, however, that the Tennessee Valley Authority has achieved successful treatment of AMD using wetlands constructed with no organic substrate (Brodie 1990, Brodie et al. 1989c).

Most constructed wetlands include 15 to 45 cm of a composted organic substrate in which emergent plants can root (Perry and Kleinmann 1991). A 20-centimeter deep substrate is adequate to achieve reducing conditions and for growth of *Typha* (Schafer and Associates 1991). Howard et al. (1989c) suggest that in harsh climates, depth of substrate is critical if the system is to operate year-round; at least

one meter of substrate is needed to maintain an unfrozen anaerobic zone. Kleinmann and Hedin (1989) recommend that organic material be composted before being placed in constructed wetlands. For wetlands designed for subsurface flow, the substrate should be tested to ensure that it will conduct water sufficiently (Howard et al. 1989c). Incorporation of gravel beneath the organic substrate may enhance permeability and improve the contact area of the substrate (Howard et al. 1989c). Wildeman et al. (1993a) further suggest that constructed wetlands planted with vegetation may provide increased permeability of the substrate which in turn enhances biological treatment of AMD. At sites with very acidic drainage, 8 to 16 cm of crushed limestone is often placed beneath or incorporated within the organic substrate to provide some neutralization (Perry and Kleinmann 1991).

In contrast, an experiment by Hedin (1989) indicated that with the exception of iron, the type of organic substrate (spent mushroom compost, composted animal manure, hay/straw, *Sphagnum* peat, a mixture of composted animal manure with hay/straw, a mixture of composted animal manure and spent mushroom compost, a mixture of *Sphagnum* peat and hay/straw, and no organic material) used has no effect on treatment efficiency for the chemical components of AMD including acidity, hydrogen ions, aluminum, manganese, and sulfate. Similarly, Hedin (1989) reported, based on the same experiment, the type of chemical amendment had no effect on treatment efficiency for any of the acid mine drainage chemical constituents; chemical amendments tested included lime/limestone, fertilizer, both, and neither. Stillings et al. (1988) has shown that once the microbial-substrate-vegetation system becomes

established, the type of substrate employed is relatively unimportant. However, this may not be the case for treatment of metal mine drainage in harsh environments (Howard et al. 1989c).

In a column experiment testing the effectiveness of various substrates (wood waste/sludge, mushroom composts, and peat) at treating acid mine drainage, Schafer and Associates (1991) found only spent mushroom composts were effective in treating metals (except manganese) for the long term (6 months) and maintaining circumneutral pH values; removal was greater than 99% of influent aluminum, iron, and zinc. Alkalinity was generated mostly from the inherent neutralization potential (NP) of the substrate and to a lesser degree sulfate reduction. At first, all substrates removed metals and produced neutral effluent pH values due to cation exchange capacity (CEC) and inherent NP. As CEC sites were filled and neutralization potential was exhausted, metal levels rose and pH declined. The metals which were retained in the substrates were predominantly sorbed and solid phases including oxyhydroxides and carbonates, with only a minor percentage of metals present as sulfides, indicating removal of metals by sulfate reducing bacteria was minor. All substrates tested were net manganese producers (Schafer and Associates 1991). It should be noted, however, that this experiment was conducted solely through the use of columns; no constructed wetlands were tested.

Spotts et al. (1992) tested the performance of numerous substrates (composted wood waste/sewage sludge, peat, straw, cattle manure, sugar beet waste, and dried poultry waste) in terms of capability to improve water quality of mine drainage via

column experiments. Results indicated that only the substrate mix containing 15% (by volume) dried poultry waste significantly reduced metal levels and maintained a neutral pH for the duration of the study (6 months). This performance can be attributed in part to the high NP of the dried poultry waste (Spotts et al. 1992). Researchers concluded that the use of sulfate-reducing wetlands alone may not be sufficient to treat moderate to severe cases of AMD (Spotts et al. 1992).

Since relatively alkaline conditions are required for sulfate-reducing bacteria (Rice and Rabolini 1990, Schafer and Associates 1991, Silver 1989), it is recommended that alkalinity be introduced into the system prior to the entry of acid mine drainage in order to provide an environment where sulfate reduction can occur at an optimum rate given adequate carbon, nitrogen, and other nutrients are available (Schafer and Associates 1991).

Mushroom Composts

Spent mushroom compost consists of a composted mixture of manure, straw, corn cobs, (McIntire et al. 1990) hay, and wood chips conditioned with gypsum and limestone and is used for cultivating mushrooms (Dvorak et al. 1991). Currently, the most successful and inexpensive substrate is spent mushroom compost, which is capable of supporting relatively high rates of sulfate reduction (McIntire and Edenborn 1990); it also contains limestone, which contributes additional alkalinity (Perry and Kleinmann 1991).

Both mushroom composts tested by Schafer and Associates (1991) were found to exhibit significantly higher neutralization potentials (NPs) and higher resultant equilibrium pH values than other substrates. Hammack and Hedin (1989) have established that the NP of fresh mushroom compost is approximately 3.5% CaCO_3 equivalents.

Of three organic substrates (peat, wood waste/sewage sludge, and mushroom compost) tested for their ability to improve mine waste water quality, only mushroom composts were effective in treating metals (except manganese) and maintaining circumneutral pH values from the effluent chemistry results (Schafer and Associates 1991). Mushroom composts removed greater than 99% of influent aluminum, iron, and zinc.

Similarly, in the Big Five Tunnel constructed wetland, after about 2 years of operation, the mushroom compost showed a much better capability to remove metals and raise the pH than the substrates (one cell contained peat, aged steer manure, and decomposed wood shavings and sawdust; and the other cell contained the same components plus limestone rock) in the other wetland cells at similar flow rates; the most abundant heavy metals in the mine drainage were iron, manganese, zinc, and copper (Howard et al. 1989c, Wildeman and Laudon 1989). Consistent removal of copper, iron, and zinc was demonstrated at the Big Five wetland using fresh mushroom compost with flow rates of less than 1.5 L/min in a system with an area of 18.6 m², corresponding to a loading rate of approximately 100, 1000, and 4000 mg/day/m², for Cu, Fe, and Zn, respectively (Wildeman et al. 1990). Manganese was not effectively

removed and at low flows was added to solution; thus, manganese removal must be accomplished in later stages in the wetland after the pH is raised and the other heavy metals have been removed (Wildeman et al. 1990). The difference in the effectiveness of the substrates to improve water quality has been tentatively attributed to the presence of microbial activity as the mushroom compost contained the highest populations of sulfate reducers in the original soil and after two months of flow (Batal et al. 1989, Wildeman and Laudon 1988).

