DESIGNING AND SIZING PASSIVE MINE DRAINAGE TREATMENT SYSTEMS

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The passive treatment of contaminated coal mine drainage is a rapidly growing and evolving technology. Passive systems typically require less operation and maintenance efforts and are less expensive than conventional treatment systems. Three principal types of passive technologies currently exist for the treatment of coal mine drainage: the aerobic system, the compost wetland, and the anoxic limestone drain.

In aerobic systems, oxidation reactions occur and metals precipitate as oxides and hydroxides. Most aerobic systems are simple wetlands; they contain cattails growing in a clay or spoil substrate. However, plantless systems have also been constructed and function similarly to systems containing plants.

Compost wetlands are similar to aerobic wetlands in form, but also contain a thick organic substrate. This substrate promotes chemical and microbial processes that generate alkalinity and neutralize acidic components of mine drainage. Typical substrates used in these wetlands include spent mushroom compost, peat moss, haybales, and manure. The term "compost wetland" is general and is meant to include any wetland which contains an organic substrate in which biological alkalinity generating processes occur.

The anoxic limestone drain (ALD) adds alkalinity to the mine water by forcing it to flow through a buried bed of limestone (Turner and McCoy 1990). By keeping the limestone and mine water anoxic, limestone dissolution can occur without armoring reactions that make limestone useless in a surface environment. ALI)s are intended to generate alkalinity, and must be followed by an aerobic system in which metal oxidation and precipitation reactions occur.

Each of the three passive technologies is most appropriate for a particular type of mine water problem. Often, they are most effectively used in combination with each other. In this paper, a model is presented to aid reclaimationists in deciding whether their mine water problem is suited to passive treatment and in designing and constructing effective passive treatment systems. The model uses mine drainage chemistry to determine system design, and contaminant loadings to define system size. A flow chart of the model is shown in figure 1. The text that follows details the use of this flow chart and also discusses uncertain aspects of
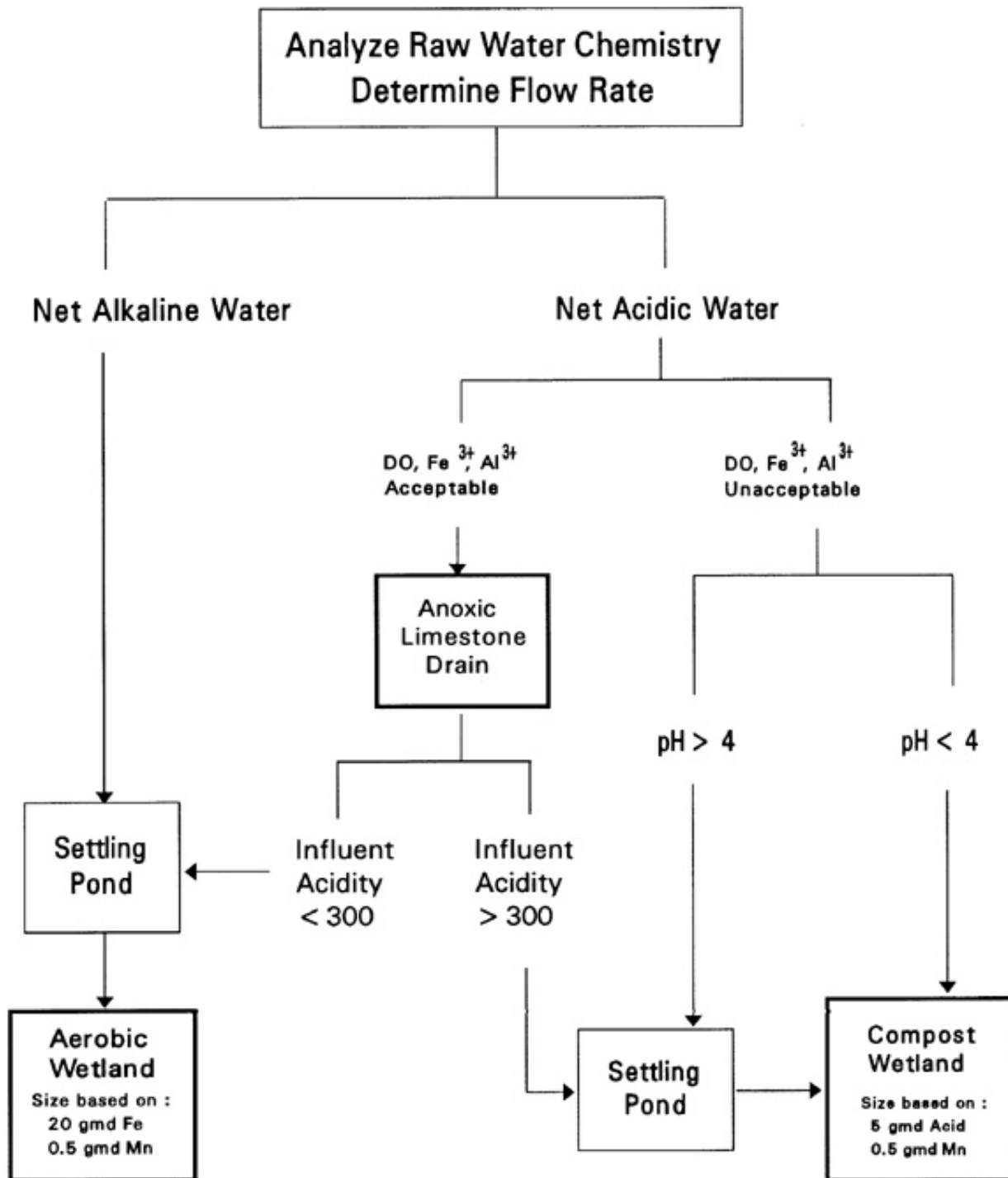
the model that are currently under investigation.

