United States
Environmental Protection
Agency

Engineering Issue

# Management and Treatment of Water from Hard Rock Mines

## Index

## 1.0 PURPOSE

The U.S. Environmental Protection Agency (EPA) Engineering Issues are a new series of technology transfer documents that summarize the latest available information on selected treatment and site remediation technologies and related issues. They are designed to help remedial project managers (RPMs), on-scene coordinators (OSCs), contractors, and other site managers understand the type of data and site characteristics needed to evaluate a technology for potential applicability to their specific sites. Each Engineering Issue document is developed in conjunction with a small group of scientists inside the EPA and with outside consultants, and relies on peer-reviewed literature, EPA reports, Internet sources, current research, and other pertinent information. The purpose of this document is to present the "state of the science" regarding management and treatment of hard-rock mines.

Internet links are provided for readers interested in additional information; these Internet links, verified as accurate at the time of publication, are subject to change.

## 2.0 SUMMARY

Contaminated water draining from hard rock mine sites continues to be a water quality problem in many parts of the U.S. The types of water range from strongly acidic water laden with metals, to variable water quality in mining pit lakes, to alkaline water being released from closed cyanide heap leach operations.

Prevention of water contamination at mine sites is usually the best option and can sometimes be realized by appropriate management of waste material, or by hydrologic control in underground systems, or by using a variety of capping methods for waste rock dumps or closed heaps.

However, long-term (decades and beyond) treatment is, and will continue to be, required at many sites. Once a contamination source is established (e.g., reactive waste rock dumps), elimination of these as sources of water is often very expensive and technically challenging.

Acid drainage remains the most problematic water quality, in large part due to the ability of acidic water to dissolve a variety of toxic metals (e.g., cadmium, zinc, nickel) and release of that water to surface or ground water. The most common treatment is neutralization using lime, or another suitable alkaline agent, followed by oxidation and precipitation of metals. This will also reduce sulfate to near the gypsum solubility limit (approximately 2,000 mg/L, depending on

calcium concentrations). A variety of methods have been utilized to add lime to acidic water, and, particularly for large flows (>100 gal/min) and/or high acidity/metals loadings, this option is usually the most cost effective.

Other methods for treatment of acidic water include a variety of wetland systems and bioreactors that are based on sulfate reducing bacteria that reduce sulfuric acid to hydrogen sulfide, which consumes acidity and allows precipitation of metals as metal sulfides. These systems can either utilize the wetland organic carbon or an exogenously supplied carbon source (e.g., ethanol) for sulfate reduction. These systems show particular promise where the flows and acidities are relatively low. The advantage of these systems is that they commonly do not require the same level of monitoring and operational expense as the lime systems. They also can reduce sulfate levels to well below the gypsum solubility limits, depending on the characteristics of the bioreactor/wetland system utilized.

Drainage from precious metals heaps and tailings facilities offers a different set of challenges. While most of the precious metals heaps and tailings are not acid generating, several examples of acid generating processing wastes exist in western states. In most cases in mine closures, the residual water used in cyanide extraction of precious metals remains net alkaline, and was continuously recycled during operation. The soluble constituents were concentrated as water evaporated, and often contain elevated sodium from the sodium cyanide used in the process. Thus, land application of these fluids should be limited, due to salts, arsenic and other constituents. Other than ion removal technologies (e.g., reverse osmosis), few cost effective methods for treatment and release of these water are available. In arid regions, evaporation is often the only option available for such heaps and tailings facilities.

Mining pit lakes that are derived from open pit mines that penetrated ground water are particularly prevalent in the precious metals and copper mines in the western United States. The water quality can vary from a highly acidic system in high sulfide host rock to slightly alkaline, better quality water in carbonate host rock. While treatment of pit lakes is potentially expensive, at least one example of neutralization of an acidic pit lake (Sleeper Pit Lake in Nevada) has demonstrated that this is technically possible.

Because of the long-term nature of many of these drainages, methods for cost-effective treatments are still needed. Many of the presently available technologies have been derived from coal mine drainage research. While effective in many cases, further research is required to reduce water treatment costs and increase the reliability of these technologies.

## 3.0 INTRODUCTION

This Engineering Issue document on treatment of mining waters is a practical guide to understanding and selecting technologies for the environmental management of waste materials and effluents at hard-rock mines. For the purposes of this discussion, hard-rock mining primarily refers to open pit and underground mines that produce base metals (e.g., copper, zinc, lead) and precious metals (e.g., gold and silver). While drainage from coal mines has similar water quality issues, coal drainage has been considered extensively in other publications. It responds to the need for environmental management at new and abandoned hard-rock mines by providing guidance for selecting among available technologies for the stabilization of mine waste, treatment of mine water, and management of mine pit lakes. Target audiences are operators, regulators, stakeholders, and technical consultants involved in selecting technologies for environmental management of hard-rock mines. The general contents of this Engineering Issue document are listed above in the Table of Contents.

The goal of this document is to increase the efficiency of decision makers in defining the scope of mine-related water quality problems and selecting the least expensive *effective* management technology. It begins with technical overviews and conceptual models of contaminant sources (i.e., environmental behavior in the dominant hard-rock mining facilities—waste rock and heap leach facilities, tailings impoundments, and pit lakes). A general overview of remedial technologies (acid neutralization, biologically induced sulfide treatment, and pit lake management) follows. With these technical foundations reviewed, specific remedial technologies are presented individually and described using the context of the feasibility study process—a practical framework for selecting remedial technologies based on *implementability*, *effectiveness*, and *cost*.

### *3.1 Background: Environmental Problems at Hard-Rock Mines*

Few environmental problems are as widely documented as the legacy of historic hard-rock mines. Small mines, operating in the era before environmental regulation, removed and milled ore primarily from vein deposits, leaving unvegetated spoils, unsealed adits, and often-acidic seepage laden with metals. Modern U.S. hard-rock mining is in sharp contrast, with closure designs and associated financial bonds for environmental management commonly required even before operations begin and water discharges permitted only within the constraints of the Clean Water and Safe Drinking Water Acts. Environmental issues remain, of course, and economies of scale have produced larger operations, but today's mines are better managed—water is treated, and waste is capped and revegetated. Closure requirements for modern mines depend on the regulatory requirements and the environmental capabilities and risk of the associated mine.

Collectively, the economic liabilities and technical challenges of hard-rock mining are immense (if poorly constrained):

> *Although no global estimation of the impact of acid drainage exists, total liability costs for potentially acid-generating wastes at mining sites is estimated to be US\$530 million in Australia, between US\$1.2 and 20.6 billion in the USA, and US\$1.3 and 3.3 billion in Canada. Effectively dealing with acid drainage has been—and continues to be—a formidable challenge for which no global solutions currently exist. Acid drainage is one of the most serious and potentially enduring environmental problems of the mining industry. Left unchecked, it can result in such extensive water quality impacts that it could well be this industry's most harmful legacy. (INAP, 2004)*

There remains an enormous need for development and evaluation of effective low-cost technologies for stabilization and treatment of mine waste.

## *3.2 Conceptual Models at Hard-Rock Mines*

The main processes responsible for water quality degradation at hard-rock mines are reviewed here briefly to provide a foundation for understanding remediation technologies. Mine water contamination comes from two sources: release of constituents contained in rock that has been mined and chemical reagents used in mining, milling, extraction, and ultimate recovery of the valuable metal or mineral. The largest source of water contamination is nearly always the material being mined. Ore and waste rock has generally been isolated from oxygen and water for geologic time frames, and bringing the material to the surface potentially results in reactions that release contaminants that degrade water quality.

Acidic drainage is the dominant environmental problem associated with hard-rock mining. Many valuable metals in ore deposits are bound to sulfide sulfur, forming sparingly soluble sulfide minerals such as sphalerite (ZnS), covellite (CuS), or galena (PbS). Acidic drainage forms primarily when iron sulfide, pyrite ($FeS_2$), comes in contact with water and oxygen, producing dissolved sulfuric acid and iron. The three steps summarizing this overall reaction are:

$$FeS_{2(S)} + \tfrac{7}{2}O_2 + H_2O = Fe^{2+} + 2SO_4^{2-} + 2\,H^+ \tag{3-1}$$

$$Fe^{2+} + \tfrac{1}{4}O_2 + H^+ = Fe^{3+} + \tfrac{1}{2}H_2O \tag{3-2}$$

$$Fe^{3+} + 3H_2O = Fe(OH)_{3(S)} + 3H^+ \tag{3-3}$$

Pyrite oxidation liberates soluble iron ($Fe^{2+}$) and acidity ($H^+$) and sulfate, with the primary limit on the oxidation rate being the availability of oxygen. The oxidation process also liberates other sulfide-bound metals (e.g., cadmium, zinc, copper, lead, uranium) and metalloids (e.g., arsenic, antimony, selenium). In addition, most metals are more soluble under acidic conditions (i.e., at low pH), so oxidation and acid production tend to be associated with increasing metal concentrations. The result is that acidic conditions (low pH) in mine effluent tend to be highly correlated to elevated heavy metal concentrations.

The primary offset to acid production in natural systems is the consumption of acidity by calcite ($CaCO_3$):

$$CaCO_3 + 2H^+ = Ca^{2+} + CO_{2(G)} + H_2O \tag{3-4}$$

and precipitating sulfate as calcium sulfate ($CaSO_4$).

$$SO_4 + Ca^{2+} = CaSO_{4(S)} \tag{3-5}$$

Acidic mine water that is neutralized by reaction with calcite generally contains 1,500–2,200 mg/L sulfate. Further, the consumption of acidity in Equation 3-4 increases the pH, which tends to decrease total heavy metal concentrations as these constituents precipitate or adsorb to surfaces.

Figure 3-1 is a model simulation illustrating how pore water in waste rock or tailings will change as pyrite and calcite are consumed during oxidation. The top graph shows the amount of acid-generating potential (AGP) (as pyrite) and acid neutralizing potential (ANP) from calcite ($CaCO_3$) remaining, with increasing oxidation (also related to increasing time) shown along the graph from left to right. As long as calcite remains, pore water pH remains near neutral and sulfate concentrations are limited to below 3,000 mg/L. Once all calcite is consumed, acid buffering ceases, the pH drops to below 3, and dissolved sulfate increases. Results indicate the dramatic change in water quality that can occur when excess AGP remains in materials.

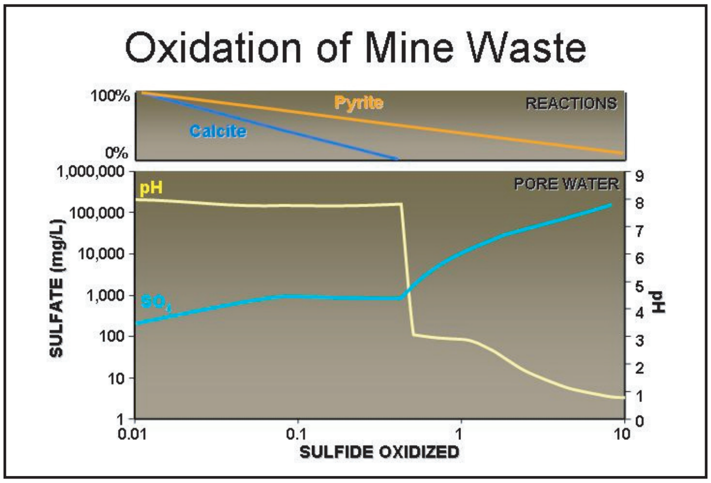

**Figure 3-1.** Oxidation of mine waste and production of sulfate. This idealized 10% sulfide rock contains both pyrite and calcite and demonstrates how sulfate concentration and pH are changed as the sulfides in the rock are oxidized and the capacity to neutralize the acid is consumed.

Contaminated mine waters may also be neutral to alkaline, depending on the type of rock being mined and the reagents used to selectively extract the valuable substance. Thus rock with excess calcite will produce pH-neutral effluent. However, neutralized mine waste effluent can still contain elevated metalloids, such as selenium, arsenic, and antimony. Zinc and other heavy metals have also been observed in pH-neutral mine waters (e.g., the Burleigh Tunnel and Wellington Oro Mine discharges in Colorado). Metal recovery reagents that may present water quality issues include a variety of flotation agents for concentration of metals, as well as lixiviants, particularly cyanide. In the latter case, cyanide can form complexes with a variety of metals that are very weak (e.g., zinc cyanide) to strong complexes (e.g., cobalt, iron, and mercury cyanides) and also transformation products of cyanide, particularly thiocyanate and nitrate.

### 3.3 The Process of Selecting Remedial Technologies

Selection of an optimal technology for a specific remediation problem would, ideally, follow from tightly constrained algorithms or flow charts. Regrettably, critical decision variables, such as cost per cubic meter to treat water, net percolation through caps, etc., are generally too dependent on site-specific conditions to allow direct transfer between projects. *Design methods are transferable across sites, but not specific designs.* In response, remedial technologies discussed in this document are presented in a format that supports the EPA's feasibility study process. An overview of each technology (Tables 4-1, 4-5, and 4-6) is provided to facilitate early screening of inappropriate options. Details of each technology are presented in the text, with a focus on identifying those parameters most critical in evaluating implementability, effectiveness, and cost. Where possible, specific examples of cost and effectiveness under pilot- or full-scale implementation are provided.

The feasibility study process provides a framework for selecting from a range of remedial technologies for specific site conditions amidst the interests of regulators, stakeholders, and technology developers. The process begins with a characterization of the problem (e.g., chemicals of concern, risks, exposure paths, identification of remedial goals, etc.), then identifies potential technologies (screening process), and finally evaluates the feasibility of a short list of technologies to select a remedy.

The primary technical evaluation criteria for feasibility under Superfund (EPA, 1988) are:

- **Effectiveness**—the potential for the alternative to achieve remedial goals established for the site.
- **Implementability**—the ability to comply with technical and administrative issues and constraints involved in implementing a technology at a specific site.
- **Cost**—typically an estimate of net-present cost for each technology.

In practice, implementers identify the technologies that can meet their water quality goals ("effectiveness"), eliminate those that can't be applied for practical reasons ("implementability"), then implement the least expensive option ("cost"). This document is intended to support this technology selection process, providing descriptions of the common environmental technologies for hard-rock mining and identifying the critical components affecting the feasibility of each.

Selecting a technology can be more difficult than implementing it. The critical components in the evaluation and selection of a technology include:

- **Source definition**—water flow rates, material mass, solute concentrations, expected duration, etc.
- **Identification of environmental goals**—discharge standards, compliance points, and human or ecological risk thresholds.
- **Identification of applicable technologies**—those technologies potentially capable of meeting goals.

- **Identification of critical parameters**—early determination of values for parameters that typically drive cost or effectiveness.
- **Impartial evaluation**—a feasibility analysis that is completely independent from technology vendors.

Other feasibility study evaluation criteria that should be considered during the technology selection process are community and regulatory acceptance. These criteria are covered to a lesser degree in this document than the technical criteria, but can have a great impact on the final selection.

### 3.4 Resources for Additional Information

Below are prominent organizations dedicated to research on the causes and remedies for management of drainage at hard-rock and coal mines:

- Mine Environment Neutral Drainage (MEND): http://www.nrcan.gc.ca/mms/canmet-mtb/mmsl-lmsm/mend/default_e.htm.
- Acid Drainage Technology Initiative (ADTI): http://www.unr.edu/mines/adti/.
- International Network for Acid Prevention (INAP): http://www.inap.com.au/. Includes clear overview of topics and reports on INAP-funded research.
- Australian Center for Mining Environmental Research (ACMER): http://www.acmer.com.au/.
- The U.S. Geological Survey Toxic Substances Hydrology Program section on Hard-Rock Mining Contamination: http://toxics.usgs.gov/topics/minelands.html.
- The Restoration of Abandoned Mine Sites Technology Database (RAMS tech): http://www.unr.edu/mines/ramstech/techintro.asp.
- U.S. Army Corps of Engineers Restoration of Abandoned Mines Sites (RAMs) Western Region: https://www.nwo.usace.army.mil/html/rams/rams.html. Includes project summaries and documents for nine western states
- West Virginia University Extension Service Land Reclamation Program: http://www.wvu.edu/~agexten/landrec/land.htm#ACID. Focuses on acid drainage from coal sites.
- EPA's Web site for abandoned mine lands: http://www.epa.gov/superfund/programs/aml/. Gives current technology updates on application of new treatment methods.
- EPA's Web site for the EPA/DOE Mine Waste Technology Program: http://www.epa.gov/minewastetechnology.

The following publications provide additional information on innovative acid mine drainage treatment technologies and a comprehensive summary of recent closure and bonding practices and related costs at hard-rock mines, respectively:

- "Acid Mine Drainage: Innovative Treatment Technologies," by Christine Costello for the EPA (Costello, 2003): http://clu-in.org/s.focus/c/pub/i/1054/.
- "Hardrock Reclamation Bonding Practices in the Western United States," by the Center for Science in the Public Interest for the National Wildlife Federation (Kuipers, 2000): http://www.csp2.org/REPORTS/Hardrock%20Bonding%20Report.pdf.

## 4.0 TECHNOLOGY DESCRIPTIONS

The strategy for management and protection of water quality at mine sites varies on a site-specific basis. The range of contaminants that is released into water from one ore may be very different from another ore, and the management methods utilized will depend on the volume and environmental characteristics. However, there are common threads that are fundamental to water management, and sufficient similarities exist that a generalized discussion is useful. While a large variety of metals are mined in the U.S., nearly 90% of the value of metals (excluding iron ore) consists of a group including gold, copper, zinc, lead, silver, and molybdenum (USGS, 2006). As such, this discussion will primarily consider waters released from these important metals mines.

Specific remedial technologies are divided into three categories:

1. **Source Control:** typically the chemical stabilization of reactive rock, or physical isolation or diversion of water away from the mine waste.
2. **Water Treatment:** methods of reducing contaminants in mine waters or otherwise managing contaminated water to reduce impacts to humans and the environment.
3. **Pit Lake Management:** treated separately here due to the unique physical characteristics of lakes, although many elements of source control and treatment also apply.