McIntire and Edenborn (1990) and McIntire et al. (1990) successfully achieved neutralization of approximately 23% of the AMD via sulfate reduction when forcing the mine drainage through mushroom compost substrate underlain by limestone gravel in a constructed wetland (Table 3). The bacteria effectively decreased concentrations of many metals added to their growth media through the process of sulfate reduction; the by-product hydrogen sulfide formed insoluble precipitates with nickel, copper, zinc, cadmium, and lead. Water quality of the mine drainage was improved by increasing contact between AMD and the anaerobic zone of the organic substrate through the use of subsurface infusion pipes, and was enhanced even further via the introduction of calcareous gravel beneath the organic substrate (McIntire and Edenborn 1990).

Table 3. Metal toxicity tests on a mixed culture of sulfate-reducing bacteria (McIntire and Edenborn 1990).

METAL CONCENTRATION (PPM)				Percent Removal
Metal	Toxicity	After Inoculation ¹	After Growth ²	
Cd	104	35	0.1	> 99
Cu	> 14	14	< 0.1	> 99
Mn	> 223	223	60	73
Ni	> 211	211	0.4	> 99
Pb	> 5	5	0.5	89
Zn	310	194	2	99

¹Metal concentration immediately after inoculation

²Metal concentration at the end of the experiment

Work by Taddeo and Wieder (1991) demonstrated that of five substrates tested, mushroom compost had the greatest contribution of iron sulfides to total iron accumulation (38%). It is conceivable that improved rates of sulfate reduction could be achieved by forcing the AMD through the substrate as McIntire and Edenborn (1990) and Wildeman et al. (1993a) recommend.

Wood Waste/Sewage Sludge

In a column study of the effectiveness of various substrates in improving AMD water quality, Schafer and Associates (1991) found that the pH in the wood waste/sewage sludge columns decreased significantly from a range of pH 6-8 to pH 3-4.5. Effluent iron and aluminum concentrations were actually greater than influent values; although initially removal was >99% and >95%, respectively. Removal of aluminum and iron was by adsorption in wood waste/sewage sludge and peat substrates (Schafer and Associates 1991). Manganese concentrations remained high throughout the study. Removal of zinc in the wood waste/sewage sludge substrates was greater than 99% throughout the study. The wood waste/sewage sludge substrates were significantly less effective in treating AMD than were the mushroom composts.

Very limited sulfate reduction occurred in the wood waste/sewage sludge columns due to the acidity of the substrates. Nitrogen may have been the limiting factor for sulfate reduction in wood waste/sewage sludge and peat substrates (Schafer and Associates 1991).

Peat

Peat has been defined as an organic soil which contains greater than 50% organic matter by mass (Tarleton et al. 1984). Wetlands containing peat substrates can

provide a reasonably rapid removal of trace metals from mine drainage for an extended period according to Lapakko and Eger (1988). The four major processes of metal retention in peat include cation exchange, complexation with peat organics, precipitation as oxides, and precipitation as sulfides (Henrot and Wieder 1989). Sulfide accumulation typically contributes just 1 to 4 % of total iron accumulation in *Sphagnum* peat wetlands (Tarleton et al. 1984, Wieder et al. 1985, Wieder and Lang 1986), although at least one researcher has reported significant sulfate reduction in a constructed wetland receiving AMD and containing peat as the substrate (Calabrese et al. 1991).

Iron oxidation and iron binding on peat organics were found by Henrot and Wieder (1989) to be the predominant processes operating in experimental wetland units receiving AMD, making up 62 and 22% of the total iron in peat, respectively, at the end of the experiment. Similarly, Wieder et al. (1985) reported that iron oxides composed 85% of the total iron retained in peat within a wetland constructed to treat seeps from a surface coal mine. The ability of peat to incorporate iron into an organically bound fraction results in part from the large number of negatively charged sites on the peat itself (Tarleton et al. 1984). After saturation of organic sites, precipitation of iron as iron oxides was the primary retention mechanism in an experiment exposing peat to AMD (Henrot and Wieder 1989).

It should be noted that both experiments conducted by Henrot and Wieder (1989) and Wieder et al. (1985) were performed on sediments from aerobic wetlands. Experiments by Calabrese et al. (1991) with a mixed aerobic/anaerobic constructed

wetland show that it is possible to produce anoxic sediments capable of accommodating sulfate reduction processes with peat as the substrate. Sulfate reduction accounted for 28% of the total iron retained by the wetland (Calabrese et al. 1991), a rate comparable to that achieved by McIntire and Edenborn (1990), who, using mushroom compost as the substrate, neutralized 23% of the AMD flowing through their constructed wetland via sulfate reduction (Table 3).

Laboratory studies have shown that peat is capable of reducing trace metal concentrations in mine drainage by more than 90% (Lapakko and Eger 1988) although experimental results are quite variable depending in part upon study duration. Guertin et al. (1985) noted total and dissolved iron concentrations were reduced by an average of 96.5 and 99%, respectively, after passage through a peat bog. On the other hand, Wieder (1991) evaluated changes in water chemistry within three wetlands constructed with *Sphagnum* peat, straw mixed with manure, or mushroom compost substrates to find that of the three substrates, peat was the least effective in improving the quality of the AMD after ten months of exposure. However, use of peat as an organic substrate generally increases the effectiveness of the constructed wetland in the treatment of AMD over systems with no substrate. An investigation of the effectiveness of various materials (peat, wood chips, sterile tailings, and glacial till) for removing metals from AMD over a thirteen month period revealed that peat was the most successful (Lapakko and Eger 1981).

The efficiency of peat to remove iron from AMD decreased rapidly with increasing iron concentration in the mine drainage, mainly because of the negative

effect of iron concentration greater than 100 mg/L on microbial iron oxidation (Henrot and Wieder 1989). Since acidic conditions were also found to reduce the removal efficiency of the peat, metal removal will be enhanced if the pH of acidic drainages is elevated prior to entering the wetland; trace metal removal was reduced as pH decreased from 7.4 to 4.0 (Lapakko and Eger 1988). The optimal temperature for iron retention in peat exposed to AMD was 25°C, although the optimum temperature was not very pronounced (Henrot and Wieder 1989). Lapakko and Eger (1988) found that peat preferentially sequesters copper over nickel, cobalt and zinc. Manganese retention in *Sphagnum* peat was minimal, and manganese was retained almost exclusively as exchangeable Mn^{2+} (Henrot and Wieder 1989). Other researchers (Wieder and Lang 1986, Stillings et al. 1988, Gerber et al. 1985) have also found that peat wetlands were not very effective in manganese removal from AMD or in accumulating Mn in the substrate.

