

# **Environmental Mine Waste Management: Strategies for the Prevention, Control, and Treatment of Problematic Drainages**

## **Volume 1 of 2**



**Advances in Mine Waste Management Project  
Final Report to the Minerals Coordinating Committee  
June 30, 2001**

**Minnesota Department of Natural Resources  
Division of Lands and Minerals**



**ENVIRONMENTAL MINE WASTE MANAGEMENT:**

**STRATEGIES FOR THE PREVENTION, CONTROL, AND**

**TREATMENT OF PROBLEMATIC DRAINAGES**

Volume 1 of 2

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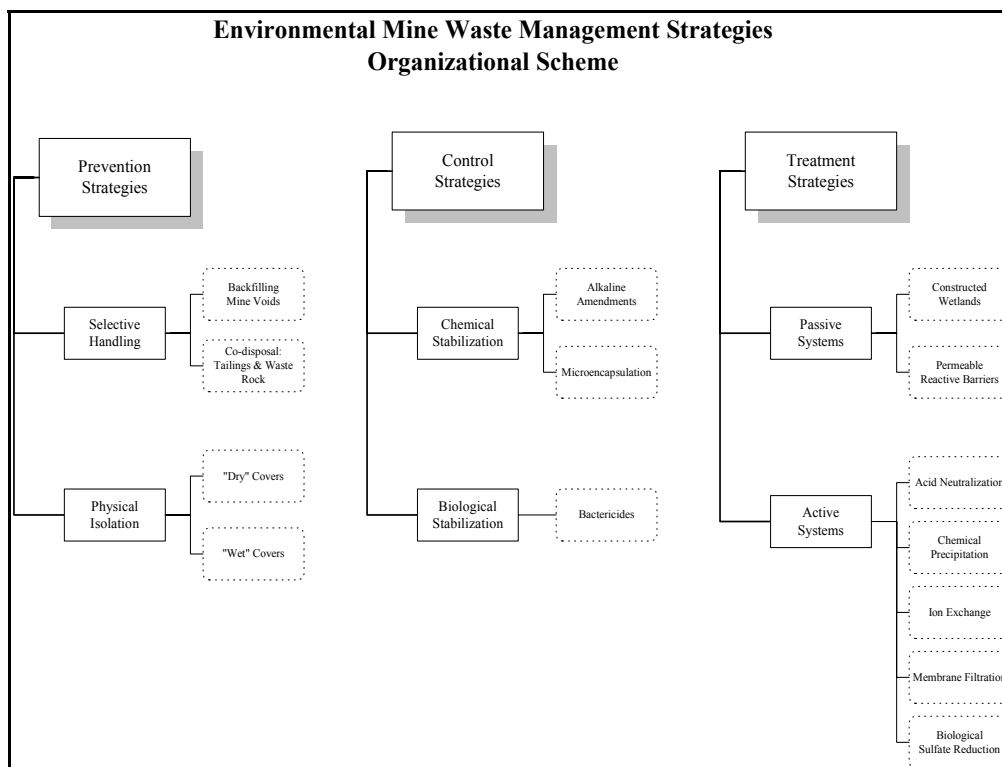


## EXECUTIVE SUMMARY

Considerable mineral potential for base and precious metals is associated with Minnesota's Precambrian rocks, specifically in Archean metavolcanics, metasedimentary formations, and the Duluth Complex. If mineral development occurs, the potential for generation of acidic mine waste drainage is the primary water quality concern. The goal of this document is to evaluate current methods and identify strategies for environmental mine waste management that show potential for application in Minnesota.

A search of literature contained in the Minnesota Department of Natural Resources, Division of Lands and Minerals literature database and various mine waste management conference proceedings resulted in a large number of citations. Consequently, it was necessary to refine the scope of this document. This was accomplished by reviewing similar efforts in Canada, Australia, the United States, and Sweden to compile information on the current state of environmental mine waste management technology.

As a result of the literature search and scope refinement, fifteen environmental mine waste management strategies were identified for further investigation. In order to create an organized and efficient presentation, the fifteen strategies were divided into three generalized categories: prevention, control, and treatment. The first category deals with strategies intended to **prevent** the generation of problematic drainages. Strategies that **control** problematic drainage in situ are considered in the second category. Finally, existing problematic drainages generally require **treatment**, the third category. Within this context, the fifteen target strategies are arranged as follows:



Selected literature regarding these fifteen environmental mine waste management strategies will be reviewed in terms of: 1) a brief description; 2) methods used to assess performance; 3) generalized cost analysis; and 4) applicability to environmental mine waste management issues in Minnesota. When available, case studies are included in the appropriate appendix. It is important to note that this review does not represent a comprehensive literature review on the prevention, control, or treatment of acid mine drainage, nor does it address the entire range of possible preventive technologies currently in use.

As a result of this literature review, it was determined that there is no single, correct approach to environmental mine waste management. Successful mine waste management and mitigation programs implement multiple strategies to address a wide array of site- and waste-specific variables. Integrated management systems, including contingencies to treat impacted waters, are often necessary to meet regulatory standards. Four examples of integrated environmental mine waste management approaches used in Minnesota, Wisconsin, and South Dakota are presented.

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## INTRODUCTION

### 1. Mining in Minnesota

Minnesota has an extensive mining industry and potential for mineral expansion and diversification. Iron mining began in Minnesota over a century ago and led to the taconite mining industry which, in 1996, shipped 45 million long tons of iron ore valued at 2.4 billion dollars (Minnesota Department of Revenue, 1997). Nonferrous mining development shows promise for the future. The state is presently the subject of extensive mineral exploration, with 59 leases covering more than 26,000 acres of state land (MN DNR, 1998).

Considerable base and precious metal mineral potential is associated with Minnesota's Precambrian rocks, specifically in Archean metavolcanics, metasedimentary formations and the Duluth Complex. The Archean metavolcanics and metasedimentary formations, or greenstone belts, of Minnesota extend north into Canada, where they have yielded substantial mineral production. These formations are potential hosts for gold, zinc-copper massive sulfides with various by-products, and magmatic sulfide deposits containing copper, nickel, and platinum group elements. Recent exploration of greenstone belt metasedimentary formations has focused on gold, base metals, and silver-cobalt-copper deposits. The Duluth Complex contains an estimated copper-nickel resource of 4.4 billion tons (Minnesota Environmental Quality Board, 1979), as well as significant titanium resources. Drill core analyses have also revealed the presence of chromium, vanadium, cobalt, and platinum group elements.

If mineral development occurs, tailings and waste rock, as well as the mine itself will be wastes remaining after the operation is abandoned. The potential for generation of acidic mine waste drainage is the primary water quality concern, and this potential is largely determined by the mine waste composition. Mine wastes capable of producing problematic drainage must be managed to ensure that the quality of Minnesota waters is not adversely impacted. Mine waste management strategies directed at this objective include prevention, control, and treatment of problematic drainage. However, implementation of mitigative strategies alone is not sufficient insurance of non problematic drainage. Empirical data on the long-term performance of most mine waste management strategies is limited. Consequently, surface and ground water monitoring must also be included in any environmental mine waste management plan.

#### 1.1. Mine Waste Drainage Mitigation in Minnesota

The Reclamation Section within the Division of Lands and Minerals, Minnesota Department of Natural Resources (MN DNR) has conducted numerous studies on environmental mine waste management (Lapakko et al., 1998a). Mitigative laboratory and small-scale field studies include the use of alkaline solids and subaqueous disposal of sulfidic waste rock. Operational-scale mitigative measures include capping and selective placement of reactive waste rock, drainage diversion, and passive and active treatment systems.

In addition to these studies, disposal of taconite tailings within existing open mine pits and associated surface and ground water quality have been investigated (Berndt et al., 1999; Lapakko and Jakel, 1999). However, taconite tailings contain very little sulfidic material and sufficient carbonate minerals to prevent acidic drainage in waters associated with them. Thus, these studies are beyond the scope of this presentation and will not be mentioned here.

#### 1.1.1. MN DNR laboratory and small-scale field studies

A two-phase, laboratory and field program examined the feasibility of removing trace metals (Cu, Ni, Co, Zn) from stockpile drainage using readily available materials (peat, till, wood chips, Cu-Ni tailings, and zeolite) in low-cost, low-maintenance systems at the AMAX/Kennecott site. The results of the entire program are presented in two reports (Lapakko et al., 1986a,b), and synopses of various research segments are presented in several symposium proceedings (Lapakko and Eger 1981, 1983, 1988; Lapakko et al, 1983; Eger et al., 1984).

Laboratory experiments were conducted to examine the mitigative potential of mixing alkaline solids with acid-producing mine waste. In one of these laboratory experiments, rotary kiln fines (RK fines; a waste product generated by the conversion of limestone to lime), ! 10 mesh limestone, and +10 mesh/-0.25 inch limestone were each mixed with finely crushed Duluth Complex rock ( $0.053 < d \leq 0.149$  mm) to examine their effectiveness in reducing the release of acid and trace metals in drainage from the rock (Lapakko and Antonson, 1989a, 1990a, 1991; Lapakko et al., 1997, 2000).