The descriptions of individual technologies under each category are intended to provide enough information to design a feasibility study at a particular site. This includes specific information on target analytes, treatment efficiency, examples of field-scale applications, critical parameters, and references for additional information. Key information on the feasibility of each technology is also summarized in tables. References are provided for supporting information, and links to reliable Web pages are included throughout.

## *4.1 Source Control*

Treatment of contaminated mine waters is very often a long-term commitment, and resources will be required for the duration of the flows. Reducing or eliminating these flows through source control methods clearly has long-term benefits and should always be considered for determining the appropriate method for surface and ground water protection. However, the uncertainty of successfully implementing source control and costs of preventing releases of contaminated water also needs to be evaluated as a component of the water quality management decision. In some cases, source control needs to be considered early in the mine development; in other cases, it can be applied to mine sites that have long been closed.

Source control can be applied to two broad categories of drainage waters: drainage from surface waste facilities and drainage from underground workings (e.g., adits). Both are ultimately supplied by meteoric (precipitation runon/runoff) water, so flow rates are related to precipitation. However, surface waste facilities generally have a more local response to rainfall and can be managed by appropriate caps, while drainage from underground workings requires consideration of the regional ground water system.

### 4.1.1 Capping and Revegetation for Source Control

Capping and revegetation technologies seek to reduce or eliminate the flow of water and oxygen into surficial mine waste, producing a corresponding decrease in the production and transport of solutes out of these potential sources. “Store-and-release” caps are simply vegetated surface layers of material with a high moisture-retention capacity that store water in the cap until it can be transpired or evaporated. The goals are to minimize net percolation, support vegetation, reduce erosion, and isolate acid-generating rock from the surface. These caps can dramatically reduce, although not eliminate, net percolation of water into mine waste.

Suitable cap materials include topsoil, run-of-mine waste rock, or waste rock amended to improve its performance (e.g., with nutrients to enhance plant growth, with fine material or tailings to increase water retention, or with alkali reagents to offset acid-generating potential). Selection criteria include moisture retention characteristics (measured directly or estimated from particle-size distribution), shear strength (measured in a laboratory), and acid-generating potential (AGP) (chemical analysis). Benign waste rock, with or without amendment, is a particularly attractive cap material because it is typically readily available and often strong enough to resist erosion on slopes. Vegetation type is entirely site specific, but seed mixtures typically focus on perennials that are efficient at extracting water, have deep roots, are drought resistant, and are consistent with post-reclamation land use.

Inhibition of oxygen is often cited also as a goal of mine waste caps, as oxygen flux is approximately proportional to acid rock drainage (ARD) formation. Diffusive oxygen flux into waste rock facilities typically produces several kg sulfate/m$^2$-yr for at least tens of years. Measurements inside waste rock indicate that oxygen advection through coarse zones may be a larger oxygen source (Andrina et al., 2003), so total oxidation rates in mine waste facilities may be several times higher. However, long-term oxygen exclusion has been demonstrated only with subaqueous disposal. Void space in waste rock is typically 40% (Wilson et al., 2000b), and the high air diffusivity of oxygen (~10,000 times greater than in water) allows rapid oxygen transport. Oxygen-consuming layers (e.g., wood chips) are effective, but have a very short life. Although models indicate that water-saturated zones could be maintained, even in semi-arid climates, other variables often make them impractical. Short of subaqueous disposal, no practical cap designs currently provide complete long-term barriers to oxygen. Wet covers are facilities that maintain a permanent water body above reactive mine waste—a form of subaqueous storage. They have been found to minimize oxidation and release of contaminants; a numerical analysis study utilizing modeling concluded, “a water cover alone leads to a reduction of approximately 99.1%, in the [sulfide] oxidation rate relative to uncovered tailings” (Romano et al., 2003). However, such caps generally require perpetual management to ensure continued water saturation of the surface.

One important caveat for cap effectiveness is that reducing net percolation in acid-generating material may delay the onset of impacts, but not the magnitude, as pore waters become more concentrated in slower-flowing waste (Ritchie, 1994). Decreasing flow in mine waste does in fact allow more time for attenuation reactions, including silicate mineral buffering and precipitation of sulfate salts, so reducing net percolation may in many cases reduce environmental impacts from mine waste. However, reduced infiltration is not a guarantee of reduced impacts.

Finally, two theoretical technologies potentially offer walk-away designs. An *umbrella design* with sloping layers of fine material—the most conductive in unsaturated waste (Wilson et al., 2000a)—could shed water around net acid-generating rock (Barbour, 2000). Potential drawbacks are that this design still requires select handling of acid-generating rock and that low shear strength of fine materials may limit its applicability on steep slopes. Second is a *tailing and waste rock blend design* (~$\frac{1}{3}$ tailings and $\frac{2}{3}$ waste rock). This material has shear strength comparable to waste rock, but with moisture retention high enough to maintain saturated conditions, providing a long-term barrier to oxygen introduction (Wilson et al., 2000b; Wilson et al., 2003). Potential limits may include high blending costs, long-term physical stability of blends, and suitability of such blends for revegetation. Neither has been demonstrated in field-scale tests, and they are not considered further in this document.

Characterization requirements for mine waste caps include the following:

- Climate (daily temperature, precipitation, humidity, potential evaporation, and insolation)
- Reclaimed vegetation mix (post-reclamation species and their root depth and leaf area index)
- Availability of suitable cover (waste rock, soil, tailings, and limestone)
- Physical characteristics of cap (particle-size distribution, Atterburg limits, specific gravity, compaction curve)
- Hydraulic characteristics of cap (saturated hydraulic conductivity and soil water characteristic curve)
- Moisture-retention characteristics of proposed cap material (can be estimated from particle-size distribution or determined more reliably with pressure-plate laboratory hydraulic tests)

Design and analysis of store-and-release caps can be conducted with models (e.g., HELP for screening-level analysis and SoilCover or Vadose/W for more refined analysis—see the review by O'Kane and Barbour, 2003).

***Key Web Site References***

- Overview of dry covers for mine waste, available online at INAP (O'Kane Consultants, 2003): http://www.inap.com.au/completed_research_projects.htm
- "Design, Construction, and Performance Monitoring of Cover System for Waste Rock and Tailings," a comprehensive, five-volume design and monitoring report (MEND Report 2.21.4): http://www.nrcan.gc.ca/ms/canmet-mtb/mmsl-lmsm/mend/mendpubs-e.htm

### 4.1.2 Plugging Drainage Sources and Interception of Drainage by Diversion Wells

Because these systems often intercept ground water and can even change the hydrologic system, source control is limited, complicated, and uncertain. Two general approaches are often considered: interception of water from the underground workings and plugging the drainage routes from the underground workings.

***Plugging of drainage routes:*** Plugging of adits and grouting of drainage pathways have sometimes been demonstrated to be effective in reducing the volume of contaminated water from underground mines. The goal is to retain the contaminated water in the underground workings and allow the ground water table to rise. This approach also is coupled with the expectation that the local ground water level will cover the underground workings to prevent continued oxidation of the rock. While release of contaminated water through new routes is often observed, further management of these sources can potentially cover the historic workings and reduce the contaminant load in the water considerably. If successful, and assuming the adit plugs and grouting are stable, the costs of treatment can consequently be substantially reduced. The implementability of this technology is highly site specific and requires an understanding of the hydrologic system, as well as the mine workings. While adit plugs can work well under favorable conditions of geology, hydrology, and mine development, such favorable combinations have been found to be rare.

***Interception of ground water:*** In some cases, drainage patterns of surface and ground water can be altered to keep good-quality water away from reactive underground workings, pits, or waste rock dumps to reduce the volume of contaminated water that is produced. Each case requires an extensive study of the hydrologic system and the associated contaminant source. For surface water, simple diversions via channels over areas of infiltration (e.g., faults and slopes) can reduce the amount of contaminated water that is generated. Ground water diversions can potentially involve two general techniques. The first is to establish *passive drainage systems* that take advantage of elevation differences and ground water system opportunities. In this case, water is drained away from reactive rock by drilling water conduits that change hydrologic gradients to limit the amount of water that rinses reactive rock. The second is to establish *in-perpetuity pumping programs* to keep good-quality water from the reactive rock underground workings. In this case, wells are drilled upgradient of reactive rock surfaces to lower the water table to reduce the contaminant load in the surface or ground water. Such proposals have been developed for maintaining dry pits, reducing flows of water from springs that exist under reactive waste rock dumps, and reducing flows that pass through underground workings. These techniques can potentially reduce or eliminate the need for water treatment. While almost always expensive, these types of pumping systems, under certain circumstances, can be less costly than water treatment. However, in establishing programs that require very long-term pumping, it is necessary to recognize that if the pumping is discontinued and ground water flows return to the pre-pumping condition, the contaminants in the underground workings will again be mobilized. Additionally, long-term pumping upgradient of the source area, and resultant dewatering, may increase the release of heavy metals from a negative geochemical effect.

### 4.1.3 Prevention of Acid Drainage via Protective Neutralization

The detrimental effects of sulfide oxidation in mine waste can be offset when the material contains excess acid-neutralizing minerals, such as calcite ($CaCO_3$). Neutralizing minerals react in situ with acidic leachate to neutralize acidity, precipitate most sulfate (as gypsum, $CaSO_4 \cdot 2H_2O$, or other calcium sulfate compounds) and iron (as oxides or sulfates), and reduce dissolved trace metal by inducing adsorption to surfaces. As a result, sulfidic mine waste that contains excess neutralizing potential can, theoretically, weather into perpetuity without releasing acidic water. This section describes the technologies for in-situ source control of net-acid – generating waste using the addition of neutralizing materials and identifies those factors most critical in assessing their feasibility.

In-situ acid neutralization technologies are based on the *acid base accounting* (ABA) of a material. The ABA is the balance between total acid-generating potential (AGP), which is the total amount of acidity that would be produced if all sulfide in a material is completely oxidized, and total acid-neutralizing potential (ANP), which is the amount of acid that could be consumed by neutralizing minerals. There are numerous methods for analyzing for ABA, allowing the flexibility to tailor testing to site conditions and budgets. Unfortunately, there are also several systems of ABA nomenclature in use, with no clear standard emerging. In this document, the convention in which ANP and AGP are converted to $CaCO_3$ equivalents and reported in g $CaCO_3$/kg rock (i.e., parts per thousand, ‰) is used. ABA is described using net-neutralizing potential (NNP), defined as ANP – AGP. Thus, NNP has units of ‰ $CaCO_3$ and is negative for net acid-generating material and positive for net-neutralizing material.

ABA is typically calculated from analysis of sulfide S and carbonate C, assuming a 1:1 molar ratio of sulfide S (AGP) and carbonate C (ANP).

Converting chemical analysis for sulfide S ($S_{(FeS^2)}$) and carbonate C ($C_{(CaCO^3)}$):

$$AGP = S_{(FeS)} \times (10) \times (3.12)$$
$$ANP = C_{(CaCO^3)} \times (10) \times (8.33)$$
$$NNP = ANP - AGP$$

Where

$S_{(FeS)}$ = Concentration sulfide sulfur in sample (weight % S)
3.12 = molecular weight of $CaCO_3$ / molecular weight of sulfur
$C_{(CaCO^3)}$ = Concentration carbonate carbon in sample (weight % C)
8.33 = molecular weight of $CaCO_3$ / molecular weight of carbon

Basic silicate minerals may also contribute to ANP with “silicate neutralization,” consuming acidity in the process of dissolving. Acid neutralization by silicate minerals is typically much slower than reactions with carbonate, and reaction rates depend strongly on pH, particle size, and surface area.

Pore water neutralized in-situ by calcite does not ensure perfect water quality. Neutralization of sulfuric acid by calcite can still leave sulfate concentrations greater than 2,000 mg/L. Under oxidizing conditions, iron will precipitate from neutralized water, forming hydroxide and sulfate minerals that are effective adsorption substrates for trace metals, but trace metals in pore water may remain above remedial goals. However, solute reductions upon neutralization can be dramatic, with 10-fold to 1,000-fold reductions in concentration common. Lime amendment of acid-generating waste when mixing, dispersion, and other attenuation processes are considered can, under favorable conditions, produce mine waste that meets exposure-point water quality standards. However, if lime is consumed prior to exhausting the acid-generating capacity or is inefficiently mixed, contaminated acidic water can begin to drain from these sites long into the future.

Fortunately, studies find that ANP is generally a good indicator of long-term acid release. A review of 281 kinetic tests (various humidity cell and column tests from 53 different mines) found no net-neutralizing samples (i.e., NNP > 0) that produced acidic leachate (Morin et al., 1995). A separate comparison of 307 samples from nine hard-rock mines found similar results—NNP (using carbonate carbon for ANP) was a reliable predictor of actual acid release under simulated weathering conditions (see Figure 4-1). Acid production rates in sulfidic mine waste vary enormously with intrinsic oxidation rates (i.e., oxidation rate under atmospheric conditions), with low rates being $\sim 10^{-8}\ kgO_2/m^3 - s$ and high rates being $\sim 10^{-6}\ kgO_2/m^3 - s$ (Bennett, 1998). These results indicate that when there is an excess of naturally occurring carbonate minerals in mine waste, the neutralizing reactions generally keep pace with the acid production.

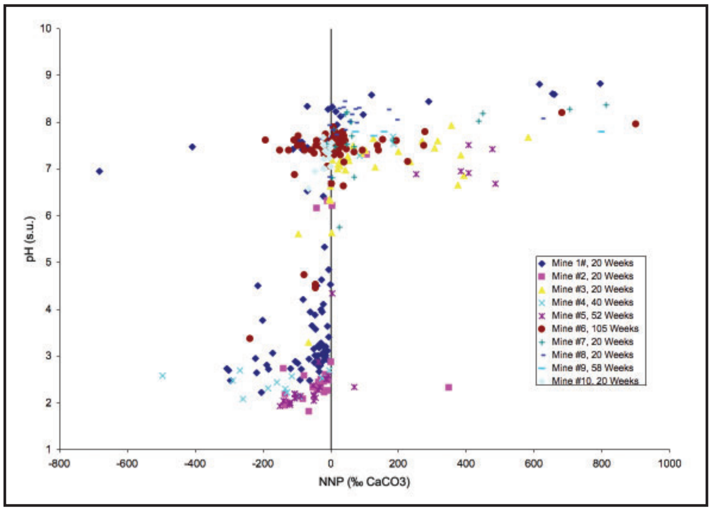

**Figure 4-1.** Humidity cell results from nine hard-rock mines: NNP vs. final humidity cell pH. (Source: Exponent, 2000)

Estimating the acid/base accounting of mine waste has two components: (1) obtaining sufficient sampling to generate a representative sample of the target material and (2) conducting chemical analysis that accurately indicates material ABA.

The number of samples required to adequately define the distribution of acid/base accounting in mine waste depends on the size of the unit targeted for treatment, the variability in the ABA of the material, and the desired accuracy. A 1989 guidance document (SRK, 1989) is one of the few references to recommend a fixed number of ABA samples based on the size of each geologic unit. When large waste rock or tailings facilities are targeted for treatment, geostatistical analysis may be warranted to identify spatial correlations in ABA.

The dominant analytical methods for acid/base accounting in mine waste, in order of increasing complexity, are:

1. **Net acid-generating test** (NAG) (Miller et al., 1997). This is the simplest ABA analysis, reacting a sample with hydrogen peroxide to completely oxidize all sulfide minerals, then noting the pH after reaction as an indicator of whether the material is net-acid generating ($pH < 4.5$) or net acid-neutralizing ($pH > 4.5$). It is rapid and inexpensive, can be conducted in simple field laboratories, and can be modified slightly to yield more quantitative information or excess AGP.
2. **Leco furnace method** (ASTM, 2003). This is a rapid method that requires sophisticated equipment but relatively little labor. Results of this method are generally consistent with comparison tests using long-term kinetic tests.
3. **Sobek titration method** (Sobek et al., 1978). This is the original ABA analytical method, based on titration of samples to determine acid and base concentrations directly. It is labor intensive and thus generally more expensive to conduct than the two methods previously described, but it is generally regarded as the most reliable indicator of long-term acid release potential.

Lime and limestone are the most commonly used amendment materials. Lime—in both the processed ($CaO$) and hydrated [$Ca(OH)_2$] forms—is more soluble and reacts more rapidly than calcite [i.e., limestone ($CaCO_3$)], and is thus considered to be more effective in controlling ARD (Evangelou and Zhang, 1995). However, due to their high solubility, lime amendments can be washed quickly from waste rock, thus limiting their long-term effectiveness in unsaturated conditions, where acid production can continue after the lime is leached out. Thus, an effective waste rock amendment strategy would be to use calcite for the cap, where long-term maintenance is required, and CaO or $Ca(OH)_2$ for subaqueous waste, where oxidation will dramatically slow after emplacement of the waste. While oxygen diffusion is slowed, it is usually not completely eliminated, and in most cases, the rate of contaminant release will depend on this rate of oxygen penetration to the reactive surfaces.

Mine waste amendment is suitable for any materials that can be accessed and subjected to complete mixing with the amendment. In practice, amendment is generally considered for mine waste that (1) can be treated as it is being excavated; (2) is near the surface (< 1 to 2 meters deep), which can be amended by surface application followed by ripping to mix at depth (may be amenable when vegetation of acid-generating waste is considered); and (3) is to be moved for additional purposes.

Excavation of large waste rock and tailings and use of amendment to ensure perpetual in-situ ARD neutralization is often more expensive than other alternatives, including perpetual collection and treatment of acidic seepage.