Experiments by Ansted (1982) investigating the capability of acid treated *Sphagnum* peat to remove heavy metal ions (Mn, Co, Ni, Cu, Zn, Cd, and Pb) from aqueous solution resulted in the following general trends: (1) recovery efficiency is lowest in the pH range of 2 to 4; (2) recovery efficiency is lowered with increasing flow rates, indicating the importance of contact time in the exchange process; and (3) recovery efficiency decreased with increasing concentration and ionic strength.

Eger and Lapakko (1988) demonstrated that metal concentrations in peat decreased with depth and that more than 80% of the metal removal occurred in the upper 20 cm. Work by Taddeo and Wieder (1991) confirm that it is the upper 20 cm

of peat where most metal removal takes place. It is likely that iron oxidation in constructed wetlands is restricted to that depth, thus, increasing peat depth beyond 20 cm is probably not as efficient as increasing the wetland surface area (Henrot and Wieder 1989).

WETLAND CONSTRUCTION AT THE CHALLIS, IDAHO SITE

History and Background

A currently defunct tungsten mine, located approximately 52 km (32 miles) southwest of Challis, Idaho, has been the source of acid mine drainage and heavy metals contamination in Thompson Creek, a tributary to the Salmon River. Fisheries in Thompson Creek include bull trout (*Salvelinus confluentus*), cutthroat trout (*Oncorhynchus clarki*), rainbow trout (*Oncorhynchus mykiss*), sculpin (*Cottus* spp.), and mountain whitefish (*Prosopium williamsoni*). The general topography of this area consists of rugged mountains with steeply eroded ridges and fans. The elevation of the site is approximately 1700 m (5720 feet) above sea level. The total water yield of Thompson Creek is 1,624 ha-m per year (13,173 acre-feet per year) with flow ranging from approximately 0.03 m³ per min (0.02 cfs) to 14.10 m³ per min (8.3 cfs), based on 1982 data.

A reclamation plan for the site has been partially completed by the USDA Forest Service. Tailings and other contaminated materials (approximately 13,528 m³) have been moved away from the creek to prevent tailings from entering the water during runoff/erosion events. The tailings were then placed in a pile and sloped to a 3:1 angle approximately 23 to 30 m from Thompson Creek. The tailings material has had minus 150 mesh agricultural limestone applied at a rate of 44,800 kg per ha (20 tons per acre). Approximately 25.4 cm or 3,360,000 kg per ha (1500 tons per acre) of

non-pyritic fill material were placed on top of the tailings material to form a cap. Next, 44,800 kg per ha (20 tons per acre) of composted sewage sludge was applied. Approximately 8.89 cm or 896,000 kg per ha (400 tons per acre) of quality topsoil were spread on top of the tailings pile. Fertilizer (24-16-16) was applied at the rate of 336 kg per ha (300 pounds per acre) the first year and 168 kg per ha (150 pounds per acre) the second year. Plant species selected for seeding included the following: orchardgrass (*Dactylis glomerata*), smooth brome (*Bromus inermis*), mountain brome (*Bromus marginatus*), tufted ryegrass (*Lolium* spp.), rough fescue (*Festuca scabrella*), redtop (*Agrostis stolonifera*), bluebunch wheatgrass (*Agropyron spicatum*), crested wheatgrass (*Agropyron cristatum*), slender wheatgrass (*Agropyron trachycaulum*), alfalfa (*Medicago* spp.), and yellow sweetclover (*Melilotus officinalis*). The seeding rate was approximately 78 kg per ha (70 pounds per acre). Straw mulch was then applied with a hydromulcher at a rate of 3360 kg per ha (1.5 tons per acre) in order to stabilize the seeds and to retain moisture. An earthen berm has been constructed between the tailings pile and the creek to contain the constructed wetlands and to protect the capped tailings from high creek flows. Between the berm and the tailings pile a series of seven wetland cells have been constructed (Figure 2). Dimensions of the constructed wetland are displayed in Table 4.

Table 4. Dimensions of the existing wetland cells at the Challis, Idaho reclamation site.

Cell Number	Dimensions (m)*	Area (m ²)*	Avg. Depth (cm)*
1	65.5 X 15.2	1,115	38.1
2	30.5 X 13	395	30.5
3	24.4 X 9.1	223	38.1
4	21.3 X 9.1	195	40.6
5	15.2 X 9.1	139	25.4
6	9.1 X 4.6	37	25.4
7	12.2 X 7.6	74	25.4

*measured July, 1993

Current Status

Chemical analysis of the water in the wetland cells has not been conducted. However, tailings material was analyzed in 1990 and 1991 (data in Appendix), thus, chemistry of the drainage seeping into the wetland cell may be inferred from these data. Potential contaminants include zinc, manganese, iron, copper, boron, and sulfate based on this chemical analysis. Another perceived problem with the AMD is