Laboratory and small-scale field tests were conducted to determine the effectiveness of limestone beds for treating mine waste drainage. In the laboratory, columns containing +10 mesh/ -0.25 inch high calcium limestone were used to treat three problematic mine drainages, one from a stockpile at the Dunka mine site and two from field test piles at the AMAX/Kennecott site (Lapakko and Antonson, 1989b, 1990a,b). Based on the results from the laboratory study, a field-scale limestone bed (1.4 m<sup>3</sup> bed volume) was constructed to treat one of the problematic drainages at the Dunka site (Lapakko and Antonson, 1989c, 1990c).

Finally, disposal of sulfidic rock in a subaqueous setting with and without various layers to inhibit oxygen transport (unmodified subaqueous, composted yard waste, taconite tailings, compost and taconite tailings, and limestone) is under investigation in laboratory experiments. Taconite tailings, tailings and compost, and limestone barriers are also being examined in field experiments (MN DNR, unpublished data).

Although these laboratory and small-scale field tests show promise as mitigative techniques for reactive mine wastes in Minnesota, it is important to note that, at this time, none of them have been tested at an operational-scale. Furthermore, each of these tests was conducted on a single rock type. Consequently, conclusions related to the application of these techniques to specific, large-scale, environmental mine waste problems in Minnesota should be regarded cautiously.

#### 1.1.2. MN DNR operational-scale field studies

Operational-scale field studies of passive wetland and active treatment systems have been implemented at the Dunka mine site. Based on the observation that metals were removed from waste rock stockpile drainage as it flowed through a white cedar swamp (Eger et al., 1980; Eger and Lapakko, 1988), studies were conducted to evaluate the use of wetlands as a technique for mitigating mine waste drainage. Literature was reviewed to summarize reported mechanisms and capacities for metal removal by peat (as well as other materials, Lapakko et al., 1985, 1986a, b) and laboratory experiments were conducted with native peat and waste rock drainage to more accurately assess the reactions applicable to situations in Minnesota (Lapakko and Eger, 1983). These results were subsequently used to estimate metal removal by peat under field conditions in Minnesota (Lapakko and Eger, 1988).

Based on the laboratory results, as well as observed metal removal by peat in a white cedar swamp and other natural wetlands, field test plots were designed (Eger and Lapakko, 1989) and metal removal in these systems was quantified (Eger et al., 1994). Based partly on these studies, wetland treatment systems were designed to treat drainage from operational-scale stockpiles (Eger et al., 1996, 1997, 2000). In addition to the wetland treatment system, the stockpiles were covered with various dry cover systems to reduce flow to these treatment systems (STS Consultants Ltd., 1993, 1994a, 1994b). An active lime precipitation treatment plant treats drainage from several of the aforementioned systems.

Sulfate reduction is an additional mechanism of metal removal (and pH elevation) in wetlands. Field experiments were initiated in 1990 to determine the efficiency, rates, and capacities of this mitigative approach (Eger, 1992, 1994a). In 1993, the field phase of this experiment was completed, although subsequent laboratory experiments have continued since 1995 (Eger and Wagner, 1995, 2001).

## 1.2. Objectives and Scope

Our goal is to further evaluate current methods and identify additional strategies for environmental mine waste management that show potential for application in Minnesota. For the purposes of this presentation, “environmental mine waste management” refers to mitigative methods used to prevent, control, and treat acidic and/or metal-laden drainage from sulfidic mine waste materials.

A literature search of the Reclamation Unit’s literature database for the keyword string “mitigation or treatment” yielded more than 900 citations (MN DNR, Saint Paul, MN; Appendix 1.1). Approximately 400 additional citations were found in proceedings of mine waste management conferences that took place between 1994 and 2000 and other sources not yet entered into the database (Appendix 1.2). It is important to note that the list of citations in Appendix 1 is largely comprised of references that were on hand in the Saint Paul office of the MN DNR. It does not represent an exhaustive literature review of mine waste mitigation strategies.

Due to the large number of citations found during the initial literature search, it was necessary to refine the scope of this document. This was accomplished by reviewing similar efforts in Canada, Australia, the United States, and Sweden to describe the current state of environmental mine waste management technology. These national efforts have resulted in several published compilations of mitigative strategies for waste management issues in metal mining (Feasby et al., 1997; MISTRA, 1998; Parker and Robertson, 1999; Gusek et al., in progress) and coal mining (Skousen and Ziemkiewicz, 1995; PA DEP, 1998; Skousen et al., 1998). Each compilation represents a comprehensive summary of mitigative approaches relevant to site-specific conditions. Our objective is to analyze these strategies to determine their applicability to environmentally sound mine waste management in Minnesota.

### 1.2.1 United States

The US Environmental Protection Agency (US EPA) has actively reviewed various active treatment options for acidic drainage from reactive mine wastes (US EPA, 1983). In 1983, the US EPA published a design manual for acid mine drainage neutralization that outlined procedures, advantages and disadvantages for different methods, and cost estimates for a variety of treatment options (US EPA, 1983). This design manual also includes information on reverse osmosis and ion exchange treatment. More recent research in the area of environmental mine waste management includes advances in active and passive treatment systems, source control, biological barriers, and engineered cover systems (US EPA, 1999).

In 1995, West Virginia University and the National Mine Land Reclamation Center published a compilation of articles on the prevention and treatment of acid mine drainage related to coal mining (Skousen and Ziemkiewicz, 1995). These articles covered a range of mitigative strategies with particular focus on the addition of alkaline materials to reactive mine wastes, neutralization of acidic drainage, and passive treatment systems for acidic drainage. Three years later, a similar publication was produced by the Acid Drainage Technology Initiative (ADTI)-Avoidance and Remediation Working Group (Skousen et al., 1998; Hornberger et al., 2000). However, this document was considered Phase 1, and summarized various mitigative and treatment technologies considered to be appropriate for specific drainage problems. Most of these strategies also appeared in Skousen and Ziemkiewicz (1995) with the addition of more active and passive treatment methods, bioremediation, and mineral surface treatment techniques.

The ADTI-Metal Mining Sector, a coalition of government agencies, industry, academia, and consultants, has been investigating metal mining environmental mine waste management issues since 1996 (Hornberger et al., 2000). One goal of the Metal Mining Sector of ADTI is to produce a technology handbook that describes the design, performance, applicability, and limitations of various mitigative strategies. This workbook will include information on sampling, monitoring, prediction, mitigation, and modeling of metal mining drainage problems (Hornberger et al., 2000; Gusek et al., in progress).

A state-wide effort by the Pennsylvania Department of Environmental Protection (PA DEP) and other governmental agencies, academia, and industry resulted in a compilation of current

strategies to predict, prevent, and control acidic drainage from coal wastes (PA DEP, 1998). Highlighted strategies include reclamation and vegetation, including bactericide technology, addition of alkaline materials to mine wastes, special handling techniques for mine wastes and drainage, and remining.

### 1.2.2 Canada

The Mine Environment Neutral Drainage (MEND) program, a cooperative effort between the Canadian government and industry, has focused on the prevention and control of acid mine drainage since 1998 (Tremblay, 2000; Tremblay and Weatherell, 2000; MEND, 2001). MEND has identified water covers as the “best prevention technology” against the generation of acidic drainage from unoxidized sulfidic wastes (Feasby et al., 1997; Tremblay, 2000). In areas where water covers are not suitable, MEND has investigated the use of various innovative dry cover designs and materials, disposal of mine wastes in cold (permafrost) environments, and surrounding reactive mine wastes with porous material to control ground water flow. Passive treatment technologies such as constructed wetlands and anoxic limestone drains were reported to have limited, site-specific applicability. However, newer passive technologies involving permeable sulfate reduction barriers and the biosulfide process were mentioned as promising (Feasby et al., 1997).

### 1.2.3 Australia

The Australian Minerals and Energy Environment Foundation (AMEEF) was established in 1991 to “promote the implementation of the principles of sustainable development in Australia’s mineral, energy and related industries.” This is a non-profit organization supported by grants from industry and government. A recent publication summarized the current state-of-knowledge on environmental mine waste management in Australia and around the world (Parker and Robertson, 1999). While this review covered a wide range of mitigative strategies, innovative cover systems, mineral surface treatments, porous surroundings, and long-term passive treatment systems were specifically identified as requiring additional investigation.

Several other Australian organizations concerned with environmental mine waste management exist. The Environmental Division of the Australian Nuclear Science and Technology Organisation (ANSTO) conducts project-oriented, applied research in many areas of environmental impact, including mine waste treatment and disposal. ANSTO’s Managing Mine Wastes Project focuses primarily on sulfidic mine waste and acidic drainage management through laboratory and field measurements, hydrologic and geochemical modeling, and ecological risk assessment (ANSTO, 1999; Bennett, 2000).

The Commonwealth Scientific and Industrial Research Organisation’s (CSIRO) is a government research organization that is involved with environmental research on a wide range of topics (Bennett, 2000). Research related to environmental mine waste management tends to be focused in the areas of acid drainage, sulfidic mine waste management, innovative capping techniques for tailings storage facilities, and mine waste characterization (CSIRO, 2000).

The Australian Mineral Industries Research Association (AMIRA) coordinates efforts by researchers and industry to develop proposals, identify sponsors, and manage research projects. The main area of focus is mineral processing. However, some of these projects involve environmental mine waste management (AMIRA, 2000; Bennett, 2000; Greenhill, 2000).