Field-scale tests indicate that mixing neutralizing minerals with acid-generating waste may need to be nearly ideal to prevent ARD formation (Mehling et al., 1997) and that neutralizing amendments should be 2 mm in diameter or smaller. Field- and large-scale test plots indicate that there may need to be as much as a 100% excess of amendment to ensure perpetual acid neutralization. Amendment rates should exceed those estimated solely on the basis of an ABA (Day, 1994; Cravotta et al., 1990). However, the amount of excess neutralization capacity required to ensure pH-neutral effluent varies from site to site. Mehling et al. (1997) summarized three wide-ranging guidelines taken from successful waste rock blending schemes at historical coal mining sites:

1. NNP > 80 ‰ $CaCO_3$ (Erickson and Hedlin, 1988)
2. NNP > 10 ‰ $CaCO_3$ and ANP > 15 ‰ $CaCO_3$ (Brady et al., 1990)
3. ANP/AGP > 2 (Day, 1994)

Not surprisingly, U.S. regulatory guidelines for classifying waste as non–acid generating also vary widely. Some state guidelines consider waste to be non–acid generating without additional kinetic testing if it has 20 percent excess neutralizing capacity [i.e., a safety factor of 1.2, ANP:AGP ratio ≥ 1.2:1 (NDEP, 1990)]. Bureau of Land Management (BLM) guidelines set this criterion at 300% excess ANP (i.e., a safety factor of 3, ANP:AGP ≥ 3) and also suggest an ANP greater than 20 ‰ $CaCO_3$ (BLM, 1996).

Cost considerations are listed below. These costs are estimates based on current quoted costs in 2005. However, the costs will vary, based on availability of raw materials, energy costs, and other specific requirements at a site.

**Limestone (crushed to < 2 mm, assuming local source):**

- Delivered from off-site source: $US30–50/tonne
- Mined and crushed from on-site source at operating mine: $US2–3/tonne

**Lime, variable, depending on source and haulage:**

- Hydrated lime: $60–140/tonne
- Lime (CaO): $80–$240/tonne

**Safety factor for neutralized waste:**

- ANP greater than 20 ‰ $CaCO_3$ (BLM, 1996)
- ANP/AGP ≥ 1.2 (i.e., 20% excess-neutralizing potential)

**Cost to mix amendments into waste rock (mixing costs only):**

- Complete mixing (Grizzly to separate waste rock and pug mill to mix): $US0.75–1.50/tonne
- Surface mixing by ripping in amendment with bulldozer (maximum depth ~6 ft.): $US0.04–0.06/tonne-treated rock

### *Performance Data*

Laboratory and field-scale studies demonstrate that the effectiveness of mine waste amendment is affected primarily by the mixing efficiency and the particle size of neutralizing materials. Specifically, mixing at less than complete homogenization can allow acid production, followed by migration of acidic leachate in preferential flow paths; neutralizing amendments, particularly limestone, greater than approximately 2 mm in diameter are significantly less effective at neutralizing acidity. Following are a few studies from the literature that illustrate these conclusions. Further, many states have guidance on ABA requirements for mine waste, suggesting that blending programs are generally accepted. However, the failure of several field-scale, neutralization-blending tests is likely to be a cause of concern for the scientific community and possibly for experienced representatives in industry and the regulatory community. Two cases of mixed success are presented in the following table.

| | |
|---|---|
| **Site Name and Location** | *Samatosum Mine in south-central BC (Morin and Hutt, 1996; Mehling et al., 1997)* |
| Experimental Design | Field-scale horizontal layers of acid-generating waste rock; NNP/ANP ratio of 3. |
| Results | The waste rock pile produced acidic leachate despite being amended to obtain a three-fold excess ANP. Hydraulic short-circuiting was cited as the probable cause of failure of a waste rock amendment scheme. |
| **Site Name and Location** | *Kutcho Creek Project, BC (Mehling et al., 1997)* |
| Experimental Design | Field-scale test constructed with 10-cm thick horizontal layers of acid-generating waste rock with net-neutralizing rock to achieve an ANP/AGP ratio of 1.1. Two-year duration. |
| Results | Partial success, with ARD released from net-acid – generating comparison, but not from the net-neutralizing material. However, projections indicate that ARD was likely from amended layers, leading to the conclusion that blending was not effective for preventing ARD in material with an ANP:AGP of only 1.1. Cost analyses of complete blending suggested that this method might be prohibitively expensive on a large scale. |

#### 4.1.4 Passivation of Sulfidic Rock

In recent years, emerging technologies have been examined that are designed to limit the release of acidic components by forming a thin protective layer on the reactive rock surface. This "passivation" has the potential to reduce or eliminate oxidation of the rock and thus reduce or eliminate release of contaminants from the rock. Each of the technologies examined utilizes liquids that can be applied to pit walls or reactive waste rock. The following three technologies are examples that are being investigated:

1. Potassium permanganate: Pyritic surfaces are first rinsed with a solution of lime, sodium hydroxide, and magnesium oxide at a pH > 12, followed by treatment with potassium permanganate. The manganese/iron/magnesium surface formed is resistant to further oxidation and substantially reduces the amount of acidic water draining from the treated rock.
2. Ecobond™: MT$^2$ (http://www.metalstt.com) uses a phosphate-based solution to coat acid-generating rock to form a stable, insoluble coating on the surface. The technology forms stable iron phosphate complexes that resist hydrolysis and prevent further oxidation.
3. Silica Micro Encapsulation (SME): KEECO has a patented process that treats acid-generating rock with a solution of silica. It encapsulates metals in an impervious microscopic silica matrix and prevents additional acid generation or metals migration. This technology has been utilized previously for metals-contaminated soils.

These technologies are being tested at several sites, but their efficacy has not been thoroughly established. The EPA/DOE (Department of Energy) Mine Waste Technology Program has examined these technologies (http://www.epa.gov/minewastetechnology/). The future utilization of these methods for waste rock requires consideration of the following:

- **Hydrologic characteristics of the waste rock:** For pre-existing waste rock dumps, how can complete (or near-complete) coverage of the reactive surfaces be ensured? Is it sufficient to treat the top of the waste rock dump (1–5 meters), or does the entire waste rock dump require treatment?
- **Longevity:** Each type of coating is thin, and the reactive bulk rock remains reactive. How long will these coatings prevent release of contaminants?
- **Cost:** What are the site characteristics that will change the cost of these treatments? How do these costs compare to conventional water treatment (e.g., lime precipitation)?

Table 4-1 on pages 12 and 13 provides an overview of the source control technologies covered in this section, technology selection factors, and limitations.

### *4.2 Treatment of Contaminated Water*

As discussed previously, waters draining from mine sites can vary dramatically, and the methods used to treat those waters will similarly vary. For this discussion, mine waters can be put in three groups, although the distinction between the groups is sometimes not clear.

- **Acidic water (pH < 5.5):** Most commonly, these waters are contaminated as a result of pyrite oxidation and contain elevated metals and sulfate. While any water with a pH < 7 is acidic, the most problematic waters are those with a pH < 4 since metals loadings increase substantially. Because of the problems with acidic waters, extensive research has been conducted on cost-effective methods of treatment. Chemical neutralization consumption and sludge management are large factors for selecting a treatment method.

*(Continued on page 14)*

**Table 4-1.** Source Control Technologies for Hard-Rock Mine Waste

| Technology Name | Technology Description | Target Analytes | Critical Feasibility Factors | | | Important Limitations |
|---|---|---|---|---|---|---|
| | | | Implementabilty | Effectiveness | Cost | |
| **Capping and Revegetation** | Cover waste rock or tailings with suitable growth medium and establish vegetation. ("Suitable" means soil, waste rock, tailings, or blend of these materials that is non-acid generating and contains heavy metals below levels that are phytotoxic or that may cause ecological or human health risk.) A "Store and Release Cap" retains meteoric water long enough for plants to uptake and transpire the water, minimizing the "net percolation" (i.e., the flux of water from the cap into the waste) and associated rate that solutes in waste are flushed out. (No specific chemical reactions.) | All soluble constituents in the solid waste | • Stability of waste or cap to slope failure and erosion<br>• Availability of suitable cap material (i.e., non-acid generating, low metals, high moisture retention)<br>• Access to slopes for seeding and grading<br>• Commonly performed, with often good but variable results | • Moisture retention capacity of cap material<br>• Cap thickness<br>• Potential for metals uptake from cap by plants (affects ecological risk)<br>• Fire frequency<br>• Fraction of precipitation as snow | • Availability of local cap material, either topsoil, benign waste rock or tailings, or mixture of these<br>• Need to possibly amend cap with neutralizing agent or nutrients<br>• Cap thickness (at least 1 m in semi-arid climate to sustain plants; may be thinner in wetter climates)<br>• Life expectancy of cap | • A zero-net-percolation cap has not been demonstrated<br>• Precipitation as snow greatly increases net percolation<br>• In acid-generating waste, reduced flow delays impact, but may not reduce contaminant load rate<br>• Highly engineered caps (e.g., liners, capillary breaks) have finite life<br>• Oxygen barrier to permanently stop ARD production is theoretically possible, but has not been demonstrated for long-term applications |
| **Wet Covers** | Storage of acid-generating rock and tailings under water to prevent (or minimize) oxidation and release of contaminants | All | Can be utilized when reactive rock can be submerged in a permanent water body, including a permanent tailings facility pit lake or flooding underground workings | Will generally reduce oxidation rates of reactive rock | Highly variable, depending on the availability of water and site-specific considerations | • Requires in-perpetuity coverage of the reactive rock and management of the water body<br>• Previously oxidized rock surfaces may still release problematic contaminants |
| **Hydrologic Controls** | Drainage is controlled by directing water flow from reactive rock surfaces, by diverting ground water flows, or by pumping ground water. A second option is to plug adits and shafts to allow water to fill underground workings and cover reactive surfaces. Water and/or oxygen reaction with surfaces is minimized. Rerouting surface water around surface or underground disturbances is also sometimes an option. | All | • Requires an understanding of the hydrologic system surrounding the reactive rock<br>• Not commonly undertaken due to difficulties in obtaining a full understanding of the hydrologic system | • Elimination of water from the reactive surfaces can effectively stop acidic drainage<br>• Effectiveness is dependent on the ability to control water in the underground and surface water systems | • Highly variable depending on site-specific factors<br>• Long-term pumping has O/M costs, while an effective drainage system can be highly cost effective | • Effectiveness depends on the ability to control water, either by plugging or draining the adits<br>• This method, while it always should be considered, is not often successful<br>• Plugging of underground workings has resulted in blowouts of the plugs in certain cases |

**Table 4-1.** Source Control Technologies for Hard-Rock Mine Waste *(continued)*

| Technology Name | Technology Description | Target Analytes | Critical Feasibility Factors | | | Important Limitations |
|---|---|---|---|---|---|---|
| | | | Implementabilty | Effectiveness | Cost | |
| **Acid Potential Neutralization** | Waste rock or tailings that are net acid-generating are amended with neutralizing agents (e.g., crushed calcite [$CaCO^3$] or lime [CaO], alkaline industrial wastes), producing waste that will remain permanently net neutralizing. Amendments can be applied surficially to existing waste and mixed (~0.5 – 2 m depth–typical root-zone depth for cap only treatment) or added to new waste and mixed during emplacement. | Heavy metal cations (e.g., Cu, Cd, Pb, Zn), acidity, sulfate (pore water reduced to ~1,500 - 2,000 mg/L) | • Access to all acid-generating waste rock targeted for blending in neutralizing agents<br>• Stability of slope during blending<br>• For cap amendment, sufficient access and slope to permit distribution and mixing of neutralizing amendment into near-surface (~2 m depth) | • Ability to uniformly blend neutralizing agents with acid-generating waste<br>• Crushing neutralizing agents small enough to ensure reaction (e.g., 2 mm) | • Availability of local source of limestone or other neutralizing agent<br>• Cost to mine, crush, and deliver neutralizing agent (ideally 2 mm diameter for blending)<br>• Cost to spread and mix neutralizing agent into cap<br>• For existing waste, cost to excavate and uniformly amend | • Surficial amendment of acid-generating waste does not stop ARD production below the cap<br>• Typically, it is not economic to excavate and amend existing buried waste<br>• Layering or sequential placement of neutralizing waste with acid-generating waste often does not stop acid release |
| **Passivation** | Reduces or halts the oxidation of reactive surfaces, primarily pyrite. Application methods vary, but usually involve coating rock surface with fluids and allowing the specific passivation reaction to occur. These technologies are still in a research and demonstration mode. | Acidity and soluble constituents in the solid waste | The effectiveness of passivation remains to be established. Laboratory and pilot-scale treatments show promise for the various treatments, but full-scale applications have not been undertaken. | Unknown. The costs need to be evaluated against the probability of long-term treatment using more conventional methods. | Costs are highly variable, depending on the technology utilized and the need for periodic treatment | Full passivation of waste rock dumps is difficult, due to the problems with delivering fluids in such a way that all surfaces are contacted. Also unknown is the longevity of these treatments. Requires further investigation. |

- **Near-neutral water (pH 5.5–9):** These waters are common at many non-acid – generating sites, particularly those with high net neutralization in the waste rock. Sulfate concentrations are generally less than 2,000 mg/L, but may contain elevated concentrations of certain metals (e.g., zinc, copper, or nickel), oxyanions, or arsenic, antimony and selenium, particularly at the higher pH ranges. Common examples are drainage from carbonate-hosted waste rock dumps, closed precious metals heaps, and pit lakes.
- **Alkaline water (pH > 9):** With few exceptions, these are commonly associated with process fluids, and the elevated pH is due to chemical reagent addition (e.g., sodium cyanide plus lime). The solubility of a variety of oxyanions can be enhanced at alkaline pH. Over time, the pH of these waters is reduced when atmospheric carbon dioxide dissolves.

### 4.2.1 Treatment of Acidic Waters

Acidic water is generally considered the most problematic mine-related drainage water, and it offers the greatest potential for degradation of surface and ground water. While prevention of acid drainage is a common goal for management of acid-generating rock, treatment of acidic drainage at many mine sites will be required far into the future.

Methods for treatment of acidic drainage vary considerably, but most focus on increasing the pH to above pH 7, which will subsequently reduce the solubility of a variety of contaminants in the drainage water. This is especially true for the divalent metals and aluminum, which are precipitated as hydroxides. The literature on treatment of drainage from coal mines has examined the various types of neutralizing agents in detail, and although differences exist between coal and hard-rock mine waters, treatment of coal mine waters has been extensively examined in the past 25 years (http://www.wvu.edu/~agexten/landrec/chemtrt.htm). Five chemicals that have commonly been used for treatment of acidic water are listed in Table 4-2.

Ammonia has also been utilized for treatment of coal mine waters, but is uncommon for treatment of hard-rock mine waters and will not be considered further here. This leaves two general types of neutralization agents, the calcium- and sodium-based systems. Of these, the calcium-based systems are generally preferable to sodium due to the ability of calcium to remove sulfate as calcium sulfate compounds (e.g., gypsum). Calcium will also ultimately precipitate as calcite when the water is equilibrated with carbon dioxide in air if the pH is slightly elevated. Alternatively, while sodium-based neutralization agents are effective in raising the pH, elevated sodium in irrigation water causes soil structure to collapse (sodic soils). Also, particularly when handled in bulk, lime is generally less expensive than either sodium carbonate or sodium hydroxide.

**Table 4-2.** Neutralizing Reagents for Treatment of Acidic Water from Mines (Source: Modified from Skousen in http://www.wvu.edu/~agexten/landrec/chemtrt.htm.)

| Common Name | Chemical Name | Formula | Conversion Factor* | Comments |
|---|---|---|---|---|
| Limestone | Calcium carbonate | $CaCO_3$ | 1.0 | Inexpensive chemical cost, but difficult to dissolve—tends to armor and reduce effectiveness. Utilization of only 30% of neutralizing capability. |
| Hydrated lime | Calcium hydroxide | $Ca(OH)_2$ | 0.74 | Relatively inexpensive chemical cost and most utilized form of lime as a slurry. Requires control to maintain suspension. Neutralizing efficiency of 90%. |
| Lime (quicklime) | Calcium oxide | $CaO$ | 0.56 | Also commonly utilized, although more effort is required to maintain a suspension. Requires slaker to convert to hydrated lime. Neutralizing efficiency of 90%. |
| Soda ash | Sodium carbonate | $Na_2CO_3$ | 1.06 | Dissolves rapidly; less caustic alternative to sodium hydroxide. Does not remove sulfate effectively. Increases sodium content of treated water. Neutralizing efficiency of 60%. |
| Caustic soda | Sodium hydroxide | $NaOH$ | 0.8 | Does not remove sulfate effectively and increases sodium content of treated water. Neutralizing efficiency of 100%. |

* The conversion factor is the relative amount of weight of each material (compared to limestone) to neutralize a given amount of acid. The estimated tons of acid/year can be multiplied by the conversion factor to get the tons of chemical needed for neutralization.

#### *4.2.1.1 Conventional Physical/Chemical Treatment of Acidic Water Using Lime*

Use of lime (calcium hydroxide or calcium oxide) for neutralization is the accepted conventional water treatment for most hard-rock mine acidic waters, particularly when the acidity and/or flows are high. Not only does this treatment raise the pH in a cost-effective manner, it also reduces the sulfate concentrations to below 2,000 mg/L due to the relatively low solubility of gypsum (calcium sulfate). (See INAP, 2003 for summary of methods for treating sulfate in water.) For this discussion, "conventional water treatment" refers to fixed facilities of pipes, metering pumps, reaction vessels, clarifiers, and solid management mechanical fixtures (e.g., filter presses). Lime is mixed with acidic water as a 10–15% slurry that is most commonly generated on site from a storage tank of hydrated lime using a slurry mixer.

Conventional acidic water treatment using alkaline sources is used for removal of Al, Fe, Cu, Cd, Pb, Zn, Ni, and Mn as metal hydroxides. The dissolved concentrations of the oxyanions, including Cr, Se, Sb, Mo, As, and U, can also be substantially reduced by the co-precipitation with the metal hydroxide.