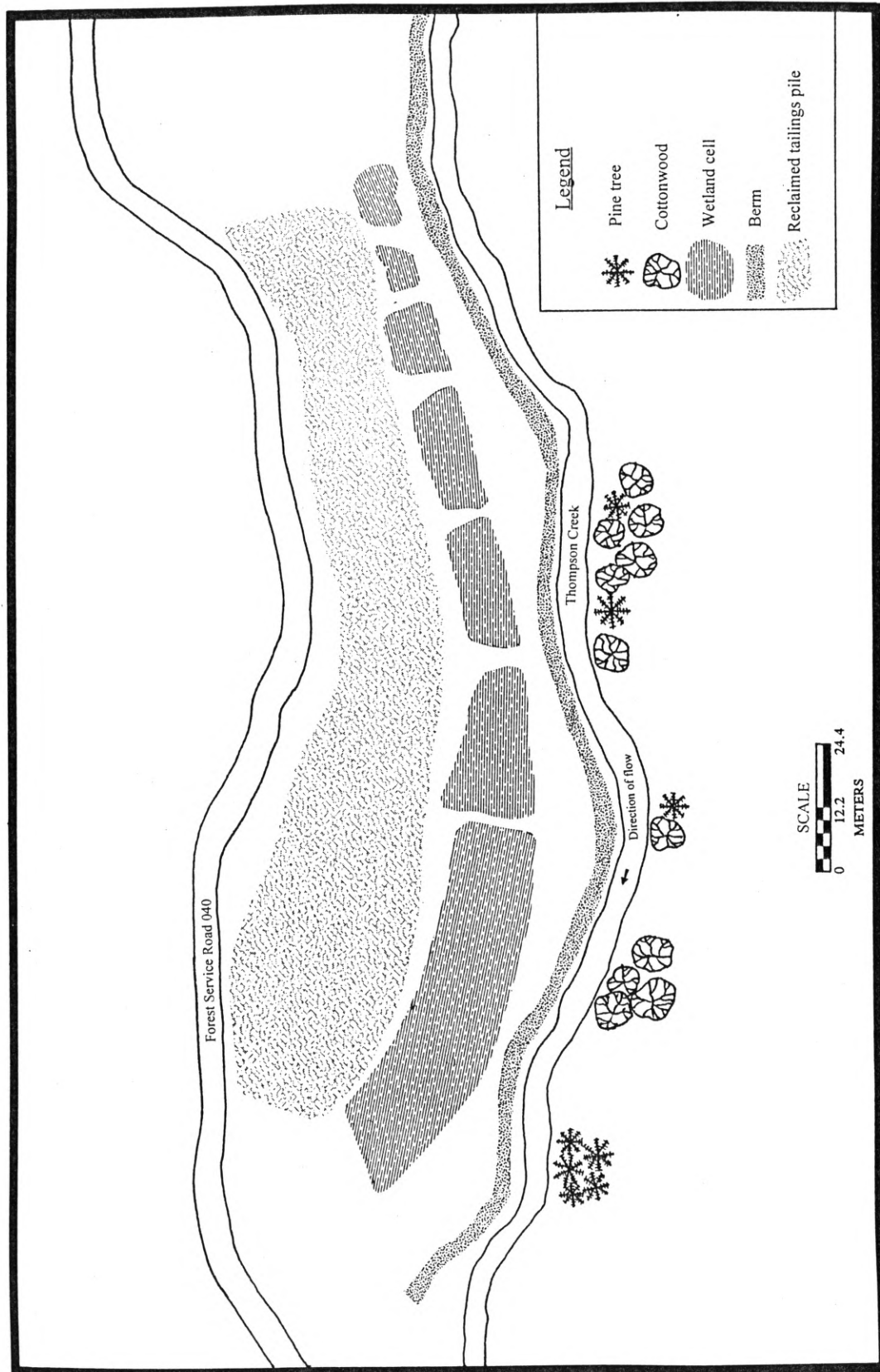


Figure 2. Diagram of wetland cell system located at the Challis, Idaho reclamation site.

excessive acidity; pH values of the water in the wetland cells range from 4.1 to 7.1 (based on data collected by the USDA Forest Service on July, 1993)(Table 5).

Table 5. Values of pH for water in existing wetland cells at the Challis, Idaho reclamation site.

Wetland Cell Number	pH*
1	5.0
2	6.7
3	4.1
4	6.5
5	6.9
6	6.9
7	7.1

*pH values measured July, 1993.

No substrate has been applied to the wetlands with the exception of an unknown quantity and quality of agricultural limestone. Hay-bales have been placed between the wetland cells. Water downstream of hay-bale dams often has a higher pH (Hedin et al. 1988, Stark et al. 1988), attributable in part to sulfate reduction activity in the hay bales themselves (McIntire and Edenborn 1990). Hay bale barriers may also enhance serpentine flow patterns, prevent channelization, and increase the contact of AMD with the substrate and vegetation (Hedin 1989, Kleinmann and Hedin 1989, Perry and Kleinmann 1991).

No attempt has been made to establish vegetation in the wetland cells. The

Yankee Fork Ranger District of the Challis National Forest of the United States Forest Service has requested technical assistance in determining whether or not the application of substrate into the wetland would maximize treatment efforts. Additionally, assistance has been requested to determine which plant species can be planted in the wetland to best treat the AMD.

Technical Approach

The chemistry of the mine drainage and the hydrology of the site are two of the most significant factors that determine wetland construction techniques (Brodie et al. 1988a and b, Brodie 1991). Since AMD from each source varies, it is recommended that laboratory testing be performed to determine which contaminants are present in the AMD and that, where possible, small scale experiments be conducted to evaluate which plant species are tolerant to the conditions (i.e., pH, iron, manganese, sulfate) of the AMD. Preliminary hydrologic evaluations should consist of a complete water budget, including all inflows and outflows of surface water; the precipitation regime; subsurface gains and losses; and losses from evaporation and transpiration (Howard et al. 1989c).

It has been recommended by many sources that an impermeable material be placed beneath constructed wetland cells to prevent seepage and assure water level control (Howard et al. 1989c, Watson et al. 1989). Howard et al. (1989c) recommends sealing with bentonite or placing a plastic liner (e.g. Hypalon[®]) to prevent metals from

reaching the underlying groundwater.

For wetlands designed for subsurface flow, the substrate should be tested to ensure it will conduct water sufficiently; gravel may be incorporated beneath the organic substrate to improve permeability and enhance the contact area of the substrate (Wildeman et al. 1993a). Where wetlands are to be constructed in cold climates, depth of substrate is critical if the system is to operate year-round; at least one meter of substrate is needed to maintain an unfrozen anaerobic zone (Howard et al. 1989c).

Ideal configuration of the wetland system is one which incorporates parallel or series components. A series of wetland cells allows for different removal processes to operate in each cell (Wildeman et al. 1993a). Additionally, a series type system allows for a failure in one cell without destroying the entire project. A mixed aerobic/anaerobic system is highly recommended. One advantage of a mixed system is the production of dense, relatively stable iron sulfides which provide a renewable metal retention mechanism unavailable in simple aerobic systems. Also, sulfate reduction, a removal mechanism associated with anaerobic systems, generates alkalinity.