The Australian Centre for Mining Environmental Research (ACMER) was established as an industry supported organization with the goals of conducting environmental research and disseminating information throughout the mining industry. Several research organizations, including ANSTO, have been involved in the research aspects of their work (ACMER, 1998; Bennett, 2000).

#### 1.2.4 Sweden

In Sweden, the Mitigation of the Environmental Impact from Mining Waste (MiMi), has also initiated a coordinated effort to improve the economic efficiency of mine waste management (MISTRA, 1998; Hoglund, 2000). The multidisciplinary MiMi program goals include using predictive modeling to identify existing mine waste deposits that can be reclaimed using simple, cost-efficient methods. The MiMi program is also developing strategies that will prevent future environmental problems associated with reactive mine wastes. To this end, five research projects were initiated during the first program period from 1999 to 2000. These projects will address dry and wet cover systems, biotic barriers (i.e. shallow wetlands in tailings ponds), passive leachate treatment systems, permeable reactive walls, and co-deposition of tailings and waste rock (MISTRA, 1998).

#### 1.2.5 International

In October 1998, the International Network for Acid Prevention (INAP) was officially launched (Brehaut, 2000; INAP, 2000). This international movement was initiated after the success of the Canadian MEND program led to the realization that this level of technology transfer on an international scale was important. INAP consists of an international committee with representatives from industry, academia, and government from Australia, Canada, and the United States. INAP's objectives are "to promote significant improvements in the management of sulfidic mine materials and the reduction of liability associated with acid drainage through knowledge sharing and research and development of technology" (INAP, 2000).

At the present time, INAP has developed a web site, which provides access to information on professionals working on acid drainage issues, INAP research projects and proposals, case studies on the implementation of new techniques, and a forum for online discussion (Brehaut, 2000; INAP, 2000). Research areas specifically mentioned in INAP media releases include wet and dry cover systems for reactive mine wastes and passivation techniques to prevent the oxidation of sulfide minerals present in waste materials.

### 1.3 Structure of this Document

Thirty six different strategies for the environmentally sound management of reactive mine wastes were identified in aforementioned compilations (Appendix 1.3). Six of these strategies were referred to in more than one compilation. These strategies were:

- layered barriers or water cover systems to inhibit oxygen and/or water transport,
- alkaline amendments for in situ acid neutralization,
- microencapsulation to inhibit sulfide mineral reactivity,
- bactericides to inhibit biological sulfide oxidation in mine wastes,
- passive treatment of mine waste drainage, and
- active treatment of mine waste drainage.

These six strategies were used as a starting point for this project. Additional strategies have been included to provide a more complete review of the current state-of-technology of environmental mine waste management. For the organizational purposes of this document, they have been divided into three generalized sections: prevention, control, and treatment (Figure 1).

The first section deals with strategies intended to **prevent** the generation of problematic drainages. This is generally accomplished by physically isolating reactive mine wastes from the surrounding environment. Topics covered in this section include backfilling mine wastes into existing mine voids (section 3.1), co-disposal of fine and coarse-grained waste materials (section 3.2), and cover systems (section 4).

Strategies that **control** problematic drainage in situ are dealt with in the second section. In general, control strategies are used to stabilize reactive mine wastes either chemically or biologically. Addition of alkaline amendments (section 5.1) and microencapsulation (section 5.2) are two chemical stabilization strategies presented. The use of bactericides to inhibit biological sulfide oxidation represents biological stabilization (section 6).

Finally, existing problematic drainages generally require **treatment**. Both passive and active treatment options have been applied to problematic mine drainages. Passive treatment systems often utilize a combination of constructed wetlands, anoxic limestone drains, and settling/aeration ponds to treat surface waters and seeps (section 7.1). More recently, permeable reactive barriers have been developed to treat ground waters in situ (section 7.2). In some cases, an active treatment system may be the only option to meet regulatory standards. Numerous

active treatment systems have been developed. Acid neutralization (section 8.1), chemical precipitation (section 8.2), ion exchange (8.3), membrane filtration (8.4), and biological sulfate reduction (8.5) will be presented in this document.

Selected literature will be reviewed in terms of 1) a brief description of the mitigation strategy; 2) methods used to assess the strategy performance; 3) generalized cost analysis; and 4) applicability of the strategy to environmental mine waste management issues in Minnesota. When available, case studies of the use of each strategy are included in the appendices. It is important to note that this review does not represent a comprehensive literature review on the prevention, control, or treatment of acid mine drainage, nor does it address the entire range of possible preventative technologies currently in use.

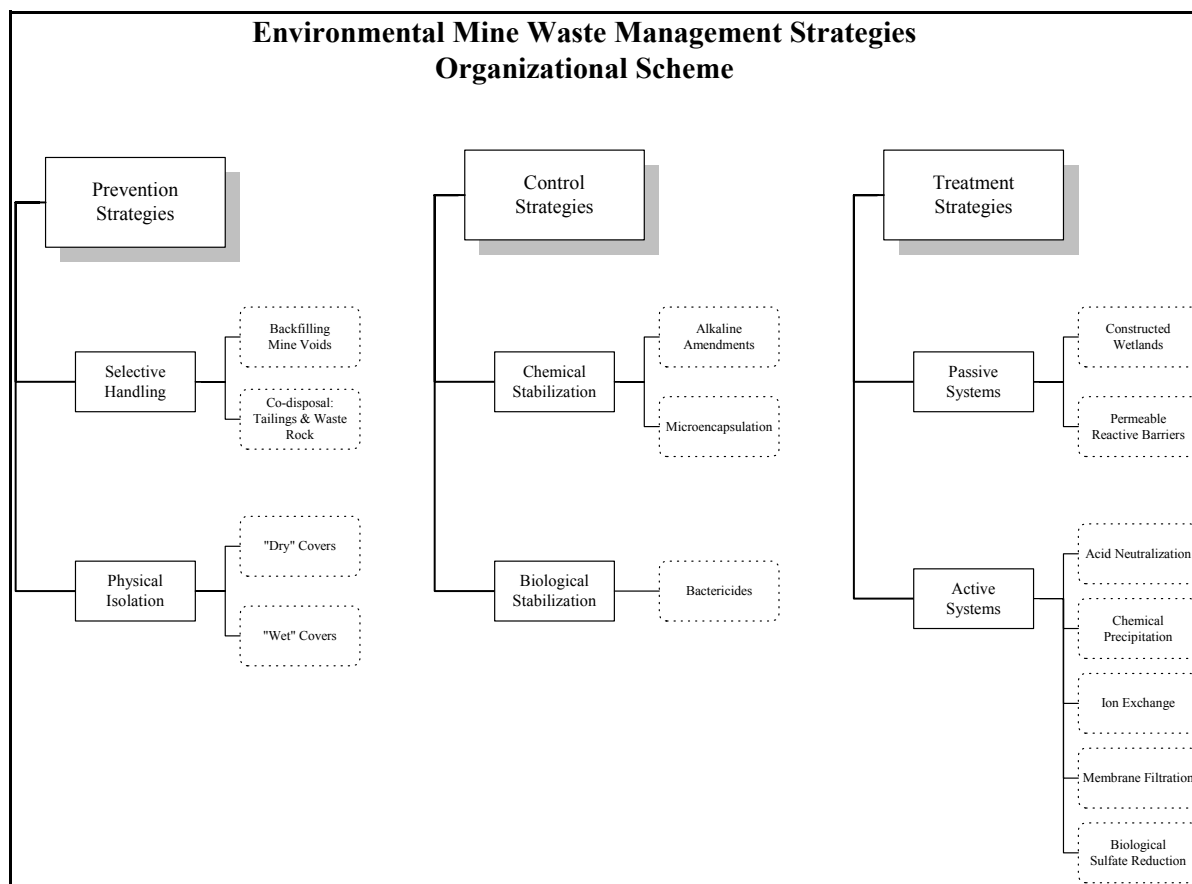


Figure 1. Environmental mine waste management strategies addressed in this document are grouped based on their general objective: prevention, control, or treatment of problematic drainages from reactive mine wastes.

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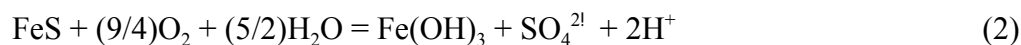
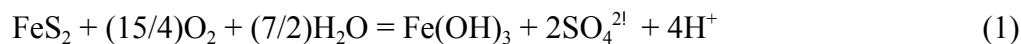
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## 2. Geochemical Background

### 2.1. Mine Waste Dissolution and Acid Mine Drainage

The dissolution of iron sulfide minerals such as pyrite and pyrrhotite is responsible for the majority of mine waste acid production (Stumm and Morgan, 1981). Equations 1 and 2 are commonly published reactions representing pyrite and pyrrhotite oxidation by oxygen (Nelson, 1978; Stumm and Morgan, 1981).



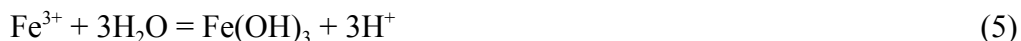
The rate of oxidation and attendant acid production is dependent on solid-phase compositional variables. Oxidation rates vary among sulfide minerals, and it is often reported that reactivity decreases in the order marcasite > pyrrhotite > pyrite (e.g. Kwong and Ferguson 1990). For a given sulfide mineral, the oxidation rate increases with the reactive surface area available. It also varies with the crystal form of the mineral. For example, the oxidation rate of framboidal pyrite is reported to be much more rapid than that of euhedral pyrite (Pugh et al. 1984; White and Jeffers 1994).