Oxidation of soluble ferrous iron to ferric iron is required for treatment of most mineral acidic waters and utilizes atmospheric oxygen at an elevated pH > 7. Ferric oxide rapidly precipitates at neutral or alkaline pH. Mixing of the acidic water with lime slurries requires aeration to ensure good contact with atmospheric oxygen. Most conventional treatment systems utilize a stirred aeration basin to accomplish this oxidation (see Figure 4-2).

A relatively new method for oxidation uses the Rotating Cylinder Treatment System (RCTS) to provide rapid oxygen transfer to the solution and efficient utilization of the lime slurry (http://www.iwtechnologies.com). The RCTS uses shallow trough-like cells to contain the impacted waters and rotating perforated cylinders for improved atmospheric oxygen transfer and improved agitation during treatment of the water.

Following oxidation and neutralization, agitation of the suspension and addition of flocculants allows the metal oxide solids to settle out by growth of precipitants to sufficiently large particles to form sludge. The sludge produced is generally of low density and requires thickening or filter presses to decrease the water content. Additionally, management of the sludge generally requires a de-

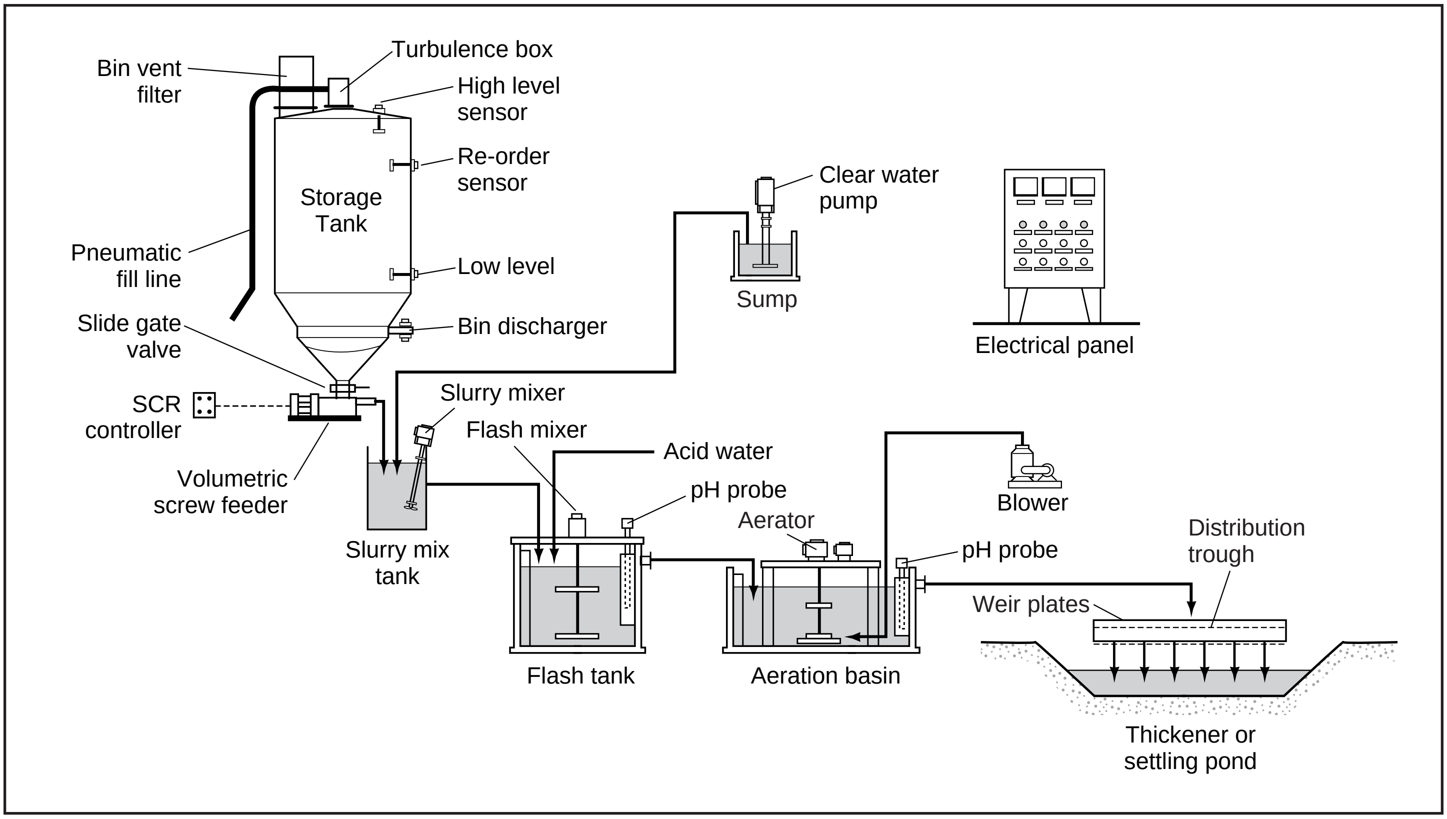

**Figure 4-2.** Conventional water treatment utilizing lime. (Source: http://amd.osmre.gov/Cost.pdf)

termination of its contaminant leachability to decide whether the sludge can be managed on-site or needs to be transported to an off-site hazardous waste management facility. The Toxicity Characteristic Leaching Procedure (TCLP) is usually used to determine if a material is Resource Conservation and Recovery Act (RCRA) – hazardous due to its leaching characteristics. The Synthetic Precipitation Leaching Procedure (SPLP) is a better indicator of leaching behavior under natural environmental conditions. State regulations also apply to how sludge is managed, on site or off site.

Because of the wide range of flows, the amount of lime required, the length of time for each type of reaction, the method of settling or filtering out the solids from the water, and the method for residual (sludge) management, site-specific information is required for the design of each system. Shakedown operations and modifications of original design are often performed to meet target discharge requirements and to optimize operations to reduce costs or volume of residual (sludge) produced.

The costs for construction of active lime treatment facilities can be substantial due to the requirements of power, pumps, lime addition systems, tanks, and sludge management equipment. Several organizations have developed guidelines. An example of such guidelines is one developed by the Office of Surface Mining (2000) (http://amd.osmre.gov/Cost.pdf). While this document is focused on costs for treatment of acidic drainage from coal mines, the same approach can be used for estimating costs for treatment of acidic drainage from hard-rock mines. Because the characteristics of water quality, flows, remoteness, and reagent costs, as well as other factors, can vary substantially, it is difficult to provide a reliable estimate for treatment at a specific site until a careful engineering estimate is developed. However, estimates for treatment cost vary from less than $1/1,000 gallons to well over $10/1,000 gallons on an annual operating and maintenance basis.

While the newer designs for lime treatment systems are increasingly automated, these systems still require frequent monitoring and oversight due to the caking and scaling problems common with the use of lime. Additionally, these active systems utilize pumps and mixing systems that require routine maintenance. Thus, operation of a lime treatment plant has inherent fixed construction and operation/maintenance costs that make these treatment systems expensive on a cost per volume of water treated *when the flows are low*. However, as flows increase (e.g., > 100 gal/min) or the acidity and metals loadings increase, the fixed costs become a smaller fraction of the total cost, and lime treatment is generally the most cost-effective method for treating large volumes of acidic drainage from mines. Comparison of costs of treatment at different flows (using reagent costs in 1996) is available at http://www.wvu.edu/~agexten/landrec/chemtrt.htm#Chemical. While the reagent costs change with time and location, as well as the implementation of a treatment system at a specific location, this example provides an indication of the non-linear cost differences with differing flows. Each treatment alternative needs to be evaluated relative to the total costs and intended characteristic of the effluent water.

Using an engineered system of conventional water treatment requires proper road access, a power supply, stable land area, and manpower. In remote areas of the western U.S., access may be difficult and expensive during the winter months, and a conventional lime treatment system may not be appropriate.

### *Key Web Site References*

- Overview of chemicals available to treat AMD: http://www.leo.lehigh.edu/envirosci/enviroissue/amd/links/chem1.html
- AMD abatement cost-estimating tool developed cooperatively by the Pennsylvania Department of Environmental Protection, the West Virginia Department of Environmental Protection, and the Office of Surface Mining (OSM) Reclamation and Enforcement: http://amd.osmre.gov/
- The MEND manual is a set of comprehensive working references for the sampling and analyses, prediction, prevention, control, treatment, and monitoring of acidic drainage. The document provides information on chemistry, engineering, economics, case studies, and scientific data. http://www.nrcan.gc.ca/mms/canmet-mtb/mmsl-lmsm/mend/mendmanual-e.htm
- UK summary of active and passive treatment: http://www.parliament.uk/commons/lib/research/rp99/rp99-010.pdf
- Britannia Mine Water Treatment Plant Feasibility Study. An example of a feasibility study for a site-specific conventional system: http://www.agf.gov.bc.ca/clad/britannia/downloads/reports/tech_reports/WTP_feasibility.pdf.

- Example of evaluating options for sludge management for a conventional water treatment system: http://www.agf.gov.bc.ca/clad/britannia/reports.html
- Detecting change in water quality from implementation of limestone treatment systems in a coal-mined watershed of Pennsylvania: http://www.mbcomp.com/swatara/Cravotta.pdf
- Abandoned mine remediation clearinghouse for treatment of acidic drainage in Pennsylvania: http://www.amrclearinghouse.org/Sub/AMDtreatment/ZZTreatmentStrategies.htm
- National Lime Association Web site. A wealth of information about neutralizing acidic water with lime: http://www.lime.org
- Army Corps of Engineers document "Engineering and Design: Precipitation/Coagulation/Flocculation": http://www.usace.army.mil/inet/usace-docs/eng-manuals/em1110-1-4012/toc.htm

#### *4.2.1.2 Physical/Chemical Treatment in Alkaline Ponds and Lagoons*

Physical/chemical treatment in alkaline ponds and lagoons is very similar to conventional treatment as described in the preceding section. Using ponds and lagoons for aeration, settling, and solids accumulation has the benefits of exploiting natural processes. Lime (calcium hydroxide) is added using the same type of equipment that is used in conventional plants.

Physical/chemical treatment in alkaline ponds and lagoons is used to remove metals, including Al, Fe, Cu, Cd, Pb, Zn, Ni, Mn, and the oxyanions Cr, Se, Sb, Mo, As, and U. Depending on the water chemistry, the oxyanions are reduced in dissolved concentrations by co-precipitation with metal oxides and calcite. Formation of metal hydroxide precipitates and formation of calcium carbonate with flux from the atmosphere result in solids settling out in the ponds. While the primary neutralization of the acidic drainage is through lime addition, the ponds and lagoons can improve the overall water treatment and metals reduction by the photosynthetic activity in the water (see Figure 4-3).

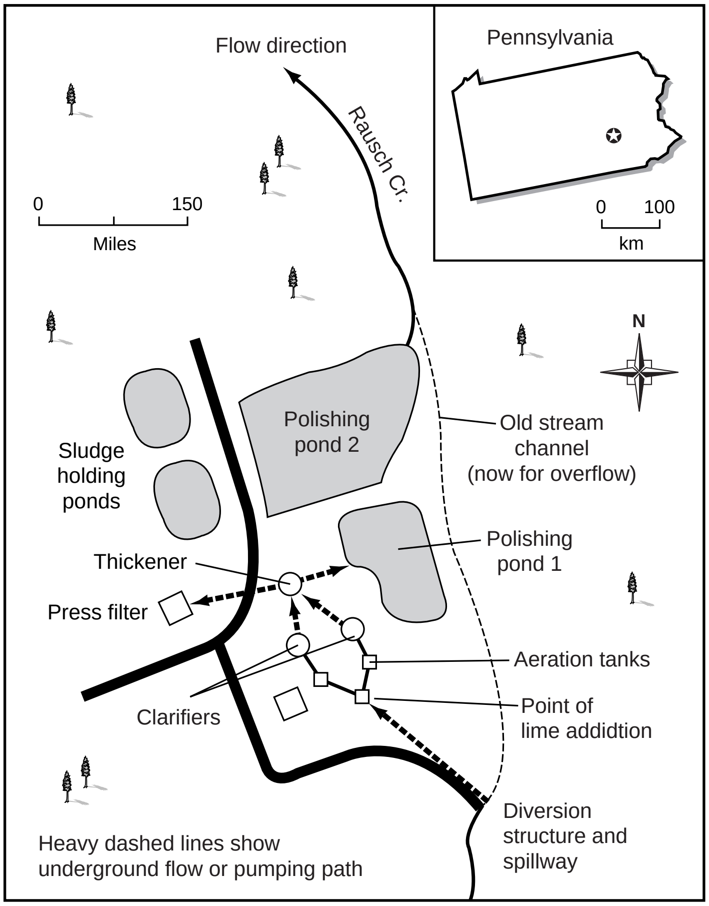

**Figure 4-3.** Use of aeration ponds for polishing lime treatment process. (Source: http://www.facstaff.bucknell.edu/kirby/RCr.html)

A larger area of land is needed for physical/chemical treatment in alkaline ponds and lagoons than for conventional treatment plants. Pond or lagoon treatment systems are often easier to construct if existing settling ponds, tailing ponds, or excavated areas are available for use. Site-specific information is critical for design of these systems because of the wide range of flows, the amount of lime required, the length of time for each type of reaction, the method of settling or filtering out the solids from the water, and the method for residual (sludge) management. Alkaline ponds and lagoons have been used to effectively remove metals, metalloids, and uranium from mine waters when designed to account for variations in flow and composition. Removals at a treatment lagoon in Butte, Montana, are presented in Table 4-3, and a photograph of the polishing pond is shown in Figure 4-4 on the next page. Shakedown operations and modifications of original design are often performed to meet target discharge concentrations and to optimize operations to reduce costs or the volume of residual (sludge) produced.

The construction cost for physical/chemical treatment in alkaline ponds and lagoons is usually similar to the additional chemical components of a conventional treatment. Due to land availability and status of land relative to the hydraulic profile of proposed system, site-specific factors can make ponds or lagoons less expensive than clarifiers and reaction vessels. Due to the relative larger size of ponds and lagoons than most conventional treatments, fewer operator hours are required to account for system variations and to physically manage solids produced. Smaller systems can be designed with cleanout and sludge management at frequencies of a few years to decades. An additional advantage of using alkaline ponds and lagoons is the buffering capacity of the lagoons, which corrects minor process upsets or variations.

**Table 4-3.** Influent and Effluent Concentrations for Treatment Lagoon in Butte, Montana, for Year 2003

| Basis | Analyte | Untreated Concentration (ppb) | Treated Concentration (ppb) | % Removed |
|---|---|---|---|---|
| Total | Ag | 5 | 5 | Detection Limit |
| Total | Al | 155 | 34 | 78% |
| Total | As | 35 | 7 | 80% |
| Total | Cd | 15 | 0.3 | 98% |
| Total | Cr | 11 | 10 | Detection Limit |
| Total | Cu | 388 | 15 | 96% |
| Total | Fe | 1,499 | 41 | 97% |
| Total | Mn | 2,478 | 72 | 97% |
| Total | Pb | 6 | 1 | Detection Limit |
| Total | Zn | 4,526 | 107 | 98% |

Using an engineered system of physical/chemical water treatment in ponds and lagoons requires proper road access, a power supply, stable land area, and manpower. One additional advantage of a lagoon system is the attractiveness and wildlife attributes of a wetland, although these features need to be weighed against metals bioavailability and insect breeding issues.

**Figure 4-4.** Final pond in the treatment lagoons in Butte, Montana, used for polishing and robustness of system.

***Key Web Site References and Pictures***

- Silver Bow Creek/Warm Springs Ponds One-Page Summary: http://www.epa.gov/superfund/programs/recycle/success/1-pagers/bowcrk.htm
- Pictures of a lime lagoon at Leviathan Mine, California, which has no biological component due to limited size of the pilot project. Filter bags are used to capture and manage the majority of the sludge, while a lined pond is used for settling and polishing: http://yosemite.epa.gov/r9/sfund/sphotos.nsf/0/75c4f97d7640242488256e98006656ab/$FILE/Leviathan_04%20p7-22.pdf

#### 4.2.1.3 Low-Flow/Low-Acidity Chemical Treatment Options

While conventional lime treatment has distinct cost and treatment advantages, the costs of treating lower flows on a per-gallon basis can potentially be reduced using alternative neutralization methods in certain cases. Examples include the following:

**Automatic lime addition using an Aquafix system:** As discussed previously, addition of lime to acidic water in a controlled and efficient manner requires lime addition technology that increases the fixed costs and is often infeasible for small streams. Jenkins and Skousen (1993) have shown the utility of an Aquafix pebble quicklime (CaO) water treatment system that utilizes a water wheel concept for coal mine drainage waters. The concept is that these systems can be operated without intensive management, and the rate of addition of lime can be controlled by the flow rate of the acidic stream. For this system, the amount of chemical utilized is controlled by a water wheel attached to a screw feeder that dispenses lime directly into the flowing acidic drainage. This system was initially developed for small flows from coal mines of high acidity because calcium oxide is very reactive. Recently, however, water wheels have been attached to large bins or silos for high-flow/high-acidity situations. These systems have received only limited applicability at hard-rock mine sites in the western U.S., although additional testing is warranted. Controlling the rate of application of the quicklime without operator attendance and problems with remote cold weather operation have somewhat limited the interest for many mineral mine sites. These systems also may require settling basins and sludge management for the metals-laden precipitates (http://www.wvu.edu/~agexten/landrec/chemtrt.htm).