Because sulfate reduction and metal oxidation are mutually exclusive processes (anaerobic and aerobic, respectively), design of constructed wetlands should be staged. First the AMD is routed through a treatment pond where oxidation is achieved and metal oxides are precipitated out of solution. Next, the AMD is treated anaerobically via sulfate reduction. Sulfate reduction requires the following conditions: drainage must be directed through the substrate; wetland must remain continuously flooded so products of reduction are not oxidized; and a sufficient supply of sulfate (i.e., from

AMD) and a readily metabolized source of organic carbon and nitrogen (i.e., from substrate) must be provided for the sulfate-reducing bacteria. It is conceivable that the failure of most constructed wetlands to exhibit sulfate reduction is due to the lack of at least one of these requirements, and particularly due to the failure to direct flow through the substrate. The addition of an underlying limestone layer is advantageous in that alkalinity is produced both from the reduction of sulfate to sulfides and from the continuous dissolution of calcium carbonates under anoxic sediment conditions. In addition, the presence of limestone can enhance abiotic iron oxidation and hydrolysis by raising the pH to produce a favorable environment for iron precipitation (Hedin 1989). Pretreating AMD before it enters the wetland through the construction of an anoxic limestone drain is one means to raise the pH and enhance oxidation processes. Also, the addition of fertilizer and/or organic matter should be considered in sulfate reducing wetlands as a source of nutrition for sulfate-reducing bacteria.

In determining the size of a wetland necessary for treatment of AMD, the most critical consideration is the loading factor. The Tennessee Valley Authority (TVA) has successfully treated AMD with constructed wetlands by estimating mass loading factors based on pH of the drainage, the amount of Fe, and the amount of Mn. The TVA has established desired discharge requirements of 3 mg/L or less for Fe and 2 mg/L or less for Mn (Wildeman et al. 1993a). The recommended loading factor for Fe is 2 $\text{m}^2/\text{mg}/\text{min}$ if the pH of the drainage is less than 5.5 and 0.75 $\text{m}^2/\text{mg}/\text{min}$ if the pH is greater than 5.5 (Brodie et al. 1988). The loading factor for Mn is 7 $\text{m}^2/\text{mg}/\text{min}$ if the pH is less than 5.5 and 2 $\text{m}^2/\text{mg}/\text{min}$ if the pH is greater than 5.5 (Brodie et al. 1988).

If manganese is present in the AMD, removal should be designed via oxidation. While difficult to oxidize manganese due to the need for optimum pH values of greater than 7.0, it is recommended that the effluent be introduced into an aerobic wetland cell so that manganese will be oxidized to MnO_2 (Klusman and Machemer 1991, Wildeman et al. 1993a). Manganese removal should be staged last, after other metals have been removed.

Researchers disagree on the importance of uptake of contaminants by plants. Some scientists maintain that plant uptake is minimal (Fennessy and Mitsch 1989, Howard et al. 1989, Sencindiver and Bhumbla 1988, Wildeman 1993). Therefore, choice of plant species to be planted in a constructed wetland is not of primary significance as long as the vegetation is able to tolerate the conditions of AMD and local climate. Other researchers claim treatment of AMD with plants can be significant (Blake et al. 1987, Chan et al. 1982, Kastning-Culp and Lockwood 1993, Martin 1976). Nonetheless, the presence of vegetation in constructed wetlands appears to be significant for stabilization and increased permeability of the substrate; continual additions of biomass; reduction of channelization in the surface flow; and, for some species, uptake of metals. In order to minimize channelization, which may decrease contact time between the mine drainage and the substrate and thereby causing a reduction in the efficiency of the system, it is important that careful and evenly spaced placement of vegetation occur.

Choosing local species of vegetation tolerant of the environment and climate is of utmost importance for enhancing survivability. This is particularly important since

removal of iron and manganese has been found to be directly proportional to the density of vegetation in a wetland. Plant-growth tests may be necessary to determine tolerance of vegetation to environmental conditions (i.e., concentrations of metals in the AMD). A secondary factor in selecting vegetation should be capability of the species to absorb contaminants. Emergent species are recommended over floating macrophytes due to better survivability. If the site is below 2400 m (8000 feet) in elevation and the pH of the drainage is greater than 3.0, *Typha* spp. is highly recommended for planting. *Typha* is readily available, transplants well, and has proven to be tolerant to a wide range of water conditions, including high concentrations of iron, manganese, copper, nickel, and zinc. Successful AMD treatment has been accomplished as well with *Eleocharis*, *Scirpus*, and *Carex* spp. For mine drainage which contains a high concentration of manganese, a wetland cell planted with *Equisetum arvense* may provide improved treatment; however, iron should be removed before drainage is permitted to enter the cell planted with *Equisetum*. High tolerance to both zinc and copper has been exhibited by *Carex aquatilis* and *Pohlia nutans*. *Carex microptera* is quite tolerant to high zinc concentrations. Sites with high concentrations of zinc, lead, and arsenic would be amenable to plantings of *Carex rostrata*. Reed canary grass (*Phalaris arundinacea*) has been found to bioaccumulate copper and *Carex stricta* removes nickel. *Betula glandulosa*, *Salix boothii*, and *Carex rostrata* are recommended for transplanting because of their abilities to absorb significant amounts of aluminum, cadmium, copper, iron, manganese, nickel, and zinc.

It should be noted that if significant metal uptake by plants occurs, plant harvest will likely be necessary in order to ultimately remove metals from the system.

Due to difficulties in establishing *Sphagnum* moss species and variable treatment results, it is not recommended that *Sphagnum* be planted in constructed wetlands. Likewise, due to poor success in transplanting *Juncus* spp. by some researchers, avoidance of this species may be advisable.

While the successful Tennessee Valley Authority wetlands are constructed without incorporating organic substrates, most researchers agree that choice of substrate is as important as the choice of vegetation. The most successful treatment of AMD has been achieved using mushroom compost thus this is the substrate of choice. If cost or availability of mushroom compost is prohibitive, a second alternative may be dried poultry waste. Substrates composed largely of carbon with little or no nitrogen should be avoided.

Recommendations for Challis, Idaho Site

Although preliminary construction has already been conducted without the benefit of a full investigation of the hydrology and chemistry at the site, it is still appropriate to perform analyses to determine key characteristics. Water contained in the wetland cells should be analyzed to ascertain the contaminants present. For this paper it shall be presumed that potential contaminants may include zinc, manganese, iron, copper, boron, and sulfate, as well as excessive acidity based on analysis of

tailings material (Appendix).

One of the problems associated with the current design of the Challis, Idaho reclamation site is that there is neither an actual influent nor effluent point (Figure 2). The existing design serves as a drainage trap rather than a staged cell system. In other words, there is no means of directing drainage into the wetland system, through each of the cells, or out of the system as effluent. It is suggested that excavation be conducted to improve the design.