The rate of sulfide mineral oxidation also increases as pH decreases into a range conducive to bacterial catalysis of ferrous iron oxidation. Nordstrom (1982) reported that as “pH decreases to 4.5, ferric iron becomes more soluble and begins to act as an oxidizing agent.” As pH further decreases bacterial oxidation of ferrous iron becomes the rate limiting step in the oxidation of pyrite by ferric iron (Singer and Stumm, 1970), which is the only significant oxidizing agent in this pH range (Singer and Stumm, 1970; Kleinmann et al., 1981; Nordstrom, 1982).

These weathering reactions produce acidic, iron- and sulfate-rich aqueous water which can 1) contact sulfide minerals and accelerate their oxidation, 2) evaporate partially or totally to precipitate hydrated iron sulfate and other minerals, and/or 3) contact host rock minerals which react to neutralize some or all of the acid. Acidic flow which migrates through the mine waste may eventually exit as acid rock drainage.

Hydrated iron sulfate minerals precipitate during the evaporation of acidic, iron- and sulfate-rich water within mine waste materials and store (for potential subsequent release) acid generated by iron sulfide mineral oxidation. The more common hydrated iron sulfate minerals that occur as efflorescent salts on the surfaces of weathering pyrite include melanterite, rozenite, szomolnokite, romerite, and copiapite (Alpers et al., 1994). According to Nordstrom (1982) and Cravotta (1994), these sulfate salts are highly soluble and provide an instantaneous source of acidic water upon dissolution and hydrolysis. They are partially responsible for increased acidity and metals loadings in the receiving environment during rainstorm events.

As an example, equations 3, 4, and 5 summarize the step-wise dissolution of melanterite.



The net result of equations 3 through 5 is summarized in equation 6, which shows a net production of two moles of acid produced for each mole of melanterite dissolved.



Cravotta (1994) showed that a similar aqueous dissolution of romerite produced six moles of acid for each mole of romerite dissolved. Their cumulative storage and incremental release may help explain the lag from mine waste placement to acid mine drainage formation, particularly in arid climates.

According to Nordstrom (1982), the formation of hydrated iron sulfates is an important intermediate step that precedes the precipitation of the more common iron minerals such as goethite and jarosite. Jarosite is slightly soluble (Alpers et al., 1994) and can, therefore, contribute acid according to equation 7.



For example, recent preliminary leach studies on natural and synthetic jarosites conducted by the United States Bureau of Mines showed a drop in pH from 6 in the deionized water leachant to pH 3 or 4 after contact with the jarosites. Because of its relatively low solubility, the acid contributed by jarosite dissolution is probably small relative to that by dissolution of more soluble hydrated iron sulfates.

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## PREVENTION STRATEGIES

Whether or not a mine waste produces problematic drainage depends on a wide array of variables that describe the physical, chemical, and mineralogical characteristics of the waste, pre-existing site conditions, climate, and government regulations. Consequently, there is no single, correct approach to environmental mine waste management. One approach is to use strategies that prevent the generation of problematic drainages in the first place. Typical prevention strategies involve constructing a barrier between reactive mine wastes and air (i.e. oxygen) and/or water. This section will review the use of backfill options and cover systems commonly used to physically isolate reactive mine waste materials from the surrounding environment.

### 3. Selective Handling of Reactive Mine Wastes

#### 3.1. Backfilling Reactive Mine Wastes

##### 3.1.1. Objectives of Backfilling Reactive Mine Wastes

Backfilling existing mine voids with reactive mine waste materials is a prevention strategy currently practiced at both surface and underground mines. In terms of environmental mine waste management, the goal of backfilling is to isolate reactive mine wastes from the atmosphere and/or water. Additional benefits to the mining operation include increased stability of underground mine workings and reduced land development, disposal, and reclamation costs.

Numerous techniques for backfilling mine voids have evolved over the years. In a situation where an open pit is to be backfilled, transporting mine wastes from a temporary storage area may simply involve truck hauling or pumping a tailings slurry. Typically, reactive materials are placed near the center of the pit. The surrounding material usually consists of less reactive mine waste that limit exposure to water and oxygen. Highly reactive mine wastes often require additional mitigative measures. Ideally, the reactive mine waste can be encapsulated with net neutralizing mine wastes. However, if these materials are not available locally, alkaline materials such as limestone or fly ash can be intermixed with the reactive mine waste to neutralize any acid generated.

Prior to mine closure, backfilled pits must be reclaimed. Reclamation can vary from a simple replacement of topsoil and vegetation to complex multi-layer cover schemes (see section 4.2). Alternatively, if the pit intersects the ground water table, it may be preferable to inundate the waste materials. A major benefit of establishing a water cover is that oxygen diffusion will be limited, reducing the potential for acid generation (see section 4.3).

Occasionally, underground mines can be backfilled using ordinary haul trucks and bulldozers. However, more complicated methods, such as hydraulic flushing or pneumatic stowing are frequently required. Hydraulic flushing involves pumping fill material as a slurry into the void. This method has several drawbacks. Operators have very little control over the direction and

compaction of the backfill that results. Furthermore, the particle size distribution of the fill material varies with respect to slurry velocity during deposition. Finally, this method requires transport of relatively large volumes of water into the mine, which may or may not be beneficial (Walker, 1993).

Pneumatic stowing is a second method for backfilling mine voids, which eliminates the volume of water introduced into the mine. While a greater degree of compaction can be achieved using pneumatic stowing, abrasion of the injection nozzle and elbows leads to rapid equipment failure (Walker, 1993). This problem has been addressed, in part, by the development of a high efficiency ejector that uses a supersonic airstream to project fill material horizontally into the void (Burnett et al., 1995).

Handling of mine wastes, particularly tailings, can often present problems. A common goal is to create a high density fill material using paste technology. Pastes are mixtures of tailings and water, similar to a slurry. However, pastes are differentiated from slurries by the fact that they do not segregate with respect to particle size when at rest. In order for this to occur, a paste must contain approximately 15% fines ( $< 20 \mu\text{m}$ ) by weight. The fines retain moisture due to their high surface tension, preventing segregation (Cincilla et al., 1997). Portland cement can be added to the paste to increase strength and durability (Cincilla et al., 1997).

Similar to the open pit situation, alkaline materials (e.g. limestone, fly ash etc.) are often used to increase backfill strength and neutralize acidic drainage. However, laboratory studies have demonstrated the preferential dissolution of alkaline binders in cemented paste backfills (Bertrand et al., 2000). These results also indicated that the backfill material became passivated, although the exact mechanism of passivation was unclear. Consequently, cemented paste backfills may not provide long-term buffering against acidic drainage generated within backfilled mine works.

### 3.1.2. Monitoring Water Quality Associated with Backfilled Mine Works

Post-closure monitoring must be included in the disposal design in order to determine the overall effectiveness of a backfill program. Objectives are likely to include monitoring the movement of water into and out of the backfill as well as water quality. If poor water quality is anticipated, a liner and underdrain system should be installed prior to construction. These are necessary to collect drainage for additional treatment if needed.

Monitoring instrumentation may include a series probes and piezometers to measure gas concentrations (e.g. oxygen), moisture content, and water levels within the backfill. These instruments can easily be connected to an automated datalogger system to streamline data collection. Both ground and surface water quality should be monitored. Lysimeters and wells can be installed in strategic locations to obtain ground water quality samples. Seeps and ponded water, in the case of flooded wastes, also give an indication of the extent to which backfill materials may have reacted with ground and surface waters.

### 3.1.3. Economic Benefits of Backfilling Mine Voids

Backfilling mine voids with reactive mine waste materials can be a cost effective approach to environmental mine waste management. This is particularly true if abandoned pits or mine workings are available on-site. If mine wastes are to be redeposited in an operational mine, a temporary disposal plan will be required. In this situation, mine wastes are effectively disposed of twice (once as they are removed from the mine and again to their final disposal location), which increases design, transportation, handling, and management costs. However, these costs can be offset by the savings due to reductions in storage volume required. As with any disposal situation, the economic impacts of each variable must be evaluated on a site-specific basis.

### 3.1.4. Potential Application of Backfill Technology in Minnesota

Numerous mined out natural iron ore pits exist along the Cuyuna and Mesabi Iron Ranges in northern Minnesota. If mineral development were to occur in these areas, backfilling these pits with reactive mine wastes may appear to be a feasible option. However, potential impacts to aquifers (i.e. local drinking water supplies) must be considered. According to the Laws of Minnesota, 1996, Chapter 407, Section 56, the Minnesota Pollution Control Agency may issue a permit to dispose of fine taconite tailings into taconite mine pits provided, “the proposer demonstrates through an environmental impact statement and risk assessment that the deposition will not pose an unreasonable risk of pollution or degradation of groundwater.” It should be noted that this law refers specifically to taconite mine wastes, which generally do not produce problematic drainage. Sulfide-bearing mine wastes may have a higher probability of impacting local ground waters due to their reactivity and the likelihood that mine workings will intersect local aquifers. Whether or not backfilling open pits with reactive materials is a practicable mine waste management option in Minnesota will largely depend on the solid-phase characteristics of the mine waste material and the hydrogeology of the open pit and surrounding area. Consequently, practicability of this strategy is likely to be determined by site-specific evaluations.