**Open limestone channels:** While limestone beds/channels have been used with some success in neutralizing mildly contaminated coal mine acidic drainage, the rate of release of alkalinity is difficult to control, and the limestone tends to armor with aluminum and iron oxide coatings. Open limestone channels are constructed simply by laying limestone rock in a channel and allowing the acidic solution to pass over the rock or by laying limestone directly in a channel of acidic drainage (Ziemkiewicz et al., 1997: http://www.dep.state.pa.us/dep/deputate/minres/bamr/amd/science_of_amd.htm). Because of the armoring that occurs, this method has shown best treatment when the channel is sloped to allow rapid movement of the water and scouring of the coatings on the limestone. However, in mineral-mining applications, limestone channels have not been shown to be successful, and the applicability may be limited to iron-free, aluminum-free waters that only contain metals that can be removed by chemical precipitation at $\text{pH} < 7$. Depending on the requirements, these systems can be lined or unlined. Settling basins may be used under certain conditions to collect precipitates. Limestone treatment is generally not effective for acidities exceeding 50 mg/L (http://www.osmre.gov/amdtcst.htm). A somewhat more effective method for limestone treatment utilizes pulsed, fluidized bed reactors in which acidic water is injected in an upward manner at high velocity into limestone columns. This method can improve the scouring of the limestone and increase the release of alkalinity. Carbon dioxide (either from tanks or by utilization of $CO_2$ released from the limestone) aids the process by reducing the rate of iron oxidation in the reactors (http://www2.nature.nps.gov/pubs/yir/yir2000/pages/07_new_horizons/07_02_reeder.html).

**Anoxic limestone drains:** An anoxic limestone drain (ALD) is similar to an open limestone channel, except the limestone is buried under a cap and designed to exclude oxygen and reduce the amount of iron oxidation products that coat the limestone. This will tend to improve the release of alkalinity from the limestone. These systems have been used to decrease the acidity of drainage waters prior to aerobic wetlands or sulfate-reducing bioreactors (SRBs). The downside to using these systems is that if the limestone becomes armored, uncovering the limestone requires excavation of the cap. Because acidic drainage from hard-rock sites often contains appreciable amounts of aluminum that coat the limestone, these systems are not commonly utilized. Limestone dissolution can increase the pH sufficiently to precipitate oxidized iron and aluminum, but does not effectively remove most heavy metals. Limestone has the most potential as a pretreatment method for passive microbial-based systems where a decreased dissolution rate from armoring can be incorporated into the design.

**Sodium hydroxide:** Addition of solutions of 25% sodium hydroxide to acidic water can be accomplished by either gravity flow or small solar-powered pumps. This system can be very inexpensive to construct, depending on the site conditions, although the cost of sodium hydroxide is higher than a similar amount of calcium-based neutralization agents. The sodium hydroxide solution is completely utilized and an effective neutralization agent. Because 50% solutions will solidify under cold conditions, a 25% solution is generally utilized and is available in bulk solutions. However, the volumes that one will need to use will require either frequent refilling or large storage capacity. For example, a flow of 100 mL/min will utilize 52,500 liters per year (~13,900 gallons) of solution. Two other disadvantages are the safety issues associated with using sodium hydroxide, as well as the increase in sodium concentrations that remain in the treated water.

**Sodium carbonate:** A less caustic alternative to sodium hydroxide is the use of sodium carbonate ($Na_2CO_3$). Sodium carbonate briquettes are available and can be utilized by simply diverting a small stream of the acidic water through (or over) a bed of the briquettes and allowing that solution to mix with the acidic water. Control of the diversion can be managed with a weiring system. Sodium carbonate tends to cement together and change the amount of surface available for dissolution. Temperature changes can also affect the amount of delivered alkalinity. Sodium increases in the treated water are also an issue.

### 4.2.2 Treatment of Neutral and Alkaline Waters

Contaminated neutral or alkaline mine drainage waters are present at sites that have sufficient neutralization (generally from calcite) in the rock such that any acid production is offset by the neutralization available. These acidic waters are also commonly generated from precious metals ore processing using cyanide for mill circuits or heap leach processing.

Neutral and alkaline drainage from mine sites is generally less of a water quality problem than acidic drainage since the solubility of many of the problematic metals is low at neutral or alkaline pH. Neutral and alkaline drainage contaminants generally are most problematic for the oxyanions of selenium, arsenic, and antimony since the solubility of these constituents increases with higher pH. In addition, nitrate, sulfate, and other salts, as well as cyanide species, may be elevated at cyanidization facilities and exceed discharge requirements.

### 4.2.2.1 Arsenic and Antimony

Total arsenic concentrations in drainage water can vary from less than 10 µg/L to several mg/L. Antimony is generally found at lower concentrations. Because these elements are closely related (group 5A in the Periodic Table), treatments for removal are similar and will be considered together. In general, methods to remove arsenic from water also are effective for antimony.

Arsenic treatment technologies have received the greatest focus in recent years, primarily due to the need for arsenic removal in drinking water. An extensive recent EPA arsenic treatment review (EPA, 2002: http://cluin.org/contaminantfocus/default.focus/sec/arsenic/cat/Treatment_Technologies/) and the U.S. Geological Survey Web site (http://arsenic.cr.usgs.gov/) provide a more detailed discussion of the treatment options than is provided here. The most common arsenic treatment systems are briefly discussed below.

For specific application to mine-related waters, arsenic removal from large volumes of water (e.g., pit lakes, discharge water from pit dewatering) most often utilizes iron precipitation/co-precipitation methods. For these systems, ferrous or ferric salts are added to the water and allowed to precipitate. Arsenic, particularly in the +5 valence state, sorbs strongly to the surface of the precipitates and is effectively removed from the water. When arsenic in the +3 valence state is present in appreciable concentrations, a pre-oxidation step may be required since it sorbs less strongly to iron oxides than in the +5 state.

A recently developed method for arsenic treatment utilizes zero-valent iron (Su and Puls, 2001; Nikolaos et al., 2003). For this technology, arsenic-containing waters are passed over iron filings that generally have been mixed into sand at a ratio of 10–20% iron. Iron oxidizes to iron oxide, and arsenic is sorbed to the iron oxide surface. Although the iron is ultimately mobilized (albeit slowly) and the treatment system will need to be replaced, the arsenic that is sorbed is generally not available. The iron oxide/arsenic residue is generally not hazardous, although its classification is dependent on the results of site-specific waste characterization testing. Depending on the design, these systems can remain effective for an extended period of time (months to years). Zero-valent iron systems have also been effectively applied as permeable reactive barriers (PRBs) in subsurface systems for remediation of arsenic-containing ground water at a mill tailings site (EPA, 2000). PRBs are described in Section 4.2.3.2. Arsenic can also (at least partially) be removed from mine waters by sulfate-reducing bacterial systems as described below.

### 4.2.2.2 Heap Effluent

The use of heap leach technology for recovery of precious metals has evolved over the past 25 years and is commonly employed for low-grade ore (typically 0.015–0.06 ounces per ton-equivalent of gold) at many sites throughout the world. The tonnage of ore processed in this manner in Nevada, for example, is estimated to be on the order of 2 billion tons. In this process, ore is placed on high-density polyethylene sheets and rinsed with dilute concentrations of sodium cyanide. In arid regions of the world, these systems are operated in a zero-discharge mode: the amount of water evaporating is greater than the rainfall, and additional water is required to make up the difference of the amount lost to evaporation and the amount of rainfall. When precious metals recovery is completed, the process for closure of the heaps begins.

For arid sites, the most common method for initial reduction of water volume is to continue to recirculate the water to the heap using enhanced evaporation methods: water is sprayed into the air over the heaps and allowed to evaporate, subsequently increasing the concentration of soluble constituents in the remaining water. The rate of water that is recirculated will decrease over time from operational flows of several hundred to several thousand gallons/minute to residual flows that decrease to 0–50 gallons/minute. During this time, carbon dioxide dissolves in the water and reduces the pH to between 8 and 9. This process also allows volatilization/oxidation of cyanide and also enhances the activity of microorganisms that can convert the nitrogen in a variety of cyanide species to nitrate. Because mercury is mobilized as a mercury–cyanide complex, removal of the cyanide is also reasonably effective in reducing mercury concentrations in the drainage water. The amount of water that drains from heaps will vary depending on the site conditions, but it will also depend on the amount of meteoric water, the type of cap (if any) that is placed on the top of the heap, and other site-specific conditions that may be present. However, low-flow drainage from heaps has been observed at most sites and will continue for the foreseeable future. Discharge from heaps can be reliably eliminated only for sites in a highly arid region or those that have a very efficient store-and-release cap.

In higher-rainfall regions, where rainfall on the heap exceeds the amount of water that evaporates, treatment and discharge of excess water is required. Although many of the constituents in these fluids are the same as during closure, cyanide removal becomes more important and requires specialized treatment.

The constituents present in residual cyanidization fluids differ substantially from acid drainage sites. Drainage from three distinct closed heaps is described in Table 4-4. These waters contain elements that have enhanced solubility at higher pH, as well as residual cyanide components. The constituents that are of particular concern include arsenic, antimony, selenium, nitrate, sodium, sulfate, cyanide species [both weak acid-dissociable (WAD) cyanide as well as total cyanide], mercury, and nickel.

Effective treatment of heap effluent requires consideration of all of the constituents present in the drainage water (Table 4-4). While specific treatment methods are available for several of these constituents, or even groups of constituents, relatively few methods are available that can remove all of these to surface water discharge requirements.

Reduction in the volume of water by recirculation and evaporation on the heaps is generally utilized. However, the collection pond water volume is usually large, and treatment is often required for the several millions of gallons typically left after recirculation of the water to the heaps is discontinued. Since the volume is contained in a pond, this water can often be treated in a single batch mode and can utilize intensive techniques (e.g., membrane separation, ion exchange, or aggressive evaporation).

Because the water quality from these heaps is unlikely to change substantially for years to decades due to the slow migration of meteoric water through the heaps, any treatment process will need to be either continuous or allow accumulation of water for periodic batch treatments. Thus, the more intensive management techniques become very costly on a per-gallon-treated basis, and passive methods for water management (1–20 gal/min) are favored. However, few options are available, particularly for saline waters.

Current methods for residual heap drainage water treatment include the following:

**Table 4-4.** Heap Drainage Chemistry Profiles of Three Closed Heaps (Source: NDEP, 2004)

| | Heap 1 Effluent 6/23/98 | Heap 2 Effluent 4Q 95 | Heap 3 Effluent 5/02 |
|---|---|---|---|
| pH | 7.79 | 8.17 | 9.6 |
| TDS | 3,032 | 11,200 | 5,670 |
| nitrate | 54 | 171 | 96 |
| sodium | 340 | 3,880 | 1,640 |
| chloride | 160 | 1,130 | 3,200 |
| WAD CN | 3 | 0.11 | 14.3 |
| sulfate | 1,600 | 6,130 | 470 |
| antimony | 0.023 | – | < 0.003 |
| arsenic | 0.08 | 0.543 | 0.209 |
| copper | 0.007 | 0.028 | 0.515 |
| manganese | 0.051 | 0.035 | 0.01 |
| mercury | 0.022 | 0.004 | 0.102 |
| nickel | 0.034 | – | 0.535 |
| selenium | 0.18 | 5.84 | 0.109 |
| molybdenum | 0.31 | – | 0.917 |
| vanadium | < 0.002 | – | 0.642 |

All units are mg/L, except pH

**Land application and French drains:** For these methods, water is simply land-applied via irrigation systems or passively drained through perforated pipe. In both cases, the contaminants in the drainage water are released either to the land surface or allowed to move downward in the subsurface. Although this method is very inexpensive, this form of water management carries risks from whatever contaminants exist in the water. For example, the land application at Beal Mountain mine resulted in a near-complete removal of all of the vegetation due to elevated concentrations of thiocyanate, a soil sterilant. Elevated selenium and sodium have resulted in potential plant uptake problems and changes in the soil structure for land applications from heap effluent from the Zortman-Landusky mine in Montana. However, because of the very low expense of pond volume reduction, land application is sometimes used. However, it can in some cases create serious problems.

**Discharge of water to French drains:** This method of disposal of contaminated water has been permitted in Nevada for sites for which ground or surface water contamination is unlikely. While the risk factors in certain situations in extremely arid areas are low, the release of highly contaminated water (some of which meets hazardous waste criteria) into the subsurface has been criticized, and it is unlikely to be permitted for new applications.

**Evaporation:** Particularly for those sites that have high salinity, evaporative methods are one of the few options available for long-term treatment of residual heap drainage water. A recent analysis of alternatives of water management by consultants for a Nevada mine (see Heap 3 Effluent quality in Table 4-4) (Telesto Solutions, 2003) indicated that the most cost-effective method was the use of evaporative ponds. In addition to the sodium load that resulted from the addition of sodium cyanide, the source water was a geothermal water high in dissolved salts. As a result, the number of treatment options was few. Bioreactors would not be effective for treatment of the high salinity, freshwater rinsing would require very large volumes of water (unavailable), and land application had similar issues with salts. The option of geothermal aquifer injection was seriously considered, but was found to be much more expensive than passive evaporation using surface ponds.

Evaporative processes are not completely passive, however, and require regular monitoring to ensure the integrity of the pond liner and the piping system to deliver the water. Most heaps will have a soil cap to limit infiltration of meteoric water, and monitoring of this cap will be required to ensure that it retains the design characteristics. In addition, the salt loading in the heaps can be substantial, particularly when the source water has high salt loading (e.g., from a geothermal aquifer), and it will need to be managed on a year-to-decade time frame.

**Biological treatment:** Biological processes can also be used for heap treatment, particularly when salt concentrations are not excessive. SRBs (discussed below) can be successfully employed for sulfate and nitrate removal, as well as for treatment of selenium and arsenic.

**Membrane processes:** Reverse osmosis and nanofiltration are examples of processes that can also be utilized for treatment of heap effluent, although the costs for long-term treatment of low flows reduce the applicability of these methods that require intensive management and monitoring. Although one option is to accumulate a larger volume of water and follow this by periodic treatment using various membrane processes, this technique has not been utilized extensively. The most extensive literature on applicable membrane processes is in the large-scale desalinization technology. See, for example, the U.S. Bureau of Reclamation literature on desalinization: http://www.usbr.gov/pmts/water/reports.html.

### 4.2.3 Treatment of Mine Water with Microbial Processes

A variety of microorganisms can facilitate the removal of metals, metalloids, and sulfate from mining-impacted waters in both natural and engineered systems. The primary removal mechanism is the formation of oxide, hydroxide, sulfides, or carbonate precipitates. Successful removal of metals and metalloids from mining-impacted waters depends on providing appropriate environmental conditions to promote the desired microbial activity in conjunction with the appropriate chemistry.

Aerobic environments will promote the oxidation of reduced metals, particularly manganese and iron. After oxidation, manganese and iron will precipitate in neutral (or near-neutral) waters and potentially remove other contaminants (e.g., arsenic) by co-precipitation.

Anaerobic environments will promote the reduction of sulfate, nitrate, oxidized metals, and metalloids (e.g., selenium, arsenic, and antimony). A byproduct of a number of anaerobic microorganisms is bicarbonate, which increases the pH and promotes precipitation of metal hydroxides. The production of bicarbonate also promotes the formation of metal carbonate precipitates (e.g., Zn, Mn, and Pb). Biogenic sulfide (produced from sulfate reduction) will promote the precipitation of metal sulfides (e.g., Cu, Cd, Zn, Pb, Ni, and Fe) under a wide range of chemical conditions. Chromium (VI) and uranium (VI) can be reduced by a number of microorganisms (fermenters, sulfate reducers, and iron reducers) under anaerobic conditions to Cr(III) and U(IV), respectively. Subsequently, $Cr(OH)_3(s)$ and $UO_2(s)$ are precipitated from solution. Selenate (Se(VI)) can be reduced to selenite (Se(IV)), which is subsequently reduced to elemental selenium. Under sulfate-reducing conditions, As(V), Mo(VI), and Sb(V) can be reduced and subsequently precipitated as a sulfide mineral ($As_2S_3$, $MoS_2$, $Sb_2S_3$). Some metals will also be removed by co-precipitation with aluminum or iron hydroxides.

The design of microbial treatment schemes needs to consider:

1. Identification of target compound(s) and desired effluent limits,
2. Conditions for desired microbial activity,

3. Conditions for desired chemistry, and
4. Mass transfer and kinetic constraints.

Additional issues that will affect the selection of any treatment process are solids management, operation and maintenance requirements, and cost.

As with other treatment technologies for mining-impacted waters, identification of target contaminants and the associated discharge requirements are necessary for selection of a microbial system. While microbial systems can treat a number of types of contaminants effectively, the microbial treatment options are usually constrained by the contaminant load in the water, as well as the requirements for treatment. When flows are high, and consequently residence time is reduced, insufficient sulfide is generated to precipitate the metals.

Microorganisms need an electron donor and acceptor couple for energy generation, a carbon source and nutrients for cell synthesis, and appropriate environmental conditions. Most microbial-based treatment systems require organic material for the electron donor, which then also serves as the carbon source. The organic material can be supplied in a water-soluble form (e.g., molasses or ethanol) or in a solid form (e.g., wood chips or leaf compost). Water-soluble organics have been used for active bioreactor systems and ground water treatment systems. Solid-phase organics have been used in active and passive bioreactor systems, permeable reactive walls, and wetland systems.

Potential electron acceptors used for energy acquisition include oxygen, nitrate, sulfate, and carbon dioxide. Dissolved oxygen is typically insufficient for desired microbial reactions and must be added either actively or passively. Sulfate is present at adequate concentrations in many mining-impacted waters, particularly those where pyrite oxidation has occurred. Carbon dioxide is sufficient for fermentative reactions involving solid-phase organic matter hydrolysis and production of organic acids and alcohols, which are then available for sulfate reducers.

Nutrient addition (particularly nitrogen) is typically required when water-soluble organic materials are used as the carbon source. Solid-phase organic substrates used are typically a combination of a number of materials (e.g., manure, compost, or wood) and can be selected to include organic material containing sufficient nitrogen.