Since the concentration of potential contaminants present in the AMD at the Challis, Idaho site remains unknown, two treatment plans will be presented (Figure 3). The first method of treatment is recommended if the AMD is found to contain low concentrations of metals and the pH is relatively low. Alternatively, treatment two is recommended if the AMD is already oxidized, the pH is relatively high, and high concentrations of metals such as ferric iron, aluminum, and sulfate are present.

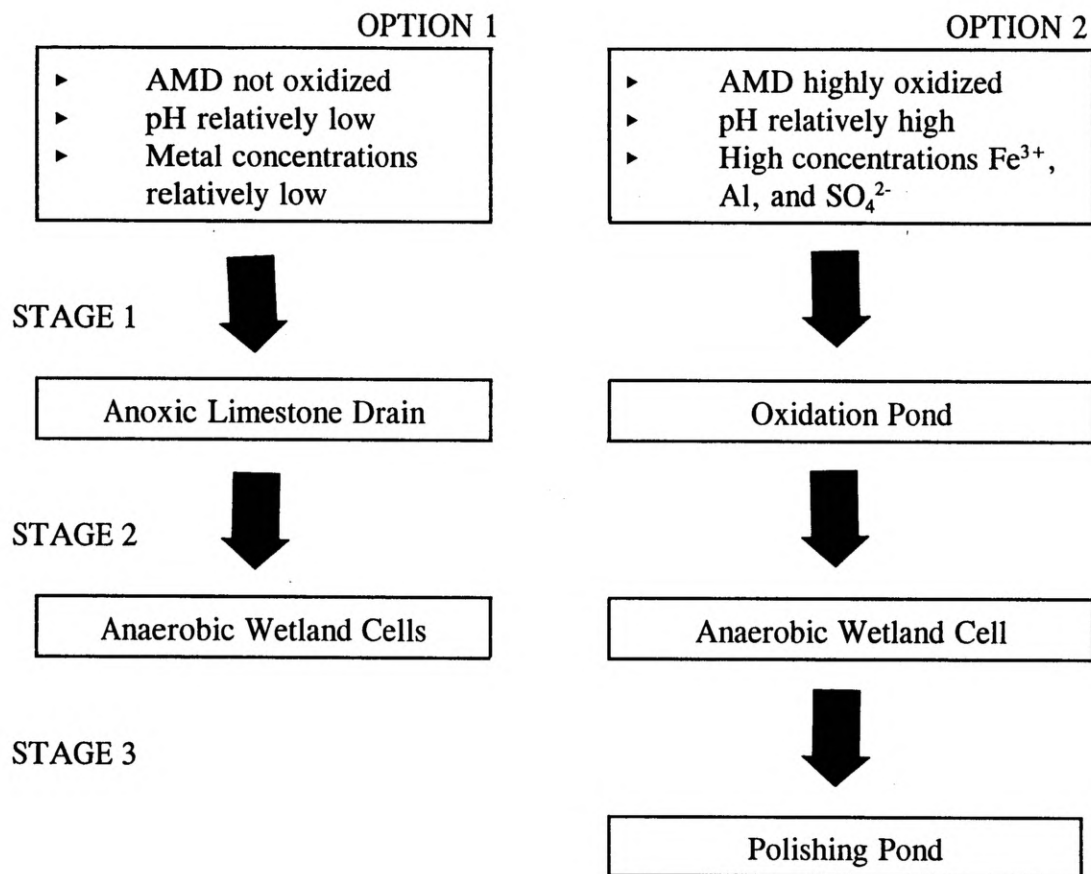


Figure 3. Flow chart illustrating the recommended treatment plan for AMD at the Challis, Idaho site as a function of water quality conditions of the AMD.

AMD Treatment Option 1

This treatment plan would involve a staged system using several wetland cells, providing a more efficient system by minimizing chemical and hydraulic overloading of all cells (Brodie 1990)(Figure 4). Treatment would entail combining primary treatment