## 3.2. Co-disposal of Tailings and Waste Rock as a Preventative Strategy

### 3.2.1. Objectives of Co-disposal

Traditional handling of mining wastes calls for separate disposal of coarse waste rock and fine-grained tailings. However, recent investigations into the co-disposal of tailings and waste rock suggest that this strategy has both environmental and economic benefits. Co-disposal simply refers to the combining of tailings and waste rock prior to disposal. Layered co-disposal involves placing layers of tailings at predefined intervals within a waste rock deposit during construction. In both cases, the resulting deposit has improved physical and chemical characteristics that decrease the potential for acid generation and trace metal release.

Generation of problematic drainages is the result of sulfide mineral oxidation, which is, in turn, a function of the amount of water and oxygen present in reactive mine wastes (equations 1 and

2). The primary goal of co-disposal is to exploit the hydraulic properties of fine-grained materials (i.e. tailings) to limit exposure of mine waste materials to oxygen. This is accomplished by mixing tailings and waste rock prior to disposal. The high moisture-retention capacity of the tailings creates saturated conditions throughout the deposit. Since oxygen diffuses more slowly through water than air (Table 8), the rate of sulfide oxidation is minimized. Additional benefits of co-disposal include:

- short-term control as reactive wastes are disposed,
- increased physical stability of the deposit due to waste rock sheer strength,
- increased density within the deposit,
- increased water recovery from tailings,
- relatively high surface area associated with fine-grained material increases potential for adsorptive trace metal removal,
- certain fine-grained waste materials (e.g. taconite tailings) may contain acid neutralizing minerals, and
- reduced land development, disposal, and reclamation costs.

Despite these advantages, co-disposal of tailings and waste rock may involve considerable planning and design. Both pumped and layered co-disposal strategies involve departures from conventional waste handling and transportation techniques (Table 1). If the objective is an intermixed deposit of tailings and coarse reject, the two waste streams can be combined and pumped to an impoundment or abandoned pit. However, specialized equipment is needed to handle the additional strain on pumps and pipelines. Waste handling and transportation issues also arise for layered co-disposal situations. Compaction of the tailings layers maximizes the capillary effect. However, compaction requires dewatered tailings. Consequently, practical issues of how to transport dewatered tailings to the impoundment must be addressed. This usually involves truck hauling (Table 1). In general, co-disposal of tailings and waste rock will involve a greater effort in the areas of deposit design, disposal scheduling, and supervision.

### 3.2.2. Evaluation of Co-disposal Performance

Generation of problematic drainages in waste rock piles is directly related physical as well as chemical processes. Without a good conceptual model of the disposal system, performance is difficult to evaluate. Numerous efforts have been made to characterize the physical processes (i.e. water and oxygen flow) operating within waste rock piles and fine-grained materials (Nicholson et al. 1989; Lefebvre and Gelinas 1995; Nichol et al. 2000; Wilson et al. 2000b). Quantification of these processes is essential in determining mechanisms that control drainage chemistry. By understanding these mechanisms, they can be exploited to optimize waste rock deposit design and develop monitoring programs to evaluate overall performance.

Table 1. Several practical waste handling design and transportation issues must be addressed when considering a co-disposal strategy for mine waste management.

Co-disposal Description	Issue	Consider...
Pumped	pump & pipeline wear	<ol style="list-style-type: none"> <li>1. high-pressure, large bore pumps</li> <li>2. ceramic- or polyurethane-lined pipes</li> <li>3. particle shape</li> <li>4. decrease solids concentration</li> </ol>
	pipeline blockages	<ol style="list-style-type: none"> <li>1. increase flow velocity</li> <li>2. increase pressure</li> </ol>
	finer wash out of pore spaces among coarse particles	<ol style="list-style-type: none"> <li>1. high coarse to fine mass ratio</li> <li>2. small size gap between coarse and fine materials</li> <li>3. decrease flow velocity</li> </ol>
	hydraulic sorting upon discharge	<ol style="list-style-type: none"> <li>1. direct discharge up slope</li> </ol>
Layered	transportation of dewatered tailings	<ol style="list-style-type: none"> <li>1. truck hauling</li> </ol>

### 3.2.3. Economic Benefits of Co-disposal of Mine Wastes

Pumping, specialized heavy duty equipment, and/or transportation of dewatered tailings will increase the expense incurred by co-disposal of fine and coarse waste materials. However, these costs may be offset by the savings due to reductions in storage volume required. For example, a coal operation in Australia pumped tailings and coarse reject to an abandoned pit, resulting in a reported savings of \$0.52/ton of product (Williams, 1997). This estimate may be higher than can be expected at other sites due to the availability of an abandoned pit and site-specific problems associated with conventional waste disposal. An alternative estimate predicted a 20% reduction in operating costs (Sellgren and Addie, 1998). As with any disposal situation, the economic impacts of each variable must be evaluated on a site-specific basis (Table 2).

Table 2. Costs associated with co-disposal of tailings and waste rock may be reduced, depending on the relative savings compared to conventional disposal.

Conventional Disposal Cost Items	Economic Benefit/Liability of Co-Disposal
tailings thickener (capital, operation, maintenance)	A portion of the tailings stream may be able to bypass the thickener.
tailings impoundment construction	Increased density of the deposit reduces the size requirements.
acquiring land	Increased density of the deposit reduces the amount of land required.
pumps & pipeline (capital, operation, maintenance)	Considerably higher due to the need for heavy duty equipment.
water recovery	site-specific
reclamation	site-specific
coarse reject hopper	A hopper to combine tailings and coarse streams will still be necessary.
conveyor system (capital, operation, maintenance)	None required
truck (capital, operation, maintenance, replacement)	No truck hauling for pumped co-disposal. May be a factor for layered co-disposal.
haul roads (construction, maintenance)	No truck hauling for pumped co-disposal. May be a factor for layered co-disposal.
drainage management (collection, treatment)	site-specific

### 3.2.4. Co-disposal Studies and Applications in Minnesota

Several laboratory-scale investigations into the effectiveness of tailings and waste rock co-disposal have been conducted (Lapakko and Eger, 1981; Iwasaki et al., 1982; Eger et al., 1984; Lapakko et al., 1983, 1985, 1986a, b; Johnson et al., 1995; Fortin et al., 2000; Lamontagne et al., 2000). This discussion will focus on those using tailings and mine wastes from northern Minnesota.

#### 3.2.4.1. MN DNR Joint Disposal Studies

Laboratory and field studies examining the ability of taconite tailings to passively treat stockpile drainages were conducted by the MN DNR (Lapakko and Eger, 1981; Eger et al., 1984; Lapakko et al., 1983, 1985, 1986a, b). The two-phase experimental program was conducted under US Bureau of Mines funding to examine the feasibility of removing copper, nickel, cobalt, and zinc from stockpile drainage using readily available materials in low cost, low maintenance systems (Lapakko et al., 1986a,b). The first phase was a laboratory program using: 1) batch experiments as a screening program; and 2) column experiments to select the most promising materials for field trials (Lapakko and Eger, 1981; Lapakko et al., 1985, 1986a).

Low sulfide tailings were chosen for the phase two field testing, utilizing both treatment beds and joint disposal bins. Saturated and unsaturated flow tailings treatment beds demonstrated the greatest removal efficiency for copper and lowest for nickel (Lapakko et al., 1983, 1985, 1986a).

In the joint disposal study, total effluent from waste rock was reduced 34% by the addition of tailings and 66% by the addition of tailings followed by an additional cover layer of tailings and vegetation (Eger et al., 1984; Lapakko et al., 1985, 1986a). During the 1983 field season, both treatments produced effluent that met the copper guideline; the nickel concentration was also reduced, but it nevertheless exceeded the guideline by more than a factor of two (Table 3).

Furthermore, these successes were short-lived. After three years, metal concentrations had increased three to five fold and effluent pH had begun to decrease (MN DNR, 1987). Among the conclusions from the MN DNR joint disposal field study were the following general considerations for operational-scale co-disposal (Lapakko et al., 1985, 1986a):

- optimal tailings and waste rock incorporation methods to reduce the area required for overall waste disposal,
- correct positioning of the discharge pipe relative to the deposit to avoid hydraulic sorting,
- quantification of the hydraulic and metal removal capacities of the system,
- construction of toe ditch and berms, settling basins, or similar structures to minimize total suspended solids in the effluent,
- deposit reclamation to reduce erosion, and
- potential long-term trace metal release from the tailings.

Table 3. Flow-weighted concentrations from joint disposal bins compared to effluent guidelines (Lapakko et al., 1985).

Parameter	Guideline Limitation (mg/L)	Average flow-weighted concentration (mg/L)		
		Controls	Unvegetated Tailings	Vegetated Tailings
Cu	0.15	0.31	0.11	0.06
Ni	0.75	3.2	1.58	1.78
Co	0.15	0.11	0.06	0.05
Zn	0.75	0.23	0.07	0.15
pH	6.0-9.0	6.0	6.2	6.2
TSS	20.0	1.4	3200	280

Table 4. Calcite-bearing taconite tailings reduced the amount of nickel and copper release from Duluth Complex ore (Iwasaki et al., 1982).