FIGURE 1. Flowchart for designing and sizing passive mine drainage treatment systems.

DESIGN AND SIZING MODEL

I. Characterize the discharge through measurements of flow and chemical analyses.

<u>Measure</u> the flow rate. Avoid visual estimates. Try to get all the water to flow through a pipe and time how long it takes to fill up a bucket of known volume. If the flow is too high for this method, use the Manning formula to estimate flow through an open channel, or install a temporary flume. Be confident that the flow measurements are accurate to within 20%.

Collect water samples for chemical analysis. Water samples should be collected at the point of seepage. Samples collected for metal analysis should be acidified in the field. Samples containing visible particulates should be filtered before acidification. Initial water analyses should include pH, alkalinity, iron, manganese, and hot acidity (1-1202 method) measurements. Accurate pH and alkalinity determinations require either field measurements or that samples be put on ice in the field and analyzed within 24 hours. The hot acidity measurements should include determinations of both acidity and alkalinity, if appropriate (when post-H_2O_2 pH is between 4.5 and 8.3). By convention, both acidity and alkalinity are expressed as mg/L CaC03 equivalent, so one value can simply be subtracted from the other. If an anoxic limestone drain is being considered, obtain measurements of ferrous iron, aluminum, and dissolved oxygen (DO) concentrations. If total dissolved iron exceeds ferrous iron, the difference represents ferric iron (Fe^{3+}).

Both the flow rate and chemistry of a discharge can vary seasonally and in response to storm events. It is important to account for this variability. Obtain measurements of flow rates and water chemistry under various seasonal and weather conditions.