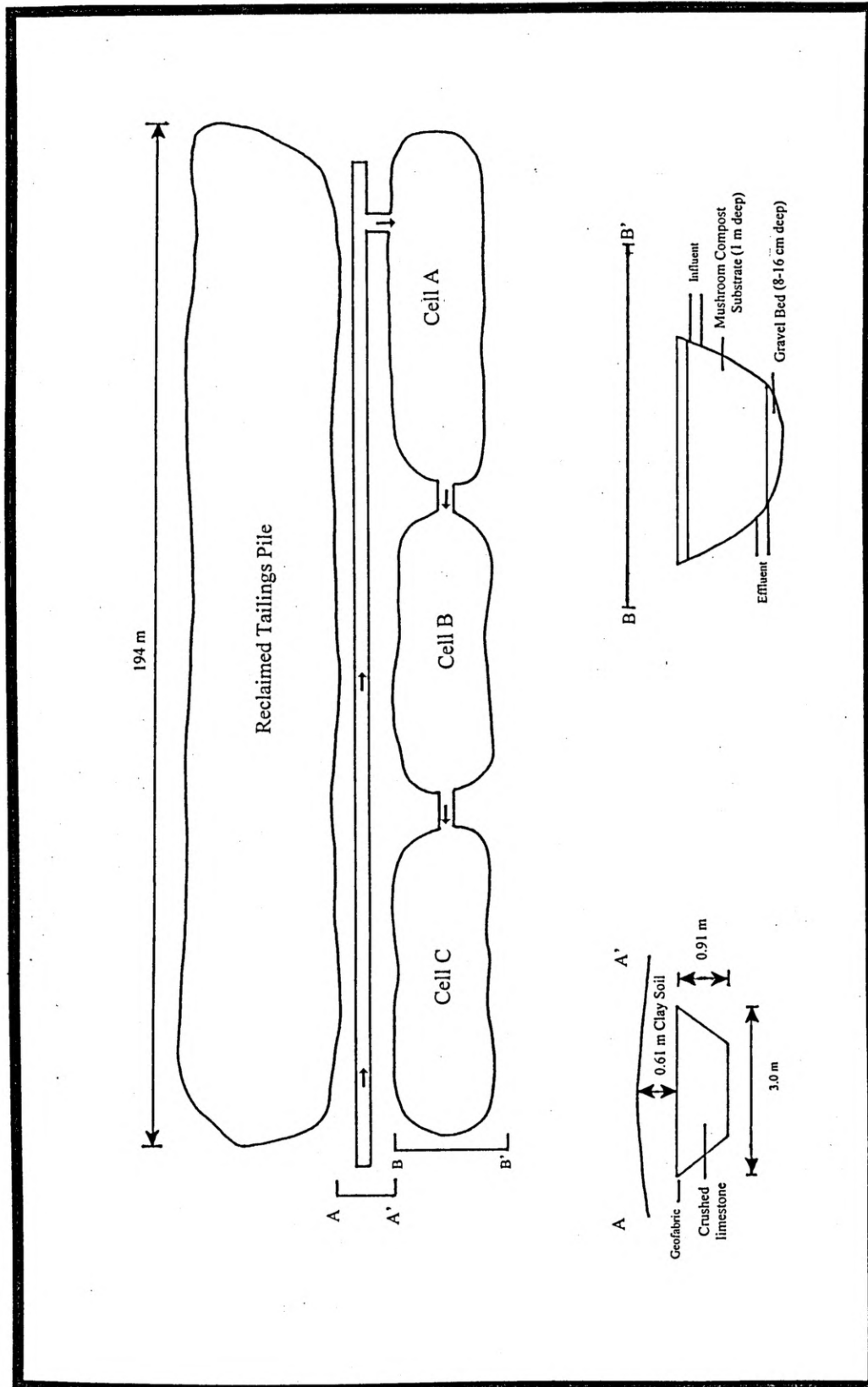


Figure 4. Schematic diagram of treatment method option 1: ALD with three anaerobic wetland cells.

which would raise the pH by running the drainage through an anoxic limestone drain (ALD) with secondary treatment where the drainage enters a wetland with high organic matter content and plants tolerant to the metals present in the AMD. The anoxic limestone drain would be installed at the toe of the tailings pile. The ALD would consist of an open, unlined trench or excavation backfilled with .61 to 1.52 m (2 to 5 feet) of gravel-sized (1.9 to 3.81 cm diameter crushed rock), crushed, high-calcium CaCO_3 (i.e., $\geq 90\%$ available CaCO_3) limestone and covered with plastic (Geofabric) and approximately .61 m (2 feet) of soil to prevent oxygen infiltration (Brodie et al. 1991). If clay soil is available, it should be placed over the plastic as a cap to minimize oxidation of the drain.

If the pH of the drainage could be raised above 6.0 after passing through the ALD, accelerated rates of abiotic iron oxidation and hydrolysis may occur (Hedin 1989). Kastning-Culp and Lockwood (1993) found that by running effluent water over a limestone bed, which is approximately equivalent to an ALD, the solubility of copper was lowered, thus more copper was removed from solution than by wetlands alone.

Next, the drainage would be diverted through a series of three anaerobic cells where treatment would occur via sulfate reduction. From Figure 4 it can be observed that the influent to Cells B and C flows in at the top of the substrate layer, while the effluent flows out through the bottom of the substrate layer. This diversion of flow helps to maximize contact of the drainage with the substrate by collecting water flow from the top surface and redistributing it to the bottom of the substrate.

It is important to note, however, that treatment option 1 is not recommended if

AMD is highly oxidized, pH is relatively high, and/or concentrations of metals such as ferric iron, aluminum, and sulfate are relatively high since under such conditions the ALD is likely to become clogged (Schafer and Associates 1994). Brodie et al. (1991) suggest that installation of an ALD is not recommended if alkalinity exceeds 80 mg/L, total iron concentration is greater than 20 mg/L, dissolved oxygen is greater than 2 mg/L and pH is greater than 6.0.

AMD Treatment Option 2

An alternative to installing an ALD is the construction of a french drain at the toe of the reclaimed tailings pile to divert the drainage into the wetland complex. In this treatment option, drainage would be diverted through a deep, aerobic oxidation pond where precipitation of iron and other metals would occur (Figure 5). No substrate would be required in this wetland cell which would act simply as a reservoir. Without substrate, precipitates may be easily removed as the cell fills in. The cell should be capable of holding four to seven days flow of AMD. Prior to excavation, AMD flow monitoring must be conducted to determine wetland design specifications, and specifically size requirements necessary for 4 to 7 days of storage.

Anaerobic treatment via sulfate reduction would be achieved in a subsequent cell provided certain conditions were met, including the following: (1) drainage must flow through the substrate; (2) the cell must be continuously flooded so products of reduction