Parameter	Ore	Ore + taconite tailings
pH <sup>1</sup>	6.3	6.6-6.7
Cu (mg/L) <sup>1</sup>	0.09	below detection
Ni <sup>1</sup>	3	0.95-1.8
SO <sub>4</sub> <sup>2</sup>	1500	900-1300

<sup>1</sup> 120 day leach test

<sup>2</sup> after 8 months

#### 3.2.4.2. Minnesota Mineral Research Center Studies Involving Co-disposal

The Minnesota Minerals Research Center also conducted laboratory studies in which the potential use of carbonate-bearing taconite tailings as acid neutralizing additives was investigated (Iwasaki, 1982). Shake flask leaching of Duluth Complex (copper-nickel) ore was conducted using 0.5%, 1.0%, and 2.0% taconite tailings. The results indicated that taconite tailings did provide some neutralization and decreased heavy metal release from the ore (Table 4).

#### 3.2.4.3. Full-Scale Application of Co-disposal in a Taconite Tailings Basin

Finally, there is one known, operational-scale case of waste rock disposal within a tailings basin in Minnesota. Upon closure of the Dunka Mine near Babbitt, MN, LTV Steel Mining Company disposed of approximately 262,000 long tons of high sulfur Virginia Formation hornfels waste rock (LTV, 1996). Hornfels and dolomitic limestone (ratio of 8.35:1) were encapsulated within a section of their taconite tailings basin. Monitoring wells were installed in 1995, however, subsequent mining activity damaged the two wells to the extent that they were no longer usable. Two additional wells were installed in the spring of 1998. Monitoring of water quality in these wells and local seeps continues on a quarterly basis (Table 5).

Table 5. Water chemistry measured from monitoring wells and seeps in LTV Steel Mining Company's tailings basin between March 1995 and February 2000. Concentrations in µg/L unless noted otherwise. See Appendix 2 for a more detailed summary (Clark, 2000).

Parameter	Wells (N=43)		Seeps (N=112)	
	Range	Average	Range	Average
pH	7.0 - 10.0	8.3	7.1 - 8.7 <sup>1</sup>	7.9
Cu	<1 - 125	13	<1 - 80.8	1.4
Ni	<1 - 243	19	<1 - 13.6	1.4
Co	<1 - 20.6	4	<1 - 7.8	1.0
Zn	<10 - 181	27	<10 - 20	6.4
SO <sub>4</sub> (mg/L)	54.5 - 825	153	132 - 531 <sup>1</sup>	210

<sup>1</sup> N = 84

Between 1996 and 1999, water samples were collected from a seep along the southwest side of the tailings basin as part of a MN DNR-University of Minnesota joint study (Berndt et al., 1999). High sulfate levels in these samples indicate transport of sulfate from the hornfels rock (Table 6). However, it cannot be determined whether the sulfate was the result of rinsing of oxidation products produced prior to placement of the hornfels rock in the tailings basin, or subsequent continued sulfide oxidation within the tailings basin. Seepage pH and trace metal levels were considered non-problematic. This suggested that the additional dolomite and taconite tailings (i.e. ankerite) contained adequate capacity for neutralization. Iron oxides present in the tailings (approximately 19 wt%) were believed to provide sites for trace metal adsorption (Berndt et al., 1999).

Table 6. Summary of water chemistry measured from a seep on the southwest side of LTV Steel Mining Company's tailings basin between June 1996 and February 1999. Appendix 2 for a more detailed summary. Concentrations in µg/L unless noted otherwise.

Parameter	Range	Average	N
pH	7.1-7.7	7.3	11
Cu	0.0-1.5	0.7	11
Ni	0.9-3.3	1.8	11
Co	1.6-5.1	2.6	11
Zn	5.0-30.2	17.3	11
SO <sub>4</sub> (mg/L)	169-284	230	11

In summary, results from laboratory studies suggest that co-disposal of tailings and waste rock can be used to mitigate problematic drainages (Lapakko and Eger, 1981; Iwasaki et al., 1982; Eger et al., 1984; Lapakko et al., 1983, 1985, 1986a, b; Johnson et al., 1995; Fortin et al., 2000; Lamontagne et al., 2000). However, these studies indicate that practical issues of waste deposit design, use of alkaline additives, transportation and discharge of wastes, reclamation, and long-term contaminant release present potential technical, economic, and environmental problems. Individual operations must resolve these issues (i.e. large-scale field tests) before co-disposal strategies can be considered a viable waste disposal strategy.

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#### 4. Physical Isolation of Reactive Mine Wastes

One approach to environmental mine waste management involves construction of a barrier between reactive mine wastes and air (i.e. oxygen), water, or both. This section will review the use of cover systems commonly used to physically isolate reactive mine waste materials. During the literature search for this document, it was discovered that each research group seemed to have developed their own terminology to describe physical isolation of reactive mine wastes. Consequently, it is important to define the terminology that will be used in this document. This section describes three general physical isolation methods. The first is a liner (section 4.1), which is defined as a material placed beneath mine wastes. The remaining two are “dry” and “water” cover systems. The dry cover systems in section 4.2 describe layers of solid-phase materials intentionally placed over mine wastes. In contrast, “water” cover systems (section 4.3) submerge mine waste materials.

##### 4.1. Liners

Although not technically a mitigative strategy, it is important to mention the use of high strength, low permeability liners in environmental mine waste management. Liners have been used in the mining industry for containment and reinforcement purposes for years. For the purposes of this document, a “liner” will be defined as a material placed beneath mine wastes or mitigative system component (e.g. constructed wetland cells are often lined, section 7.1). This is distinct from a “cover” placed over a reactive mine waste (sections 4.2).

The most commonly applied liners are either compacted clay (CCL) or a flexible synthetic membrane (i.e. HDPE or PVC). However, in recent years, geosynthetic clay liners (GCL) have been developed (see section 4.2.1; Miller and Hornaday, 1998). Improved leak protection and lower susceptibility to desiccation and freeze-thaw damage give GCLs a practical advantages over CCL and synthetic liners despite the higher cost.

The need to meet government regulations generally leads to the decision to incorporate a liners into mine waste disposal designs. Frequently, synthetic liners are used to divert water to desired locations (e.g. constructed wetlands). However, they may also be a component in a backup system designed to minimize impacts if the primary mitigative strategy fails. Many of the mine waste strategies described in this document have liners incorporated into their design, although they usually are not mentioned specifically.

## 4.2. Dry Cover Systems

This section reviews the use of dry cover systems to physically isolate reactive mine wastes from the surrounding environment, and in doing so, preventing the generation of problematic drainages. During the literature search for this document, it was discovered that a consistent terminology used to describe cover systems did not exist. Consequently, it is important to define the term “dry cover system.” Section 4.2 describes dry cover systems, in which solid-phase materials are systematically placed on top of reactive mine wastes. This may involve one or more layers of one or more materials (e.g. compacted clay, sand, top soil etc.). Some cover system layers may be designed to retain moisture within pore spaces, however, these are distinct from “water” covers in that the mine wastes are not submerged in water.

### 4.2.1. Objectives of a Dry Cover System

The amount of acidity and associated trace metals released due to oxidative sulfide dissolution is a function of the amount of water and oxygen present in reactive mine wastes. Furthermore, water is necessary to transport these reaction products from the mine wastes to the environment. The rate of sulfide oxidation can be minimized, by limiting exposure of mine waste materials to water and oxygen (equations 1 and 2). Dry cover systems are designed to achieve one or more of the following objectives: 1) minimize water infiltration into underlying mine wastes, 2) inhibit oxygen diffusion into reactive mine wastes, and 3) consume oxygen within a cover layer over reactive mine wastes. Each cover system will have an effect, to some degree, toward these objectives depending on the physical properties (permeability, grain and pore size, etc.) of the cover materials used.

In reality, there are as many dry cover system designs as there are dry cover applications. However, dry cover system designs generally fall into one of seven categories: soil covers, compacted clay covers, anisotropic barriers, capillary barriers, oxygen consuming barriers, synthetic covers, and self-sealing/self-healing barriers. Three main design components can be used to describe a dry cover system (Figure 2). Frequently, a distinct, support layer is incorporate into a dry cover system design. A barrier layer is then laid out over the support layer. Physical properties of barrier layer materials are utilized to inhibit water infiltration and/or oxygen diffusion. Certain materials (e.g. organic matter) may be used to consume oxygen within the barrier layer. Usually a protection layer is laid out over the barrier layer. Protection layers increase lateral drainage of surface runoff, minimize erosion, protect against damage caused by freeze-thaw cycles, and prevent biointrusion of the barrier layer. Frequently, an additional layer of topsoil is added as a support for vegetation, which increases evapotranspiration further reducing water infiltration into the underlying mine wastes. Each of the six dry cover system designs will be described in terms of the support layer, a barrier layer, and any protection layers incorporated into the design.