II. Calculate contaminant loadings.

Calculate contaminant (Fe, Mn, acidity) loads by multiplying contaminant concentrations by the flow rate. If the flow is measured in liters per minute (Lpm), the calculation is:

[Fe,Mn,Acidity] grams/day = flow (Lpm) X [Fe,Mn,Acidity] (mg/L) X 1.44

If the flow is measured in gallons per minute (gpm), the calculation is:

[Fe,Mn,Acidity] grams/day = flow (gpm) X [Fe,Mn,Acidity] (mg/L) X 5.45

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

III. Classify the discharge as alkaline or acidic.

Determine whether the mine water is alkaline or acidic by comparing concentrations of acidity and alkalinity determined from the hot acidity measurement.

<u>Net Alkaline Water:</u> alkalinity > acidity

<u>Net Acidic</u> Water: acidity > alkalinity

Based on the mine water characterization, select an appropriate treatment option.

IV. Net Alkaline Water.

This water already contains enough alkalinity to buffer the acidity produced by metal hydrolysis reactions. The metals (Fe and Mn) will precipitate given enough time. Build an aerobic system. No compost is necessary. The goal is to aerate the water and promote metal oxidation processes. In many existing treatment systems where the water is net alkaline, the removal of iron appears to be limited by dissolved oxygen concentrations. Incorporate features that aerate the drainage, such as waterfalls or rip rap ditches. Size the system for Fe removal based on a removal rate of 20 grams/M2 /day (gmd).

Minimum wetland size (m^2) = Fe loading (g/day) / 20

If Mn is a concern, size the wetland based on a Mn removal rate of 0.5 gmd.

Minimum wetland size (m^2) = Mn loading (g/day) / 0.5

If both Fe and Mn removal are necessary, add the two wetland sizes together.

When building an aerobic wetland, always keep in mind the goals of the system: to aerate the water and to retain the water long enough for oxidation and precipitation reactions to occur. A typical aerobic wetland is constructed by planting cattail rhizomes in soil or alkaline spoil that is obtained on-site. Some systems have been planted by simply spreading cattail seed heads, with good plant growth attained after 2 years. The depth of the water in a typical aerobic system is 6-18 inches. Often, several wetland cells are connected by flow through a V-notch weir, railroad tie steps, or down a ditch. Recently, aerobic treatment systems have been constructed that do not have typical "wetland" features. Several systems have been constructed that have little emergent plant growth. Flow is through open water ditches or ponds. In one case, where Mn is targeted for removal, water flows through a pond filled with limestone rocks. Removal of metals in these plantless, aerobic systems appears to occur at similar rates as it does in alkaline aerobic systems containing plants (Hedin and Nairn, in press). However, we still recommend that plants be included because they may help filter particulates, prevent flow channelization and provide wildlife benefits that are valued by regulatory and environmental groups.

V. Net Acid Water.

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

ALDs produce alkalinity at a lower cost than do compost wetlands. However, not all water is suitable for pretreatment with ALDs. The primary chemical factors believed to limit the utility of ALDs are the presence of ferric iron (Fe^{3+}), aluminum (Al^{3+}) and dissolved oxygen (DO). When acidic water containing any Fe^{3+} or Al^{3+} contacts limestone, both metals hydrolyze and precipitate. No oxidation is necessary. Ferric hydroxide can armor limestone, limiting its further dissolution. Whether aluminum hydroxides armor limestone has not been determined. The buildup of both precipitates within the ALD may eventually decrease the drain permeability and cause plugging. The presence of any DO in mine water will promote the oxidation of ferrous iron to ferric iron within the drain, and thus potentially cause armoring and plugging. While the short-term performance of ALDs that receive water containing elevated levels of Fe^{3+}, Al^{3+} or DO can be spectacular (total removal of the metals), the long term performance and longevity of these systems is questionable.

Although some mine waters appear to be ideal for pretreatment with ALDs, the use of an ALD for other waters will most likely result in failure in a relatively short time. For example, an ALD constructed with #57 (size 213) limestone received mine water containing 25 mg/L Al". Flow through the ALD was compromised in less than a year of operation. The mine water seeped through the clay spoil material behind the ALD rather than flowing through the limestone gravel. The use of larger #3 and #4 limestone may have allowed the precipitates to pass through the ALD.