Mining-impacted waters exist with a range of temperatures, pH, and redox conditions, and microorganisms are sensitive to all of them. Microbial activity tends to decrease with temperature, although the overall rates of reaction (e.g., sulfate reduction) can be kept constant if the number of active bacteria increases proportionally. Most of the desired microbial processes have optimal rates at neutral pH. However, many microorganisms can adapt to lower and higher pH values (5–9) or may be protected from bulk solution–phase pH in microenvironments. Redox conditions are important relative to the electron acceptor used by an active consortium of bacteria. In general, the highest energy couple is used first, followed by those of decreasing energy. However, the presence of a microenvironment and microorganisms with different metabolisms allows concurrent usage of multiple electron acceptors.

Microorganisms alter the chemical environment to promote conditions conducive to desired precipitation or co-precipitation reactions. The changes in chemical environment can include pH, redox, and reactant formation. The theoretical predictions by chemical equilibrium programs, such as PHREEQC (see http://wwwbrr.cr.usgs.gov/projects/GWC_coupled/phreeqc), provide a useful estimate of the potential of precipitates to form, but the added interactions of the microorganisms can alter the expected distribution of precipitates formed.

The rate of precipitation tends to be controlled by the rate of the microbial function of interest (e.g., sulfate reduction). One way the rate of sulfate reduction is controlled is by the rate-limiting step of the microbial community providing growth substrates for sulfate reduction. Models developed to describe the rate of precipitation range from empirical constant-rate models to models that couple microbial kinetics with a selected reactor configuration. Mass transfer is also important in describing the overall rate of precipitation for biotreatment systems. Mass transfer limitations are particularly important for biofilm systems (any system with solid-phase growth media) and will be a function of linear velocity, media size, and biofilm thickness. In biofilm systems, mass transfer can control the observed rate of reaction.

#### *4.2.3.1 Sulfate-Reducing Systems*

Sulfate-reducing systems promote the microbial-facilitated reduction of sulfate, production of sulfide, generation of alkalinity, and reduction of redox active metals, metalloids, or radionuclides. A carbon source, such as lactate or ethanol, is required to promote the growth of sulfate reducers in these systems. Solid-phase organic material can also be used to indirectly provide a carbon source for sulfate reducers from the actions of cellulolytic and fermenting bacteria. Wide ranges of microbial species are able to catalyze sulfate reduction. (See INAP [2003] for a summary of water treatment methods designed specifically for removal of sulfate.)

The range of reactions promoted in a sulfate-reducing system depends in part on the type of carbon source selected. The use of more complex organic compounds results in a greater diversity of microbial population in addition to sulfate reducers. The resulting number of reactions that control sulfate reduction also increases in complexity. The primary reactions of interest are shown in Figure 4-5 at the bottom of this page.

Sulfate-reducing systems may be implemented in active or relatively passive treatment configurations. Relatively passive configurations include anaerobic wetlands, compost-based bioreactors, and PRBs. Relatively passive systems with soluble carbon input include permeable reactive zones (PRZs) and rock-filled bioreactor ponds. Active systems include a number of patented configurations that may include partial sulfate removal as gypsum and recovery of excess sulfide as elemental sulfur. Suspended reactor systems require the highest level of operation and maintenance. A method of removing metal precipitates and excess biomass must be included as part of the overall system. Sulfide precipitation is very effective in reducing a number of metals to low levels. Reduced metalloids also may be effectively removed. The overall effectiveness is dependent on the capture of precipitated metals and metalloids and the stability of the microbial community as a whole. Note that high-flow events, if not bypassed, may damage the microbial community and disperse precipitates downstream, where the precipitates can be dissolved.

#### *4.2.3.1.1 Anaerobic Wetlands*

An anaerobic wetland is a subsurface water body that supports the growth of emergent plants, such as cattails and reeds. The vegetation and sediment provide surfaces for the growth of attached bacteria. Anaerobic removal processes control the treatment of metals and the neutralization of acid. The contaminated water is intercepted and diverted through the wetland system (see Figure 4-6). A minor aerobic component of this system is the surface vegetation, which allows the release of carbon dioxide and hydrogen sulfide, and oxidation of iron on the surface.

Anaerobic wetlands utilize sulfate-reducing bacteria to immobilize metal cations (Fe, Cu, Cd, Pb, Zn), oxyanions (Cr, Se, Sb, Mo, As), and U. In addition, the production of alkalinity allows for the neutralization of excess acid present in target mine waters.

Large areas with a relatively flat topography are required for wetland treatment systems. The area required is a function of the mass loading of each target contaminant.

The removal of metals as metal sulfides is typically based on the expected rate of sulfate reduction (sulfide production). The rate of removal for metalloids and uranium is not as well established and may require bench- and pilot-scale testing.

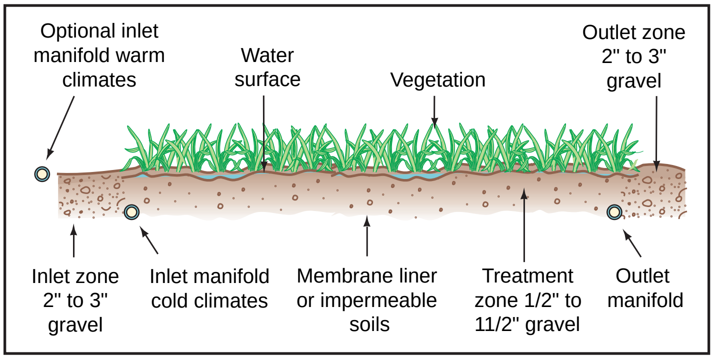

**Figure 4-6.** Schematic of anaerobic wetland with subsurface flow.

*Bioreaction:* sulfate + organic carbon ⇒ sulfide + alkalinity (bicarbonate)

$$2CH_2O_{(aq)} + SO_4^{2-}{}_{(aq)} \rightarrow HS^-_{(aq)} + H^+ + 2HCO_3^-{}_{(aq)}$$

*Chemical reaction:* sulfide + metal ⇒ metal-sulfide and carbonate + metal ⇒ metal-carbonate

$HS^- + Me^{2+} \rightarrow MeS + H^+$ and/or

*Bioreaction:* oxidized metalloid + organic carbon ⇒ reduced metalloid (e.g., Selenate → Elemental selenium)

*Chemical reaction:* reduced metalloid precipitate formation

**Figure 4-5.** Examples of the multiple reactions that can occur under sulfate-reducing condition.

The construction cost for anaerobic wetlands might be lower compared to active treatment. In addition, the operation and maintenance effort and cost are proportionally lower. Costs cannot be generalized on a per-mass basis for target contaminants because of the effect of other important factors such as flow, temperature, and pH.

The precipitation of metals modifies the pore structure within the wetland subsurface and may reduce the effective hydraulic retention time; thus, the design hydraulic residence time should include a suitable safety factor. Low temperature will reduce the bacterial activity and hence the rates of sulfate, metalloid, and uranium reduction. A larger wetland for colder climates will be required relative to more temperate areas. Highly variable flow may result in the flushing of collected precipitates; thus, subsequent polishing ponds are required. Collection of precipitates and sediments, and loss of permeability, will lead to the periodic need to rebuild the entire system.

#### 4.2.3.1.2 Bioreactors

SRBs can be designed in a number of configurations (see Figure 4-7). Configuration A simply treats the influent acidic water and allows the precipitated sludge to settle in the bioreactor, which will ultimately need to be removed, probably by flushing, and appropriately managed. Configuration B allows a more convenient settling of the sludge in a settling basin, which is more easily removed and managed from the bioreactor system. In this configuration a portion of the effluent from the settling basin is recycled to the front of the bioreactor where SRBs reduce sulfate to sulfide. The sulfide-containing water is then mixed with the influent acidic mine drainage. The metals precipitate (mostly as sulfides) in the settling basin, and the pH is raised. The flow rate of discharge is the same as the flow rate of the influent. The pumps used to recycle the water require a power source, although the ease of sludge management will usually outweigh the power costs.

Solid-phase organic material can also be used to provide a carbon source and a surface for bacterial attachment. SRBs have been used to immobilize metal cations ($Cu^{2+}$, $Cd^{2+}$, $Fe^{2+}$, $Pb^{2+}$, $Zn^{2+}$) as metal sulfides, oxyanions (Cr, Se, Sb, Mo, As), and U. They also are effective in reducing sulfate concentrations. The effectiveness for removal via sulfide precipitation is dependent on both the pH and sulfide concentration. For typical sulfide concentrations ($10^{-3} - 10^{-4}$ M), the effective pH for removal of iron sulfide is above pH 6.5.

The use of solid-phase substrate in a packed bed system is affected by the precipitation of metals, which may reduce the hydraulic retention time and exclude flow through portions of the reactive zone. Thus, the initial sizing must take this and the replacement frequency into consideration. Biofilm systems constructed of rock or plastic media may allow the release of precipitated metals, and thus the effluent from these reactors should be polished via gravity settling or filtration. These flushable bioreactor systems can allow continuous use, providing the precipitated metal sulfides, calcite, and biomass are flushed into a collection basin at a frequency that eliminates hydraulic plugging in the bioreactor.

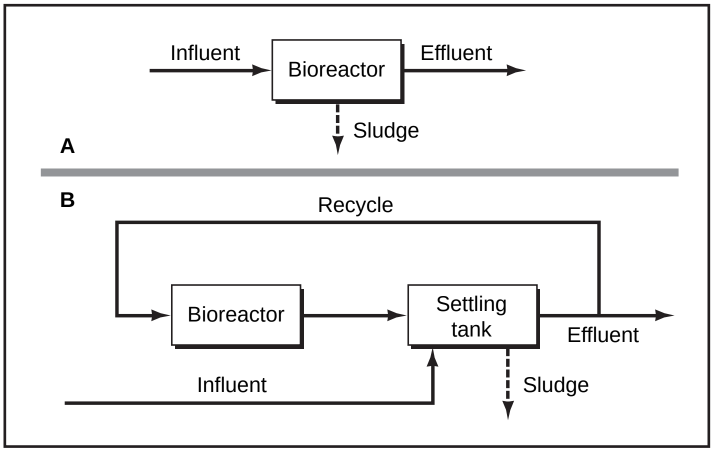

**Figure 4-7.** SRB configurations. A is a simple flow-through system that requires periodic removal of solids from main reactor. B is a modification that includes a separate tank or pond for precipitation and sludge collection.

The optimum pH for SRB systems is 7–7.5, and the effectiveness of SRB systems can be substantially reduced when the influent acidity is high. Experience at the Leviathan bioreactor, which utilizes ethanol as a carbon source to treat an influent flow of 40 L/min, has shown that the system is most effective when the influent pH is adjusted to above pH 4.5 or higher (Tsukamoto et al., 2004). This has been accomplished by the addition of a 25% sodium hydroxide solution, which can be added using a solar-driven pump. While ethanol can be added by a simple gravity flow, the more viscous sodium hydroxide requires a positive pumping system.

Alternatively, the bioreactor can be operated as a sulfide-generating system (Figure 4-7) in which a portion of the bioreactor effluent is recycled back to the front of the bioreactor. Ethanol is added to the bioreactor, and sufficient sulfate remains in the water to allow the SRB system to generate sulfide and add alkalinity. The acidic drainage is then mixed with the effluent from the bioreactor in a settling pond, and the metal sulfide precipitates and is effectively captured in this pond. This configuration allows better management of the sludge and maximizes the SRB activity by keeping the pH close to optimal. However, this configuration also requires pumping approximately 30 – 40 gal/min from the settling pond to the front of the bioreactor and requires an energy source of approximately 0.5 hp.

SRBs offer the advantage of a lower sludge management requirement since the sulfides are precipitated as metal sulfides or as sulfur. Bioreactors can also be managed more effectively at remote locations, with visitations of 1–2 times per month, rather than daily management, as is usually the case with conventional lime/treatment facilities. Site-specific criteria will determine which treatment option provides the most cost-effective approach.

The cost of SRBs varies widely from site to site and is a function of both the system type and the size required to treat the site-specific concentrations and types of contaminants. Simple, flushable lined systems to treat up to 50 L/min can be constructed for under $200,000. The cost of the carbon source (e.g., ethanol at $US2/gal) is generally a relatively small component (< 20%) of the cost of operating a bioreactor. The cost of adding base (generally sodium hydroxide or sodium carbonate) will vary, depending on the acidity. If the acidity (or flow) of the water is sufficient that the cost of raising the pH dominates the cost of treatment, lime treatment will, at some point, become a more cost-effective and reliable option.

#### 4.2.3.1.3 Alkalinity-Producing Systems

Alkalinity-producing systems (APSs) are an integration of ALD systems with anaerobic sulfate-reducing biosystems. Two configurations of APSs have been developed: the successive APS (SAPS) and the reducing APS (RAPS). The SAPS consists of an ALD overlaid with organic material (e.g., hay and manure); the RAPS consists of an ALD integrated with organic material. Under certain conditions, these systems can help increase the pH of influent water sufficiently to allow SRB systems to better thrive, as discussed previously.

APSs target acidity and metals that precipitate as hydroxides or carbonates at slightly alkaline pHs. Relative to active treatment, APSs are inexpensive and have low operation and maintenance costs. However, while they have shown success in certain drainages from coal sites, the applicability in hard-rock mine sites is complicated by surface precipitation of aluminum and iron oxide coatings on the surface that limits the availability of the calcium carbonate for neutralization. Many hard-rock acidic drainage sites have high aluminum (>30 mg/L) concentrations, and even when iron oxidation is inhibited by having anoxic conditions, aluminum coating alone will reduce the effectiveness since precipitation only requires a slight increase in pH to near pH 5 to result in armoring of the limestone.

#### 4.2.3.2 Permeable Reactive Barriers

A permeable reactive barrier (PRB) is a zone of reactive media emplaced in the flow path of contaminated ground water (see Figure 4-8). The reactive media promotes the removal of metals and radionuclides by precipitation, sorption, or ion exchange. The contaminants are retained in the barrier and eventually are removed by excavation. A related subsurface technology is a permeable reactive zone (PRZ) that is created by the injection of a reactive solution in a series of wells that transect a ground water plume.

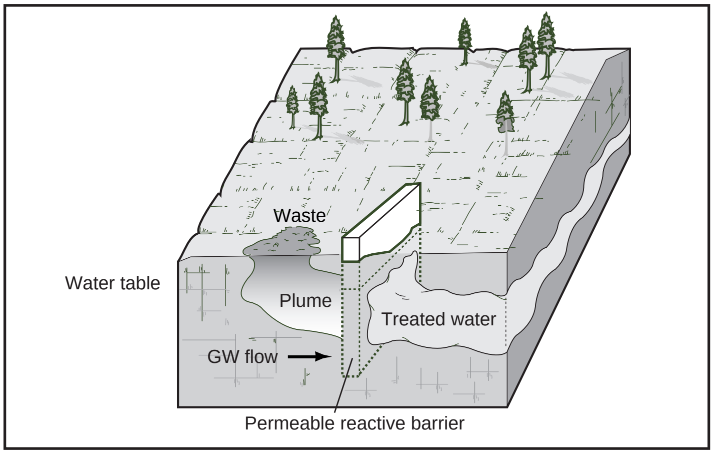

**Figure 4-8.** Schematic of PRB system. (Source: EPA/600/R-98/125)

The reactions promoted in a PRB depend on the reactive media selected and the target contaminant. The main types of reactive media used include organic material (to promote biogenic sulfide production) and zero-valent iron. However, media that promote sorption or ion exchange can also be found. Reactive media types may be mixed to promote the removal of multiple contaminants by different reaction mechanisms.

- Sulfate-reducing biozone reactions
- Zero-valent iron:

  *oxidation/reduction:* $Fe(0) + Cr(VI) = Fe(III) + Cr(III)$

  *and precipitation:* $Cr(III) + 3OH^- = Cr(OH)3(s)$

- *Sorption:*
  $Me^{2+}$ + surface site ⇔ $Me^{2+}$ -surface complex
- *Ion exchange:* $Me^{2+} + R\text{-}Ca = Ca^{2+} + Me\text{-}R$

Sulfate-reducing biozones have been used to immobilize heavy metal cations (Cu, Cd, Pb, Zn), oxyanions (Cr, Se, Sb, Mo, As), and U. Zero-valent iron barriers can reduce and immobilize redox active compounds that include Cr, U, Mo, Sb, Se, and As. Media promoting sorption or ion exchange can be selected to target cations ($Cu^{2+}$, $Cd^{2+}$, $Pb^{2+}$, $Zn^{2+}$, $UO_2^{2+}$) or anions (Cr, As, Mo, Se, Sb).

PRZs have been achieved by the injection of reactive compounds into the subsurface area to be treated. Organic compounds, such as acetate, can promote biogenic sulfide production or biological metal reduction. Inorganic compounds, such as sodium dithionite, can form Fe2+ from Fe3+ on aquifer material. The Fe2+ can then participate in the reduction of Cr(VI) to Cr(III).

The most important implementation issue for PRBs and PRZs is the ability to capture the contaminated ground water flow within the reactive zone. The second issue is the ability to promote the desired chemical reactions, given the chemical composition of the target water. A third issue is the availability of cost-effective reactive media and the frequency of media replacement.

At a minimum, column experiments are required to determine the effectiveness of a specific reactive media configuration and a specific ground water. The microbial and chemical complexity of processes in the PRB and PRZ precludes the use of a cookbook design protocol. The hydrologic and geologic properties of the subsurface also must be adequately characterized to assess potential effectiveness. Uniform mixing and emplacement of the reactive media is another critical factor, as is the ability to maintain an acceptable hydraulic conductivity throughout the reactive zone.

The cost for PRBs and PRZs varies widely from site to site. The cost is a function of both the media type and the barrier or zone size required to treat the site-specific concentrations and types of contaminants. Organic materials are the least expensive, zero-valent iron is more expensive, and the most expensive are specially designed sorptive or ion exchange materials. The frequency and cost of replacement will also vary with media type and the level of contamination. Media has been demonstrated to have a life of about 7 years, but theoretically its life could be 10 years to several decades.