# DRY COVER SYSTEMS

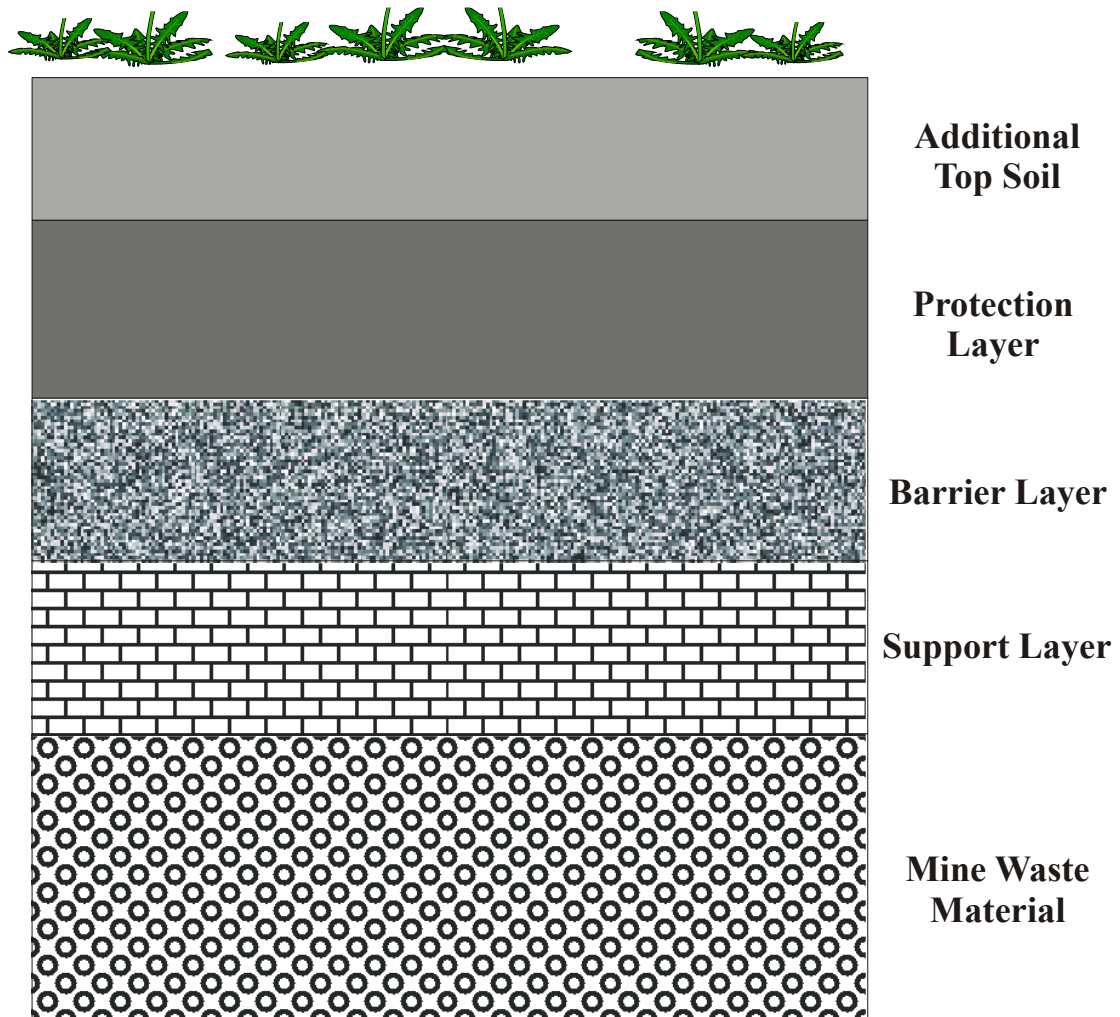


Figure 2. Schematic of a generalized dry cover system design for acid producing mine waste material.

#### 4.2.2. Dry Cover System Designs

The simplest and least expensive dry cover system to install is a **basic soil cover** (Figure 3). This cover design does not necessarily involve a support layer. Instead, a barrier layer of soil is deposited directly over the mine waste. The thickness of the soil layer depends upon the type of mine waste and site-specific requirements. While this design will reduce infiltration to the underlying mine waste, it will not eliminate it unless used in an arid climate. However, compaction of the soil barrier layer will further reduce infiltration. A protective layer of loose topsoil is often used with conventional soil covers. This layer supports vegetation as well as protects against erosion.

**Compacted clay cover** designs are similar to conventional soil covers. The main difference is that clay (e.g. bentonite) is mixed into the soil used for the barrier layer (Figure 3). The addition of clay to the barrier layer increases the cost and the complexity of this cover design. However, the higher clay content and compaction serve to minimize the hydraulic conductivity of the barrier layer, and consequently, limit downward movement of water to the underlying mine waste. Occasionally, a drainage layer of a relatively coarse grained material (e.g. sand) will be installed directly over the barrier layer. The drainage layer helps minimize infiltration into the barrier layer by enhancing lateral drainage away from the mine waste. Finally, a protection layer of loose soil is used to support vegetation and to prevent damage to the barrier layer caused by erosion or freeze-thaw cycles.

**Anisotropic barriers** utilize layers of capillary breaks to minimize vertical movement of water while maximizing horizontal drainage. Different physical properties and compaction techniques are used to maximize lateral drainage away from underlying mine wastes. One example of an anisotropic barrier consisted of four layers (Figure 3). A coarse-grained support layer of gravel is used to create a capillary break beneath an interface of fine sand. The fine sand interface increases lateral drainage of any water that percolated through the barrier layer. Native soils are typically used for the barrier layer which is designed to store water for subsequent evapotranspiration. The barrier layer is overlain by a soil and gravel protection layer which encourages evapotranspiration as well as protecting underlying layers and allowing vegetation growth.

**Capillary barrier** designs contrast hydraulic properties of cover materials to minimize downward migration of water by creating a capillary break between layers. The capillary break is achieved by placing a fine-grained (i.e. barrier) layer between two coarse-grained layers (i.e. support and protection layers; Figure 3). The capillary barrier itself consists of the support layer and the barrier layer. Coarse sand or gravel is typically used for the support layer. The support layer also has the added benefit of enhancing lateral drainage of any water that infiltrates the barrier layer. Fine sand, soil, clay, and inert tailings have been used as barrier layers in capillary barrier covers. The lower hydraulic conductivity of these materials relative to that of the support layer enhances moisture retention and inhibits oxygen diffusion across the capillary break. A protection layer of sand or gravel overlies the capillary barrier to encourage lateral drainage.

Once again, topsoil is usually used as the surface layer to support vegetation and minimize damage to the underlying active layers.

**Oxygen consuming barriers** utilize organic material as a barrier to oxygen diffusion into underlying mine wastes. If adequate organic material is present, the rate of oxygen consumption will exceed the rate of oxygen diffusion through the barrier layer, preventing oxygen from reacting with the underlying reactive wastes. The relatively low hydraulic conductivity of most organic materials coupled with increased compaction of the barrier layer as organic materials oxidize often results in the additional benefit of reduced infiltration of water to underlying wastes. Organic barrier layers are often applied as a single layer, simplifying the construction of the design (Figure 3). Furthermore, organic materials are usually locally available, reducing the cost of this cover design.

Impermeable cover systems can also be constructed of **synthetic materials** such as plastic. Oftentimes, synthetic cover materials provide a simple alternative to natural soil materials. Synthetic cover materials have also been incorporated into other cover system designs to maximize their effectiveness. A number of plastic (geosynthetic) materials are commercially available, most of which consist of flexible polymeric membranes (e.g. PVC or HDPE) that act as a barrier to water infiltration. Another option is a geosynthetic clay liners (GCLs), where sodium bentonite clay is encapsulated between two layers of geotextiles (Stewart and von Maubeuge, 1997; Miller and Hornaday, 1998).

Geosynthetic membrane installation is simple and can be accomplished year round. Installation merely involves unrolling the geosynthetic membrane out over the mine waste. Since they are relatively light weight, thin, and do not require compaction, geosynthetic membranes are relatively inexpensive to install. Geosynthetic membranes are extremely durable and resistant

to leaching, however, they are also thin and susceptible to damage (i.e. tears, leaks along seams, photodegradation), particularly during installation. Geosynthetic membranes and GCLs perform well under extreme conditions such as freeze-thaw cycles and dessication, but because they tend to deteriorate when exposed to sunlight, they are commonly covered with one or more protective layers of soil. A common configuration involves a geosynthetic membrane overlain by a drainage layer (e.g. sand) and a topsoil protective layer, although geosynthetic membranes have been used in combination with numerous natural cover materials (Figure 3).

**Impermeable hardpan barriers** represent a recent development in dry cover system design. This basic concept behind this cover system is that two parent materials, when placed in layers one on top of the other, will chemically react to form insoluble precipitates at the interface between the parent materials (Figure 3). Proposed parent materials include, but are not limited to, magnesium carbonate and high-calcium hydrated lime. The insoluble precipitates result in an impermeable seal that prevents migration of water and dissolved contaminants (i.e. self-sealing). The primary benefit of this type of cover system over the other options is that if the barrier is damaged, surrounding parent materials will react, self-healing the breach.