If, to whatever extent, mine water contains DO, Fe^{3+} or Al^{3+} the effective lifetime of an ALD will probably be compromised in some way. Although the limiting concentrations or loads of these parameters are unknown at the present time, it is apparent that an ALD is inappropriate. Given the current understanding of both treatment methods, the following criteria are suggested for determining the most cost-effective method for treating acidic water.

1. DO, Fe^{3+} and Al^{3+} acceptable for an ALD.

Mine water with low levels of DO, Fe^{3+} and Al^{3+} is currently considered to be suitable for pretreatment with an anoxic limestone drain. Mine

water intercepted after it contacts the atmosphere usually will not fulfill these criteria. In an ALD, alkalinity is produced when the acidic water contacts the limestone. It is considered important to use limestone with a high $CaCO_3$ content because of its higher reactivity compared to a limestone with a high $MgCO_3$ or $CaMg(CO_3)_2$ content. Most effective systems have used #3 or #4 (baseball size) limestone. Some systems constructed with limestone powder and gravel have failed, apparently because of plugging problems. The ALD must be sealed so that inputs of atmospheric oxygen are minimized and the accumulation of carbon dioxide within the drain is maximized. This is usually accomplished by burying the ALD under several feet of clay. Plastic is commonly placed between the limestone and clay as an additional gas barrier. In some cases, the ALD has been completely wrapped in plastic before burial (Skousen and Faulkner 1992). The ALD should be designed so that the limestone is inundated with water at all times. This has been accomplished with clay dikes within the drain or riser pipes at the outflow of the drain.

The design of ALDs has varied. Most older ALDs were constructed as long narrow drains, approximately 2-3 feet wide. At sites where linear drains were not possible, anoxic limestone beds have been constructed that are wider (30-50 feet). The bed systems have produced alkalinity in similar concentrations as the drain systems.

The proper sizing of ALDs is uncertain. Theoretical calculations can be performed to estimate the mass of limestone required to neutralize a certain discharge for a specified period of time. Important to these calculations is the alkalinity concentration expected to be produced by ALDs. A maximum value of approximately 300 mg/L has been observed at ALDs constructed recently. The minimum mass of limestone needed to treat a year's flow of mine water can be calculated from the flow rate and an assumption that the drain will produce 300 mg/L alkalinity.

> yearly $CaCO_3$ = flow (Lpm) X 158
> consumption (kg)

or, where flows are measured in gallons and masses are calculated in tons:

> yearly $CaCO_3$ = flow (gpm) X 0.6565
> consumption (tons)

Limestone used in ALDs has a density of about 1.0-1.5 ton/yd^3.

In order to determine the total mass of limestone needed in the drain, the above calculation must be adjusted for the $CaCO_3$ content of the limestone and the projected lifetime of the ALD.

When these calculations are done properly and carried through into the

actual construction of the ALD, they assure that there is <u>theoretically</u> enough limestone in the drain to generate alkalinity for the time period considered. Because the oldest ALDs are only 3-4 years old, it is difficult to assess how realistic these theoretical calculations are. Questions about the ability of drains to maintain unchannelized flow for a prolonged period of time, whether 100% of the $CaCO_3$ content of the limestone can be expected to dissolve, whether the drains will collapse after significant dissolution of the limestone, and whether inputs of DO that are not generally detectable with standard field equipment (0-1 mg/L) might result in armoring of the limestone with ferric hydroxides, have not yet been addressed.