The precipitation of metals modifies the pore structure within the reactive zone and may reduce the hydraulic retention time and exclude flow through portions of the reactive zone. Solutes may be released from the dissolution of solid-phase materials, ion exchange, or desorption.

#### Key Web Site References

- EPA remediation technologies development forum: http://www.rtdf.org/public/permbarr/default.htm
- EPA hazardous waste cleanup information: http://clu-in.org/techfocus/default.focus/sec/Permeable_Reactive_Barriers/cat/Overview/
- University of Waterloo, Department of Earth Sciences, Groundwater Geochemistry and Remediation, Permeable Reactive Barriers: http://www.science.uwaterloo.ca/research/ggr/PermeableReactiveBarriers/PermeableReactiveBarriers.html
- An example of the use of a PRB for treatment of uranium in ground water (Chapter 16) and acid remediation in ground water (Chapter 17): http://www.image-train.net/products/proceedings_first/

### 4.2.3.3 Other Bioremediation Systems

A number of non–sulfate-reducing biosystems have been used for removal of contaminants from mine water. Aerobic wetlands have been commonly used for coal mine drainage and, to a lesser extent, for hard-rock mine drainage. New systems are constantly being developed. The microbial oxidation of elemental sulfur to sulfide for metal sulfide precipitation is a recently developed proprietary process, as is a patented process for the microbial oxidation of manganese. New proprietary systems require independent verification of effectiveness before selection.

#### 4.2.3.3.1 Aerobic Wetlands

An aerobic wetland is a shallow water body (less than 2 ft. deep) with a free water surface that supports the growth of emergent plants, such as cattails and reeds (see Figure 4-9). The vegetation and the sediment provide surfaces for the growth of attached bacteria. Aerobic removal processes control the treatment of metals. The contaminated water is intercepted and diverted through the wetland system.

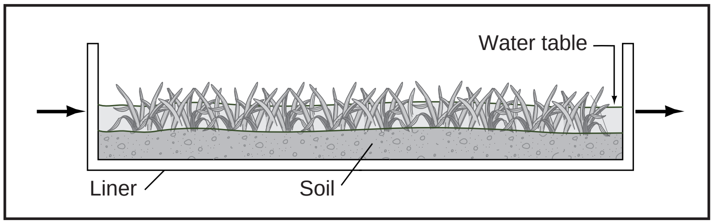

**Figure 4-9.** Schematic of a free water surface aerobic wetland.

The reactions promoted in an aerobic wetland are primarily the oxidation of iron and manganese. The rate of abiotic oxidation is increased by the presence of bacteria.

*Bio-oxidation:* Fe(II) + oxygen $\Rightarrow$ Fe(III) + water
and chemical precipitation $Fe(III) + 3OH^- = Fe(OH)_3(s)$

*Bio-oxidation:* Mn(II) + oxygen $\Rightarrow$ Mn(IV) + water
and chemical precipitation $Mn(IV) + O2 = MnO_2(s)$

Aerobic wetlands for iron and manganese removal are most amenable to near-neutral and net-alkaline waters. Large areas with a relatively flat topography are required for wetland treatment systems. The area required is a function of the mass loading of both iron and manganese. The removal of manganese requires a larger area per unit- mass of manganese removed than for iron.

The construction cost for aerobic wetlands is relatively low compared to active treatment. In addition, the operation and maintenance effort and cost are proportionally lower. Costs cannot be generalized on a per-mass basis for iron or manganese because of the effect of other important factors such as flow, temperature, and pH.

Low temperature will reduce bacterial activity and hence the rates of iron and manganese oxidation. Ice covers will also limit the rate of oxygen transfer to the wetland. Highly variable flow may result in the resuspension of settled iron and manganese precipitates. Most of the successful application of aerobic wetlands has been for coal mine drainages, not metals mine drainages.

### Key Web Site References

- A general discussion of passive mine water treatments: http://www.blm.gov/nstc/library/pdf/TN409.PDF.
- The science of acid mine drainage and passive treatment: http://www.dep.state.pa.us/dep/deputate/minres/bamr/amd/science_of_amd.htm.

Table 4-5 on the following pages provides an overview of water treatment technologies covered in this section, technology selection factors, and limitations.

## 4.3 Mine Pit Lake Management

Lakes are typically "windows to the ground water"—where the land surface drops below the water table, we "see" the water table as the surface of the lake. Mine pit lakes are special cases of this phenomenon, forming in open pit mines that are excavated to below the water table. In practice, excavation below a water table requires dewatering to lower the water table, leaving the open pit (or "void") within the ground water cone of depression. With cessation of dewatering, ground water flows to the center of the cone of depression, forming a lake. Steady-state mine pit lakes can have (1) throughflow to ground water (in some cases, evaporation produces concentration of ground water), (2) outflow to surface water and ground water, or (3) zero outflow (a "terminal" lake, where all inflows are balanced by evaporation).

The ultimate quality of the pit lake is strongly affected by the surrounding wall rock. Wall rock affects pit lake water quality primarily by leaching solutes released by the oxidation of sulfide minerals exposed in the pit. Further, dewatering of sulfide zones can pull air into surrounding aquifers, potentially inducing regional oxidation in aquifers and increasing, temporarily at least, solutes in ground water. The depth of rapid oxidation may be limited to a few meters into the face. Rock that is net neutralizing will produce a pit lake that is relatively benign since the problematic divalent metals concentrations will be low. Arsenic, antimony, and selenium, which are mobilized at elevated pH, can be a concern under conditions where the pH is alkaline and insufficient iron is present to cause co-precipitation of these constituents.

Pit mines are by definition in areas of elevated metals, and groundwaters often contain elevated trace metals or sulfate. When ground water is a dominant source of inflow, the effect of ground water quality on pit lake quality tends to increase with increasing lake size. Ground water quality can change over time, particularly given the long times required to fill lakes. Where evaporation is large and ground water outflow small, lakes will concentrate solutes and can eventually becomes sources of ground water exceeding water quality standards for TDS or other solutes.

*(Continued on page 32)*

**Table 4-5.** Water Treatment Technologies for Hard-Rock Mining Effluent

| Technology Name | Technology Description | Target Analytes | Critical Feasibility Factors | | | Important Limitations |
|---|---|---|---|---|---|---|
| | | | Implementabilty | Effectiveness | Cost | |
| **Conventional Lime Treatment** | Lime or hydrated lime is mixed as 10-15% slurry and added to acidic water to raise the pH of the water and precipitate metals as metal oxides and sulfate as gypsum | Acidity. Most divalent metals. Al, As, Sb, sulfate (to 2,000 mg/L). | • Requires engineered system to efficiently utilize lime, including power, pumps, tanks, mixers, and lime addition systems | • Generally considered the most proven method for acid drainage treatment<br>• Depends on types of metal loading<br>• Can treat the most concentrated acidic drainages | The most cost-effective method for treating large flows or highly contaminated water. Less cost effective for small streams due to fixed costs. | • Requires frequent monitoring and sludge management<br>• Arsenic treatment effective only with a high iron-to-arsenic ratio |
| **Limestone Ponds and Open Limestone Channels** | Establish open ponds or channels that can receive acidic water. The limestone neutralizes the acids and allows precipitation of a variety of metals as metal oxides. | Acidity, Al, Fe, Mn. Partial metal removal. | • Acidic water is (generally) passively added to the limestone pond or channel and allowed to react<br>• Turbulent systems improve release of alkalinity | • Variable, depending on the aluminum, iron, and acidity<br>• Armoring is a problem<br>• Usually low maintenance | • Relatively inexpensive and low maintenance<br>• Depends on the availability of limestone and construction costs | • High aluminum and iron waters will armor the limestone and reduce effectiveness<br>• Precipitated sludge may require management, depending on location and regulations<br>• May not treat certain divalent metals well (Cd, Cu, and Zn) |
| **Anoxic Limestone Drains** | Intercept acidic water that primarily has ferrous iron and pass this water through limestone beds under anoxic conditions. This limits the amount of oxidation of the iron and limits the amount of precipitation on the limestone. | Acidity, Al. Some metal reduction is observed. | • Care must be taken to maintain anoxic conditions<br>• Generally need sloping topography and passive transport of water | • Shown to be useful for coal acidic drainage, but less so for hard-rock mine drainage and heavy metals<br>• Decreased overall rate of reaction<br>• Longer residence times provide better neutralization and decrease in target analytes | Relatively inexpensive. Some maintenance cost is required if a biological system is used to maintain anoxic conditions. | • High aluminum-containing waters will armor limestone and decrease the rate of alkalinity addition<br>• It is difficult to remove all of the oxygen, so some iron is oxidized and tends to armor the limestone<br>• Unless sized appropriately, these systems will not respond well to large fluctuations in volume or influent water quality |

**Table 4-5.** Water Treatment Technologies for Hard-Rock Mining Effluent *(continued)*

| Technology Name | Technology Description | Target Analytes | Critical Feasibility Factors | | | Important Limitations |
|---|---|---|---|---|---|---|
| | | | **Implementabilty** | **Effectiveness** | **Cost** | |
| **Anaerobic Wetlands** | Intercept surface water flow and distribute through one or more sub-surface water wetlands | Fe, Zn, Cu, Cd, Pb, As, Cr, Mo, Sb, Se, U, sulfate, low levels of acidity | • Steepness of slope<br>• Sufficient land area | • Sensitive to low temperatures<br>• pH $>$ 5 and moderate metal loading | • Excavation<br>• Plants and supporting soil<br>• Hydraulic structures | • Relatively low flows<br>• Large land areas and flat topography<br>• Periodic sediment removal and wetland reestablishment required<br>• Oxidation and release of metals and sulfides is probable if the wetlands become dry<br>• Difficult to control metal migration |
| **Sulfate-Reducing Bioreactors** | Collect flow with pumps or natural hydraulic gradient and distribute through a vessel containing growth substrate (manure, wood chips, other organic waste) and sulfate-reducing bacteria. SRBs reduce sulfate, raise the pH, and precipitate metals. | Fe, Zn, Cu, Cd, Pb, As, Cr, Mo, Sb, Se, U, sulfate, low levels of acidity | • Availability of inexpensive organic substrates<br>• Power availability for active systems<br>• Accessibility for system maintenance<br>• Sufficient land area for passive systems | • pH $>$ 5<br>• Moderate metal loading<br>• Method of retaining metal precipitates<br>• Longevity is dependent on carbon source and the ability of SRB to maintain a pH sufficiently high to support SRB activity | • Growth substrate<br>• Bioreactor<br>• Additional tanks or ponds for process modifications | • Best for water above pH 5; effluent metal concentration may exceed discharge limitations when flows or contaminant concentrations are high<br>• Systems with media that create small pores sizes (mm) are more prone to clogging by metal precipitates<br>• Longevity is dependent on carbon available to the microbial consortium |
| **Alcohol Amended Sulfate-Reducing Bioreactor** | Alcohols (e.g., ethanol) and base added to lined impoundments containing rocks, wood chips, or other physical support. Bacteria use the alcohols as reducing sources for sulfate. The system is designed to manage sludge efficiently. | See above | See above | Alcohol and base addition can be controlled and allow better treatment of varying flows and contaminant loads. Sludge management and hydraulic control are improved compared to the more passive SRB systems. | • Higher initial costs for construction, as well as the costs of alcohols, base, and nutrients<br>• Allows substantially improved longevity of the bioreactor due to lack of plugging and a continuous carbon source | • Although these systems are more adaptable to variations in flow and contaminants, monitoring is required to maintain the bioreactor operation<br>• Requires a continuous source of carbon, base, and planned sludge management |

**Table 4-5.** Water Treatment Technologies for Hard-Rock Mining Effluent *(continued)*

| Technology Name | Technology Description | Target Analytes | Critical Feasibility Factors | | | Important Limitations |
|---|---|---|---|---|---|---|
| | | | Implementabilty | Effectiveness | Cost | |
| **Alkalinity-Producing Systems** | Intercept surface water flow and distribute through a series of shallow drains containing both limestone and reducing organic material. Metals are precipitated as metal oxides and metal carbonates. | Acidity, Al. Some metal reduction is observed. | • Steepness of slope<br>• Sufficient land area | • Flow<br>• Acid-loading rate<br>• Metal-loading rate | • Excavation<br>• Limestone<br>• Reducing organic material<br>• Hydraulic structure | • Experience primarily based on coal mine<br>• Improves water quality, but may not meet stringent discharge standards<br>• Periodic exchange of substrate required, but time frame not well established |
| **Permeable Reactive Barriers (“Reducing Reactive Walls”)** | Intercept contaminated ground water plume with a permeable barrier constructed of reactive material. Water flows through and contaminants are retained. | See above | • Stability of trench wall during installation<br>• Plume width<br>• Depth to ground water and bottom of aquifer | • Homogeneous emplacement of barrier material or injection of reactive solution<br>• Column studies required to assess potential effectiveness<br>• pH $> 5$ and moderate metal loading | • Reactive material<br>• Excavation and dewatering during excavation<br>• Soil and ground water disposal from construction<br>• Thickness along flow line to achieve residence time | • Uncertainty in PRB life – affects cost of technology<br>• Periodic replenishment of reactive media expected, but frequency not well established<br>• Concurrent iron reduction may mobilize metals sorbed to iron mineral surfaces<br>• Sulfate reduction rates ~50 mg/L-d. Rate affects cost of technology. |
| **Aerobic Wetlands** | Intercept contaminated surface water and flow through one or more free water surface wetlands. Iron and manganese oxidation form species that are less soluble and tend to precipitate as $Fe(OH)_3$ and as $MnO_2$ , respectively. Arsenic can be removed by co-precipitation with iron hydroxides. | Fe, Mn, As | • Steepness of slope<br>• Sufficient land area | • Near-neutral pH required to maximize oxidation reactions<br>• High-flow variations may re-suspend metal precipitates | • Excavation<br>• Plants and supporting soil<br>• Hydraulic structures | • Periodic sediment and precipitate removal and wetland reestablishment required<br>• Experience primarily based on coal mine drainage |

Numerous studies of mine pit lakes indicate that they behave in accordance with well-understood processes of limnology (e.g., Atkins et al., 1997). The fundamental physical process is mixing, which is a balance between wind shear acting on the surface, which tends to increase mixing, and stable density stratification caused by temperature and salinity gradients, which tend to inhibit mixing. As a result, pit lakes that mix annually (i.e., most U.S. lakes) can be approximated as stirred reactors, where ground water inflow is mixed into the lake each year, and in most cases, the water is oxygenated at least part of the year. A chemical mass balance on solutes needs to incorporate loads from inflow and outflow of ground water, precipitation, surface flows, and loss to precipitation and adsorption. This analytical solution is typically used in predictive lake models.

Biological productivity in lakes is superimposed on the physical stratification. In natural lakes, this is primarily the use of light energy by phytoplankton to convert carbon dioxide into cell mass and oxygen. Productivity in natural lakes is typically limited by nutrients, particularly phosphate. Highly productive lakes can also become anoxic at depth as dissolved oxygen is consumed in reactions with organic detritus from the productive surface. In mine pit lakes, where sulfate concentrations are elevated, the presence of anoxic conditions induced by elevated organic carbon will generally result in reduction of sulfate to produce alkalinity and hydrogen sulfide.

Finally, several studies of existing mine pit lakes demonstrate that they generally respond as predicted by established limnologic studies. Detailed measurements of seasonal profiles in mid-latitude pit lakes show that even in steep-sided lakes with high walls, the lakes stratify from surface warming during the summer, then completely mix in fall and spring. As important, observed physical stratification and biological productivity in these pit lakes matched accurately with predictions using a numerical model (Atkins et al., 1997). Field-scale nutrient addition has demonstrated that pit lake productivity can be reliably increased through the addition of limiting nutrients (e.g., Martin et al., 2003).

These fundamental characteristics of lakes management—the ability to isolate denser deep layers, to induce biological productivity and sulfate reduction, and to reliably simulate these phenomena with models—lay the foundation for remedial strategies that treat in-situ metals-contaminated pit lakes and even use pit lakes as reactors to treat mine effluent from other facilities.

### 4.3.1 Backfilling and Neutralization

Backfilling pits completely with waste rock or tailings can preclude the formation of a pit lake and can also provide permanent stable disposal of sulfidic waste rock below a water table. However, backfilling with reactive rock typically produces a plume of sulfate and other solutes released by partial oxidation caused by handling, particularly if the pH of the backfilled material is not controlled. Backfilling is often eliminated based on cost, generally over $US1/tonne, depending on site conditions, but can greatly exceed this cost at challenging sites.

Partial pit backfilling is an option that is becoming increasingly common in precious metals pits. Particularly for large pits, partial pit backfilling can be done as part of a mine plan and reduces the haulage costs of waste rock out of the pit. Reactive rock placed appropriately in the bottom of a pit during mining will then be flooded when mining is complete and effectively eliminate further oxidation of the rock that is placed below the water table. Handling of sulfide rock produces some oxidation: reaction of sulfide minerals with the oxygen in the pore space of backfilled waste rock will produce ~500 mg/L sulfate in the first flush of water, and any additional handling-induced oxidation adds to this baseline. In this case, lime or some other neutralization agent can be added to maintain a neutral pH as the acids are rinsed off the rock as the water table recovers. The effectiveness of this option was demonstrated by treatment of a large acidic pit lake using lime at the Sleeper Mine in Nevada.

Treatment of acidic pit lakes can be achieved using direct addition of powdered lime (CaO), hydrated lime ($Ca(OH)_2$) or limestone ($CaCO_3$), and treatment costs can be very low (if a local source is available, limestone crushed to $<$ 2 mm can typically be obtained for $US5 – 10/tonne, yielding lake neutralization costs of a few cents per cubic meter). However, the local type of limestone near hard-rock mining is usually not as reactive as other forms of process neutralization agents, which could increase costs. Neutralization precipitates iron and typically removes most heavy metals by co-precipitation or adsorption. However, a neutralized pit lake typically contains below 3,000 mg/L sulfate.