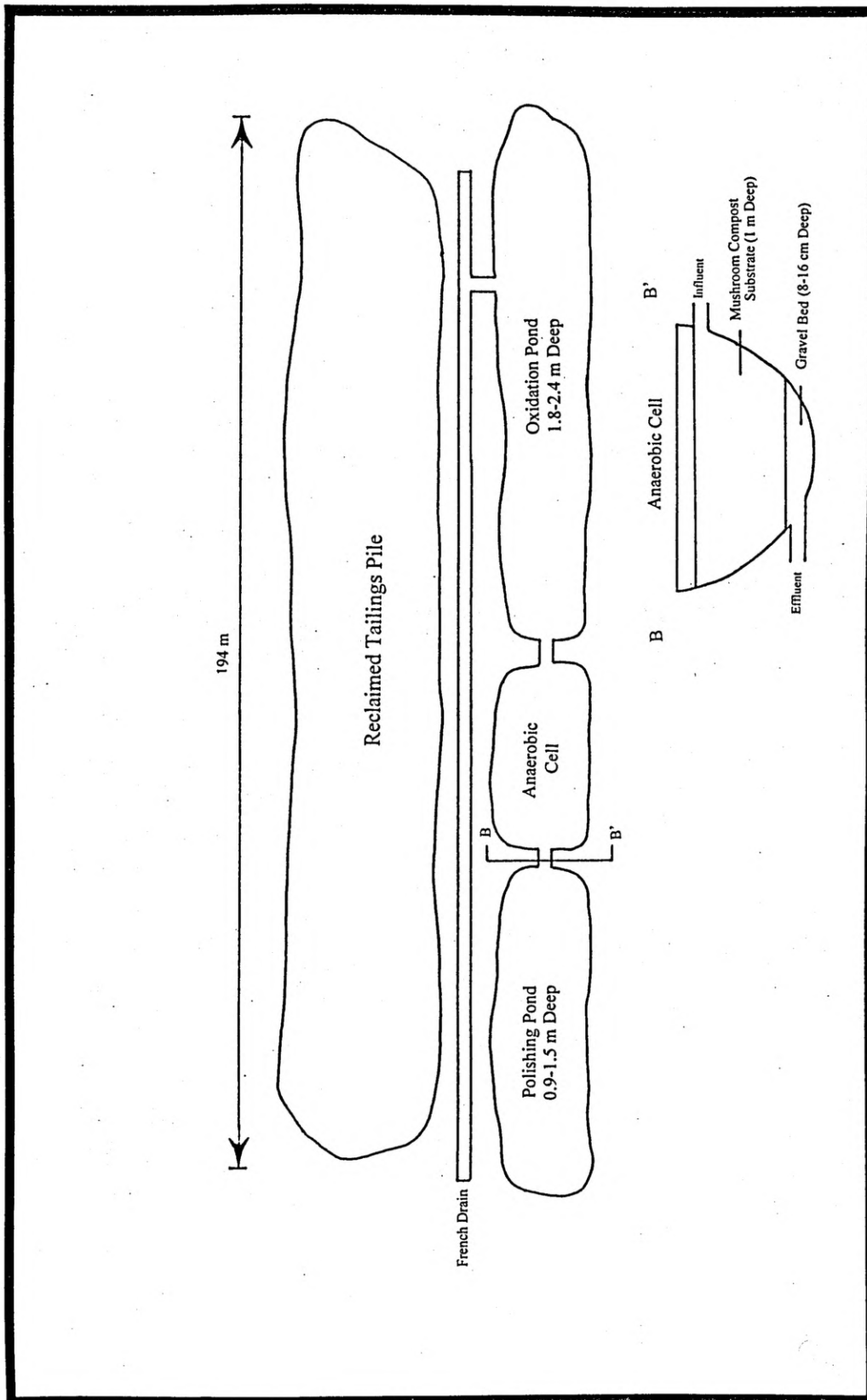


Figure 5. Schematic diagram of treatment method option 2.

are not oxidized; and (3) a sufficient quantity of organic substrate must be present. By incorporating a layer of gravel beneath the substrate layer, it may be possible to fulfill the first requirement of sulfate reduction by increasing permeability and enhancing the contact area of the substrate (Howard et al. 1989c). Recommended average water depth in the cell should be approximately 15 to 30 cm with some deeper and more shallow areas to maximize species diversity (Brodie 1990). Isolated deep pockets of up to 2 m may provide aquatic organisms refuge during drought (Brodie 1990). In addition to placing an organic substrate in the wetland cell, it may be advisable to amend the cell with fertilizer and/or organic matter to supplement the nutritional needs of sulfate-reducing bacteria.

A final wetland cell would be designed for the removal of manganese via oxidation processes. Since it is difficult to oxidize manganese because pH values must be at or greater than 7.0, this final wetland cell may require both a layer of high-calcium CaCO_3 (i.e., $\geq 90\%$ available CaCO_3) limestone beneath the organic substrate and aeration. Aeration increases the dissolved oxygen content and raises the pH through the liberation of volatile acids, primarily carbonic acid (Guertin et al. 1985). Aeration would be achieved by creating miniature waterfalls or cascades by way of elevational differences between this wetland cell and the previous cell.

Considerations for Both Treatment Methods

Howard et al. (1989) recommends that all components of the constructed wetland system be sealed or lined to prevent metals from reaching the underlying

groundwater. While neither sealing nor lining has occurred at the Challis, Idaho site, it may be appropriate to pump the existing water from the wetland cells to a holding facility, excavate as outlined above, install a plastic liner or a layer of bentonite as a sealant, add substrate, and then plant with wetland vegetation.

Mushroom compost is recommended as the substrate of choice for the anaerobic wetland cells for a number of reasons. For one, numerous field (Howard et al. 1989c, McIntire and Edenborn 1990, Wildeman and Laudon 1989) and laboratory (Schafer and Associates 1991, Spotts et al. 1992) studies have shown that mushroom compost is better capable of removing metals and raising the pH than other substrates. Wildeman and Laudon (1988) have tentatively attributed the greater effectiveness of mushroom compost to the presence of more microbiological activity. The depth of the substrate is important, given the severe winter temperatures in Challis, Idaho. At least one meter of substrate is required to maintain an unfrozen anaerobic zone, since year-round operations are planned (Howard et al. 1989c).

The most important criterion for selecting vegetation for the wetlands is for local plant species tolerant of the climate and the chemical constituents of the acid mine drainage. It is recommended that *Typha* spp. be planted in the wetland cells. Survival is expected to be high for this species as the elevation of the site is 1740 m (5720 feet), below the elevational point (2400 m or 8000 feet) where *Typha* plant survival declines. *Typha* survivability is also high when the pH of the system is above 3.0; the lowest pH recorded for the wetland cells is 4.1. *Typha* is tolerant of copper, zinc, manganese, and iron, all of which are assumed to be present in the AMD. Additionally, *Typha*

may lower the redox potential of wetlands sufficiently to precipitate iron and manganese (Sencindiver and Bhumbra 1988). *Eleocharis* and *Scirpus* spp. have been successfully transplanted in wetlands receiving AMD containing high concentrations of manganese and iron (Brodie 1990). Since *Carex aquatilis* is tolerant of high zinc and copper concentrations, and *Carex microptera* and *C. rostrata* are tolerant of high zinc concentrations (Kastning-Culp and Lockwood 1993), these species are recommended for planting in the initial wetland cell provided they are available. In addition, *Betula glandulosa* and *Salix boothii* are recommended for transplanting due to their absorptive capabilities for the probable contaminants present in the AMD at the Challis site.

Finally, the constructed wetland should be fenced upon completion of reclamation in order to limit wildlife access (Dollhopf et al. 1989). Wildlife may browse the vegetation resulting in damage or death of the plants. Additionally, if metal uptake by plants is significant, browsing of vegetation may cause adverse health effects to wildlife and waterfowl.

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APPENDIX

DATA ANALYSES FOR TAILINGS AT CHALLIS, IDAHO SITE

RANGE OF CHEMICAL LEVELS IN TAILINGS AT CHALLIS, IDAHO
RECLAMATION SITE*

Site	Zinc (ppm)	Iron (ppm)	Manganese (ppm)	Sulfur (ppm)	Copper (ppm)	Boron (ppm)
Test Hole 1	6-201 ⁺	1472-4066	79-851	NA	1-13	4-9
Test Hole 3	34-61	701-2624	79-532	NA	0-54	3-5
Test Hole 4	5-59	384-556	65-934	NA	0-4	1-6
Mill Site	3-4	27-36	11-16	17-20	1	1

*All samples analyzed by A&L Mid West Laboratories, Inc. (Omaha, Nebraska)

⁺n=3