The anoxic limestone drain is one component of the passive treatment system. When the ALD operates ideally, its only effect on mine water chemistry is to neutralize low pH and increase concentrations of calcium and bicarbonate alkalinity. Dissolved Fe and Mn should be unaffected by flow through the drain. The ALD must be followed by a settling basin or wetland system in which metal precipitation reactions can occur. The type of post-ALD treatment system depends on the acidity of the mine water. Few ALDs regularly produce more than 300 mg/L alkalinity and many ALDs produce less than 200 mg/L alkalinity. Currently, there are no design or construction guidelines that will ensure generation of a certain amount of alkalinity by an ALD. If the mine water has a net acidity greater than 300 mg/L, it is unlikely that an ALD will produce enough alkalinity to neutralize all the acidity. Thus,

   a. if the net acidity of the initial influent mine water < 300 mg/L,

it is likely that an ALD will add enough alkalinity to the water to make the effluent of the drain net alkaline. The drain effluent can then be treated with a settling basin and an aerobic wetland. If possible, the water should be aerated as soon as it exits the ALD and directed into a settling pond. Follow the settling pond with an aerobic wetland. Size the total post-ALD system according to the criteria provided earlier for net alkaline mine water.

   b. If the net acidity of the initial influent mine water > 300 mg/L,

it is unlikely that an ALD would generate enough alkalinity to totally neutralize the acidity contained in this mine water. Building a second ALD, to recharge the mine water with additional alkalinity after it flows out of the aerobic system, is currently not feasible because of the high dissolved oxygen content of water flowing out of aerobic systems. If the treatment goal is to neutralize all of the acidity passively, then a compost wetland should be built so that additional alkalinity will be generated. Such a treatment system contains all three passive technologies. The mine water flows through an ALD, into a settling pond and an aerobic system, and then into a compost wetland. Size the

complete system based on 20 gmd removal of Fe and 5 gmd removal of the excess acidity, as explained in the next section.

Minimal size (m^2) = Fe loading/20 + Acidity Loading$_{ALDeff}$/5

where the acidity loading is the net acidity of the ALD effluent. A preconstruction estimate of this value is obtained by subtracting 300 (the estimated alkalinity generated by the drain) from the original mine water acidity.

If Mn removal is desired, an aerobic wetland sized based on 0.5 gmd of Mn removal is required after the compost wetland. To our knowledge, Mn has never been successfully treated by passive methods where the influent mine water is net acidic.

2. DO or Fe^{3+} or Al^{3+} unacceptable for an ALD.

Construction of a compost wetland is recommended. Compost wetlands generate alkalinity through a combination of bacterial activity and limestone dissolution. The desired sulfate-reducing bacteria require a rich organic substrate in which anoxic conditions will develop. Limestone dissolution also occurs readily within this anoxic environment. A common substance used in these wetlands is spent mushroom compost, a substrate that is readily available in western Pennsylvania. However, any well-composted equivalent should serve as a good bacterial substrate. Spent mushroom compost has a high $CaCO_3$ content (about 10% dry weight), but mixing in more limestone may increase the alkalinity generated by $CaCO_3$ dissolution. Compost substrates that do not have a high $CaCO_3$ content should be supplemented with limestone.

The compost should be 12-18 inches deep. Experimental systems with greater depths of compost have been constructed, but it has not been determined whether higher rates of alkalinity generation occur. Most compost wetlands are planted with cattails by the same methods used in aerobic wetlands.

Compost wetlands in which water flows on the surface of the compost remove acidity (e.g. generate alkalinity) at rates of approximately 2-12 gmd. This range in performance is largely a result of seasonal variation: lower rates of acidity removal occur in winter than in summer (Hedin et al. 1991). There is recent evidence that supplementing the compost with limestone and incorporating system designs that cause most of the water to flow through the compost (as opposed to on the surface) may result in higher rates of limestone dissolution and better winter performance (Hedin and Nairn in press; unpublished BOM data).

For sizing purposes, we currently recommend using a rate of acidity

removal of 5 gmd for compost wetlands.