### 4.3.2 Bioremediation and Induced Stratification of Mine Pit Lakes

Water in hard-rock mine pit lakes can in some cases be acidic, and regardless of pH, they can contain concentrations of sulfate or metals that may be problematic. Pit lakes vary enormously in size, from a few acre-feet to over 400,000 acre-feet. Remediation requirements include monitoring only (where water quality is good), single or infrequent treatment (e.g., where sulfidic wall rock is eventually inundated by the lake and oxidation ceases), or perpetual treatment (e.g., where sulfidic wall rock remains above the lake, loading solutes in runoff or by direct sloughing). In-situ treatments include stratification, which isolates deep lake water from oxygen at the surface and potential exposure to terrestrial animals, and biotreatment technologies, which induce mineral formation, adsorption, and/or chemical reduction reactions that remove metals from solution. These technologies, often combined, offer lower-cost options for closure and management of mine pit lakes.

In-situ bioremediation induces chemically reducing conditions in lakes that remove target analytes by either transforming them to another form (e.g., acidity, sulfate) or inducing them to precipitate as insoluble minerals that settle out of solution (e.g., heavy metal sulfides). Bioremediation is a relatively well-established alternative for treatment of mine pit lakes (Castro and Moore, 2000) and has been successfully demonstrated in microcosm (Frömmichen et al., 2004) and full-scale (Poling et al., 2003) applications. Specific reactions include biologically induced reduction of sulfate to sulfide in a lake (Castro et al., 1999), which leads to precipitation of dissolved metals as sulfide (CdS, CuS, PbS) and reduction to a less soluble reduced form [U(VI) to U(V), Sb(V) to Sb(III), or Cr(VI) to Cr(III)]. Chemical reactions involve reduction of a target analyte by organic carbon. Example reactions (using CHO to represent organic carbon source) include reduction of sulfate to sulfide, which can also be used to neutralize acidity (Frömmichen et al., 2004):

$$2CH_2O_{(aq)} + SO_4^{2-}{}_{(aq)} + 2H^+ \rightarrow H_2S_{(aq)} + 2CO_{2\,(aq)} + 2H_2O$$

and reduction of metals to a less soluble form:

$$2CHO + SbO_3^- = HCO_3^- + Sb(OH)_{3(S)}$$

Target analytes are then removed from solution by being converted to a reduced form that precipitates as oxides (e.g., $UO_2$), hydroxides [e.g., $Sb(OH)_3$, $Cr(OH)_3$)], or metallic sulfide (e.g., FeS, CdS, CuS, and ZnS). In addition, enhanced biological productivity increases biomass, which can effectively remove metals such as zinc and cadmium, which adsorb and settle with detritus (Martin et al., 2003). Ideally, the long-term fate of precipitated solutes is burial in sediments in a chemically stable form.

Two fundamentally different approaches are used to introduce organic carbon to pit lakes:

- *Organic carbon addition:* the direct addition of soluble organic carbon reagents, typically alcohols, sugars, or organic waste, to the lake (e.g., Castro et. al., 1999)
- *Nutrient addition:* typically phosphate and nitrate, which stimulate the growth of aquatic biota (algae, phytoplankton, and zooplankton) near the surface of the lake, producing biologic detritus that induces reducing conditions at depth as it settles through the lake (Pederson et al., 2003; Poling et al., 2003)

Both treatments can result in rapid production of biomass, producing organic detritus that can adsorb and settle out dissolved metals.

Where sulfide production is desired, anoxic conditions must be created and maintained long enough to allow the biologically induced reactions between organic carbon and sulfate. This is where stratification is required—anoxic conditions generally require that a lake be stratified (thermally and/or chemically) during at least part of a year so that a deep anoxic zone can form in isolation from the atmosphere. Thermal stratification generally occurs each summer in temperate climates and can be a long-term natural condition in very cold or tropical climates. More stable stratification can also be induced by actively maintaining a layer of less dense water [e.g., warmer and/or less saline than the deep water (Poling et al., 2003)] on a lake.

Direct carbon source addition has higher material costs, but the treatment is generally rapid (reactions completed over a few seasons) and may thus be best where infrequent treatment is required. Nutrient addition has much lower material costs, but it generally requires longer treatment times and a more detailed analysis of lake limnology, and it may be more practical where long-term management is anticipated. Both methods have been demonstrated in full-scale applications.

Finally, developing technologies such as metal-specific microbes that precipitate arsenic and selenium as sulfides in very low-volume sludges may offer potential for more targeted treatment of metalloids. These are noted as possible future remedies for these often difficult-to-treat metalloids, but are not addressed further here.

For *nutrient addition*, the effectiveness of inducing organic carbon formation with nutrients can be estimated using standard engineering relationships for lakes (Thomann and Mueller, 1987; Martin et al., 2003). In practice, the effectiveness of nutrient addition will be limited in part by the organic carbon production rate, and the efficiency of sulfate reduction depends on the reaction rate, temperature, detritus settling rate, and reactivity of organic carbon in the sediment. Application of nutrient-addition treatment should anticipate a site-specific pilot test, numerical modeling to estimate dose rates, and several years of active treatment.

For *organic carbon addition*, dosing depends on the stoichiometry between organic carbon and the desired target reactions, adding sufficient organic carbon to remove oxygen and then producing sufficient sulfide to precipitate the heavy metals in solution. Direct reduction of specific elements to less soluble forms (e.g., U, Cr, Se, As, Sb) is less widely described and may require pilot-scale demonstration. If the water quality in the pit is sufficiently poor that the microbial community cannot thrive, alternative, pre-biological treatments may be necessary.

For *induced stratification*, a supply of less dense water for maintaining a capping layer is generally required. This can be warmer water (e.g., power plant cooling water) or less saline water (e.g., fresh water over a saline lake). The viability of maintaining an isolated deep layer of dense water can be evaluated with a numerical limnologic model [e.g., CEQUAL/W2 (Cole and Buschek, 1995)] using site-specific parameters for bathymetry, water salinities, and climate.

### Performance and Cost Data

Examples are provided at right that demonstrate the technology in practice for carbohydrate additions (commonly sugar industry byproducts) and nutrient additions (typically nitrate and phosphate).

### Carbohydrate Addition

| **Site Name and Location** | *Gilt Edge Pit Lake, South Dakota, USA. (Arcadis, Inc., not published)* |
|---|---|
| Experimental Design | In-situ pit lake (volume = 65 million gallons, dimictic lake). NaOH (125 tons, to increase pH), alcohol, and sugar in three stages over a summer (producing ~100 mg/L initial dissolved organic carbon in the lake). Duration of monitoring: 3 years. |
| Results | Cadmium, copper, lead, nickel, arsenic, selenium, and zinc decreased from above to below treatment objectives after treatment, including copper from 20 to 0.05 mg/L, cadmium from 0.2 to 0.02 mg/L, and zinc from 5 to 0.9 mg/L. Monitoring for excess sulfide in the pit lake during treatment was identified as an important issue. |
| **Site Name and Location** | *Koyne/Plessa lignite field, Germany (Frömmichen et al., 2004)* |
| Experimental Design | Laboratory microcosm. Ethanol, sugar industry byproduct (Carbocalk), and wheat straw dosed at 3.9 kg/m$^2$ Carbocalk and 9.3 kg/m$^2$ wheat straw. Duration of monitoring: 1 year. |
| Results | pH increased from 2.6 to 6.5, neutralization rate 6 to 15 equiv/m$^2$-yr |

### Nutrient Addition

| **Site Name and Location** | *Island Copper Mine Pit Lake (Poling et al., 2003)* |
|---|---|
| Experimental Design | Field-scale pit lake; water volume 241,000,000 m$^3$ (and ~5 million m$^3$ ARD added to deep layer); permanently stratified with seawater hypolimnion; brackish epilimnion [a 5-m thick brackish layer is maintained over a more saline (seawater) hypolimnion]. The contaminant load was moderate, with a range of 5 – 10 mg/L heavy metals and 500 – 2,000 mg/L sulfate. Liquid nitrate and phosphate (N:P = 6:1) added every 10 days to surface using a small boat. Duration of monitoring: 6 years. |
| Results | Treatment produced effective removal of zinc, copper, and cadmium from the lake while maintaining acceptable water quality in the epilimnion layer. Ongoing treatment is estimated at $100,000/yr and is treating between 4 and 6 million m$^3$/yr of acidic inflow. |

***Nutrient Addition*** (continued)

| **Site Name and Location** | *Equity Silver Mine, British Columbia, Canada (Martin et al., 2003)* |
|---|---|
| Experimental Design | Microcosm using "limnocorals" in a dimictic existing pit lake. Addition of 0.7, 1.4, and 14 mmole P/m$^2$/week. Duration: 1 year. |
| Results | High nutrient loading produced dramatic increase in algal productivity in the epilimnion and efficient removal of metal cations (e.g., zinc from 150 to 20 µg/L; copper from 3 to 0.1 µg/L, cadmium from 6 to 2 µg/L, and nickel from 15 to 5 µg/L). The removal mechanism is adsorption of metals to biogenic particles, which then settle out. |

Following several successful full-scale applications, in-situ bioremediation of mine pit lakes appears to be relatively well accepted by the scientific community, industry, and regulators. Successful carbohydrate treatments have been demonstrated using natural organic carbon (Frömmichen et al., 2004) and alcohols plus sugar (http://www.arcadis-us.com). Nutrient addition with induced stratification is providing ongoing treatment at the Copper Island Mine (Poling et al., 2003). Limnologic models are mature and have demonstrated the ability to reliably predict physical mixing and biological productivity in mine pit lakes. In-situ bioremediation of pit lakes offers the potential in some cases for much lower cost treatment, particularly using nutrient addition. However, this remains a research area, with site-specific conditions dramatically affecting implementability. Potential cost savings thus need to be weighed against current uncertainty and associated higher potential costs for research and characterization.

Table 4-6 on pages 36 and 37 provides an overview of pit lake treatment technologies covered in this section, technology selection factors, and limitations.

## 5.0 CONCLUSION

Each mine disturbance that is the source of contaminated water requires careful consideration of site-specific characteristics prior to choosing a strategy to manage the water. The large majority of mine drainages will require long-term treatment, on the order of decades and beyond. Few walk-away options are available, and financial requirements for in-perpetuity treatment are a significant component to the decision on which treatment option to use. Site characterization is critical and should address the following questions:

- What is the potential for reducing the flow of the water?
- What is the highest volume of water that will need to be treated during major events?
- What is the water quality, and how does it vary seasonally?
- What are the regulatory discharge requirements?

Many types of rock will only go acidic after several years, and the rate of acid generation will change over time, often increasing for several years as the oxidizing bacteria become widespread. What level of data are required to accurately predict these changes?

Finally, each treatment technology presents different financial and treatability considerations that may require pilot-scale testing in the field, in order to demonstrate that it will indeed treat the mine water to acceptable discharge limits over the long term. The state and federal regulatory agencies, public interest organizations, and the mining industry all are increasingly focused on issues related to mine water treatment. This emphasis is unlikely to go away, since the long term treatment costs can be very high. Confidence that a treatment option will actually do the job requires continual technical and financial evaluation of each option, public release and dissemination of treatment data, and continued research on new methods for mine water treatment.

**Table 4-6.** Pit Lake Treatment Technologies for Hard-Rock Mining

| Technology Name | Technology Description | Target Analytes | Critical Feasibility Factors | | | Important Limitations |
|---|---|---|---|---|---|---|
| | | | Implementabilty | Effectiveness | Cost | |
| **Induced Stratification and Bio-remediation** | Lake is stratified, using either a low-salinity cap layer over saline lake or warm-layer cool hypolimnion. Organic carbon can be created by adding nutrients, and metals adsorb and settle with organic detritus. Alternatively, direct carbohydrate addition can produce $H_2S$, precipitating metals as metallic-sulfide minerals. | Primarily heavy metals: Cd, Zn, Cu, Pb, Ni, U. Possibly metalloids As, Sb, Se. Possibly $SO_4$. | • Most lakes stratify naturally each summer, simplifying isolation of the hypolimnion<br>• Salinity stratification requires saline lake and fresh-water source<br>• Production and release of excess $H_2S$ to the atmosphere can be a health risk<br>• Access to lake is required for reagent addition | • Metal cation removal is typically effective<br>• Removal by adsorption to detritus may require several seasons<br>• Metalloid removal mechanisms are not well known<br>• Long-term stability of metals in sediments uncertain; periodic re-treatment may be required | • Materials can be significant cost<br>• Carbohydrate (sugar or alcohols): ~\$0.5 – 1.0 per kg<br>• Ammonium polysulfate (10-34-0) solution and urea ammonium nitrate (28-0-0) solution prices depend on local availability | • Inducing and maintaining stratification requires dense deep water (saline or cold) and a supply of low-density water (fresh and/or warm) for surface layer. Inducing reducing conditions can mobilize metals in sediments.<br>• Several seasons of treatment may be required<br>• Carbohydrate addition is patented<br>• Sulfide production must be closely controlled to avoid health risk |
| **Backfilling: Partial or Complete** | Waste rock and/or tailings are used to partially or completely fill the pit. Removes open surface water and access to humans and wildlife. Reactive backfill may require amendment to reduce acidity or other solute release. | Eliminates surface exposure to all analytes | Requires proximal source of waste rock or tailings for backfilling | • Reduces or eliminates further oxidation of rock below water table<br>• Reduces the water volume in the pit—important in arid areas | • Depends strongly on the mine plan<br>• Costs are low if partial backfill occurs during mining. Backfilling from rock outside the pit at \$1/tonne or higher. | Sometimes difficult to accurately predict the water quality that will result from rinsing backfilled material. Can degrade quality in throughflowing ground water. |
| **Accelerated Filling** | Surface or ground water is used to rapidly refill pit. This reduces the time for sub-aerial wall rock oxidation and may reduce the rinsing of reactive rock surfaces into the pit lake. | All analytes associated with wall rock oxidation, i.e., sulfate, heavy metals, and metalloids | Requires access to source of water. River diversion can allow rapid filling, while ground water pumping is typically slower and more expensive. | Can significantly improve water quality over what would have existed by refilling by ground water recovery. However, a rapid volume increase could force treatment earlier in time, increasing costs. | During refilling, continuous operating ground water pumps are typically the primary cost | • Rapid pit lake refilling may force poor water back into the ground water system<br>• Appropriate monitoring is required to fully understand the impacts to aquifer surrounding the pit |

**Table 4-6.** Pit Lake Treatment Technologies for Hard-Rock Mining *(continued)*

| Technology Name | Technology Description | Target Analytes | Critical Feasibility Factors | | | Important Limitations |
|---|---|---|---|---|---|---|
| | | | Implementabilty | Effectiveness | Cost | |
| **Neutralization** | Lime or other neutralizing agents are added to the pit lake. Adequate mixing (natural turnover or multi-level injection) is required to completely mix in oxygen (to oxidize Fe) to neutralize acidity throughout the lake depth profile. | Primarily heavy metals: Cd, Zn, Cu, Pb, and Ni. Possibly metalloids As, Sb, Se, and sulfate. | Well-demonstrated technology using lime addition from either floating barge or amended inflow water | Metals removal is similar to what is observed with conventional lime treatment—very effective for acidity and metal cations, less effective for oxyanion metalloids (e.g., As, Sb, Se) | Highly variable cost depending on lake acidity, cost for delivered lime, and the method used to add the lime to the lake | • Lime added to the surface may become coated, reducing efficiency<br>• Oxidation of ferrous iron is necessary for effective iron removal<br>• Sulfate concentrations are below ~ 3,000 mg/L |

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## 7.0 ACRONYMS AND ABBREVIATIONS

| | |
|---|---|
| ABA | acid base accounting |
| ACMER | Australian Center for Mining Environmental Research |
| ADTI | Acid Drainage Technology Initiative |
| AGP | acid-generating potential |
| ALD | anoxic limestone drain |
| AMD | acid mine drainage |
| ANP | acid-neutralizing potential |
| ANSTO | Australian Nuclear Science and Technology Organization |
| APS | alkalinity-producing system |
| ARD | acid rock drainage |
| ASTM | American Society for Testing and Materials |
| BC | British Columbia |
| BLM | U.S. Bureau of Land Management |
| CN | cyanide |
| DOE | U.S. Department of Energy |
| EPA | U.S. Environmental Protection Agency |
| ETSC | U.S. EPA Engineering Technical Support Center |
| ICARD | International Conference on Acid Rock Drainage |
| INAP | International Network for Acid Prevention |
| MEND | Mine Environmental Neutral Drainage |
| NAG | net acid-generating (test) |
| NDEP | Nevada Division of Environmental Protection |
| NMA | National Mining Association |
| NNP | net-neutralizing potential |
| NRC | National Research Council |
| O/M | operation/maintenance |
| OSC | on-screen coordinator |
| OSM | U.S. Office of Surface Mining |
| PIRAMID | Passive In-Situ Remediation of Acidic Mine/ Industrial Drainage |
| PRB | permeable reactive barrier |
| PRZ | permeable reactive zone |
| RAMS | Restoration of Abandoned Mine Sites |
| RAPS | reducing alkalinity-producing system |
| RCRA | Resource Conservation and Recovery Act |

| | |
|---|---|
| RCTS | Rotating Cylinder Treatment System |
| RPM | remedial project manager |
| SAIC | Science Applications International Corporation |
| SAPS | successive alkalinity-producing system |
| SME | Silica Micro Encapsulation |
| SPLP | Synthetic Precipitation Leaching Procedure |
| SRB | sulfate-reducing bioreactor |
| SRK | SRK Consulting Engineers and Scientists |
| TCLP | Toxicity Characteristic Leaching Procedure |
| TDS | total dissolved solids |
| Tonne | metric ton |
| UNR | University Nevada–Reno |
| WAD | weak acid dissociable |

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