Minimum Wetland Size m^2) = Acidity Load/5

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

UNCERTAINTIES IN THE DESIGN AND SIZING MODEL

There are two uncertain components of the design and sizing model that should be appreciated. The model indicates that for waters with a net acidity greater than zero, the incorporation of alkalinity-generating features (either an ALD or a compost wetland) is necessary. The successful treatment of mine waters with net acidities of 0- 100 mg/L using aerobic wetlands has been reported (Brodie et al. 1990). In these systems, alkalinity either enters the treatment system with diluting water or alkalinity is generated within the system by undetermined processes. Currently, there is no method for predicting which of these marginally acidic waters can be successfully treated with only an aerobic system.

The model indicates that ALDs should not be constructed for mine waters that contain substantial concentrations of DO, Fe^{3+}, or Al^{3+}. These criteria are based on the current belief that the presence of any one of these parameters will result in the premature failure of the ALD due to clogging or limestone armoring. How soon ALD failure will occur because of these parameters is partly a function of how high the concentrations are. An ALD built to treat water containing 300 mg/L Fe^{3+} and 300 mg/L Al^{3+} would likely fail much more quickly than an ALD built to treat water containing 3 mg/L Fe^{3+} or 3 mg/L Al^{3+}. No methods are currently available to economically remove these contaminants before passage through the ALD, or to account for the effect that these contaminants have on ALD longevity. Builders of ALDs should realize that as concentrations of these parameters increase beyond 1 mg/L, the risk that the ALD will fail prematurely also increases.

IMPROVEMENT VS. COMPLIANCE

The contaminant removal rates used in the model to size aerobic systems and compost wetlands are empirical and are based on data collected by the Bureau of Mines from existing constructed wetlands. Some of the data has been collected from systems that do not consistently lower concentrations of contaminants to compliance levels (i.e. pH between 6 and 9, alkalinity > acidity, Fe < 3 mg/L and Mn < 2 mg/L). In particular, the Fe sizing factor for alkaline mine water (20 gmd) is based on data from three sites, only one of which lowers Fe concentrations to compliance. All three sites, however, remove iron at approximately the same rate (Hedin and Nairn, in press). It is possible that Fe removal rates are a function of Fe

concentration, i.e. as concentrations get lower, the size of system necessary to remove a unit of Fe contamination (e.g. 1 g/day) gets larger. To account for this possibility, we suggest that systems being designed for Fe compliance be sized using a 10 gmd removal rate. This value is in agreement with the findings of Stark et al. (1990) for a constructed wetland in Ohio that receives marginally acidic water. This rate is still larger, by a factor of 2, than the Fe removal rate reported by Brodie et al. 1990) for aerobic systems in southern Appalachia that regularly meet compliance.

The manganese removal rate used in the model, 0.5 gmd, is probably accurate for systems designed to meet compliance. This removal rate is based on the performance of four treatment systems, three of which consistently lower Mn concentrations to compliance levels.

The acidity removal rate presented for compost wetlands is influenced by seasonal variation that cannot currently be corrected with wetland design. We know of no compost wetland that consistently transforms highly acidic water (> 300 mg/L acidity) into alkaline water. One of our study sites, which receives water with an average 600 mg/L acidity and does not need to meet a Mn standard, discharges water that requires chemical treatment only during winter months. While considerable cost savings are realized at the site due to the compost wetland, the passive system must be supported by conventional treatment during a portion of the year.

Because the performance of passive treatment systems is currently uncertain, current regulations require that the capability for chemical treatment exist at all bonded sites. This provision is usually met by placing a "polishing pond" after the passive treatment system. Note that the design and sizing model does not currently account for such a polishing pond.

In summary, over 400 passive mine drainage treatment systems have been constructed in the United States, but the technology is still evolving and developing. While the effluent of many systems may not meet compliance standards, the improvement in water quality results in considerable cost savings. At many abandoned sites, passive treatment provides the only means of improving the quality of the mine discharge. The following summary of relevant literature may be useful to those involved in the passive treatment of contaminated mine drainage.

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