# DRY COVER SYSTEM DESIGNS

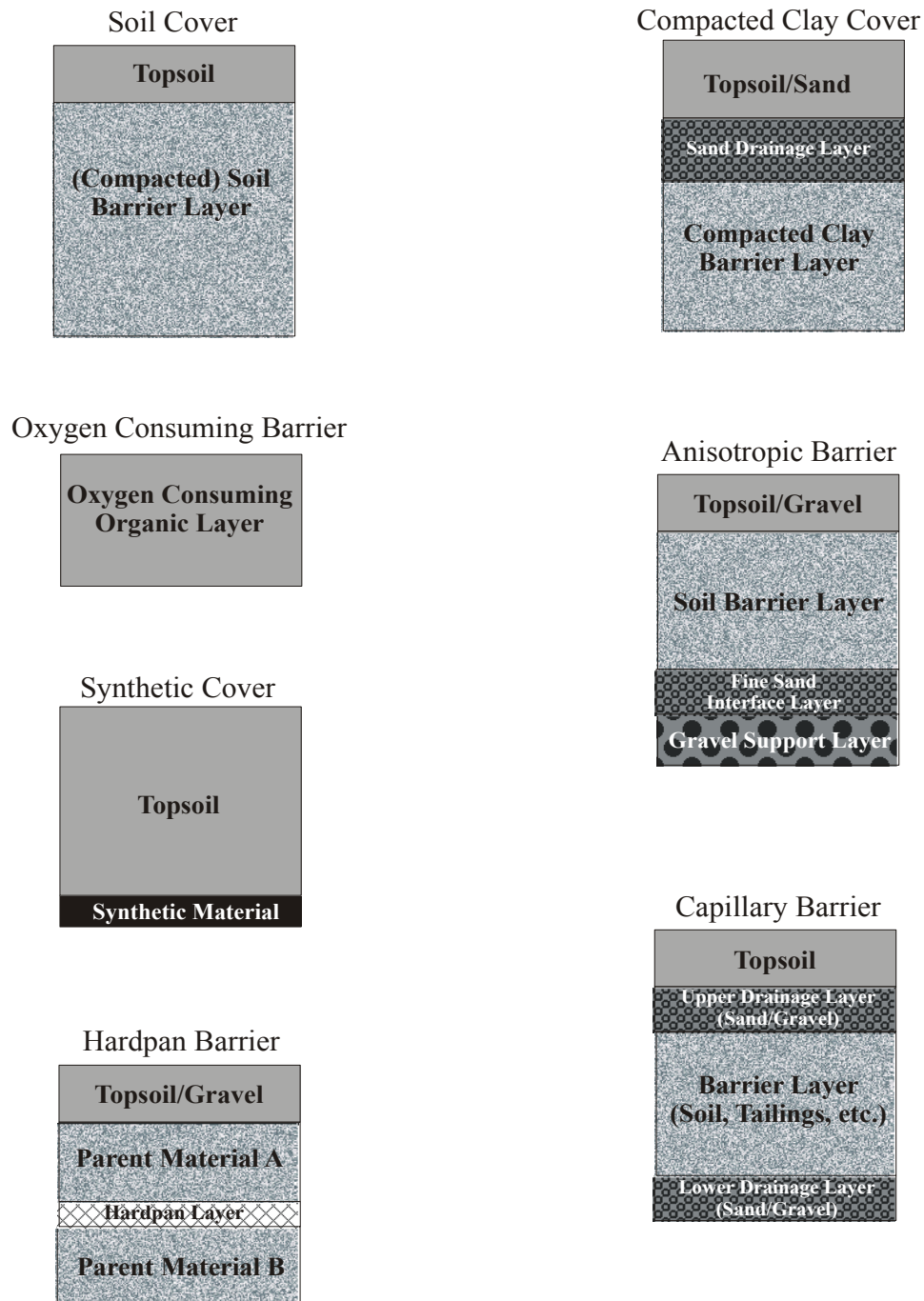


Figure 3. Dry cover designs generally fall into one of seven categories, each of which is designed to minimize water infiltration and/or oxygen diffusion into the mine waste.

### 4.2.3 Assessment of Dry Cover System Designs

#### 4.2.3.1. Variables Affecting Dry Cover System Design

The primary issue concerns the type of materials to be used in the dry cover system. Cover requirements are almost always site-specific, depending on variables such as the physical, chemical and mineralogical properties of the wastes, climatic conditions, local regulations, and availability of cover materials. Consequently, a cover system implemented at one mine site may not meet the objectives intended for similar reactive wastes at another site. The choice of cover materials is often based upon numerous variables including costs, ease of construction, and overall effectiveness.

#### 4.2.3.2. Predictive Modeling of Dry Cover System Design

How well a proposed dry cover system will perform is usually tested using predictive modeling, and occasionally, laboratory column studies. Predictive models are frequently used to predict the long-term effectiveness of a cover system, since long-term performance of a specific cover system cannot be determined prior to installation. Predictive models are typically used to meet one of the following objectives: determine ground water flow through a mine waste mass as well as over or around it, estimate surface water and precipitation infiltration through the cover system, and predict the water quality of drainage from the covered mine wastes. The use of models in this way is an inexpensive method to evaluate multiple cover systems and their long-term impacts on local surface and ground water quality in a very short amount of time.

#### 4.2.3.3. Materials Testing

Laboratory tests are typically used to determine the hydraulic and geotechnical properties of particular dry cover materials under consideration as well as how layers of multiple cover materials may interact to reduce infiltration and oxygen diffusion to the underlying waste. Hydraulic conductivity and porosity are the most frequently determined properties. However,  $D_{10}$ , specific gravity, plasticity index, and numerous other parameters have been determined as well. Occasionally, the same parameters will be determined for the mine waste, particularly if the mine waste is tailings rather than waste rock. The most thorough studies also determined the physical, chemical, and mineralogical properties of potential cover materials and/or the mine waste.

#### 4.2.3.4. Effectiveness of a Dry Cover System Design

The effectiveness of a cover system is typically measured in terms of oxygen concentrations and water content profiles throughout and beneath the cover. Column experiments are designed with instrumentation to measure temperature and pressure changes, water content and oxygen concentrations at regular intervals in the column. Since sulfide oxidation is an exothermic reaction, temperature measurements provide a qualitative indication of the extent to which oxidation had occurred in the mine waste. Pressure changes were measured using tensiometers

connected to pressure transducers, while moisture content was measured by Time Domain Reflectometry (TDR) electrodes. Oxygen sensors placed at regular intervals down the length of the column measured oxygen concentrations at various depths. These were used to construct oxygen profiles and gradients ( $\delta C/\delta z$ ), which in conjunction with the effective diffusion coefficient ( $D_e$ ), were used to determine the oxygen flux for the cover system.

#### 4.2.4. Cost Analysis of Dry Cover Systems

In general, the expense associated with each type of cover system increases as the cover system becomes more complex (Table 7). If cover materials require compaction or other special treatment, construction costs increase. However, using locally available cover materials such as glacial till or non-reactive mine wastes can substantially decrease costs.

Table 7. Summary of cover construction costs by type of cover. The estimated price range generally increases as the complexity of the cover system increases.

Type of Cover System	Estimated Price Range (US\$/hectare <sup>1</sup> )
single layer of clay or clayey till	\$9 - 62 k <sup>2</sup>
GCL	\$17 k
single layer of soil	\$ 15 - 25 k <sup>2</sup>
PVC	\$36 - 60 k <sup>3</sup>
capillary barrier (non-reactive mine wastes)	\$59-70 k
capillary barrier (general)	\$50 - 300 k

<sup>1</sup>No attempt was made to adjust for inflation or fluctuating exchange rates.

<sup>2</sup>The higher end of this range includes compaction costs.

<sup>3</sup>Range varies depending on the required thickness.

#### 4.2.5. Potential for the Use of Dry Cover Systems for Reactive Mine Wastes in Minnesota

##### 4.2.5.1. Dry Cover Systems Used in Minnesota

Dry cover systems have been shown to limit water infiltration into reactive mine wastes in Minnesota. Simple soil and synthetic cover materials were applied to stockpiled reactive mine wastes at LTV Steel Mining Company's Dunka Mine near Babbitt, Minnesota (Eger and Lapakko, 1985; Udoh, 1993). These systems were designed to minimize the amount of drainage from the stockpiles rather than prevent drainage acidification.

Six test stockpiles containing 820 to 1300 metric tons of sulfidic mine waste material were constructed in 1977 at the AMAX/Kennecott site (Eger and Lapakko, 1985). Three of the stockpiles remained exposed to the atmosphere as controls, and the other three were covered with 18-29 cm of soil obtained from a nearby borrow pit in 1978. Stockpile 2 was covered with topsoil, while piles 3 and 5 were covered with a coarse, sandy soil. However, 30 cm of sandy till was added to the coarse sand on pile 5 in 1980. Revegetation efforts began immediately in 1978. There was no flow reduction by the vegetated coarse sand cover (pile 3). The vegetated covers of topsoil (pile 2) and combined sandy till over coarse sand (pile 5) both produced runoff coefficients 30 to 50% less than the control values.

Three different soil cover systems and one synthetic cover were evaluated for their effectiveness at stemming infiltration into sulfidic mine wastes at the Dunka site (Udoh, 1993). The cover materials under consideration were glacial tills screened to minus 2.5 inches and 0.5 inches, glacial till screened to minus 2.5 inches mixed with 5% bentonite, and a 20-mil PVC membrane. Laboratory tests indicated that the permeability of these materials ranged from  $1.55 \times 10^{-6}$  cm/s (49 cm/yr) for the glacial till down to  $4.12 \times 10^{-9}$  cm/s (0.13 cm/yr) for glacial till mixed with 5% bentonite, meeting the set standard of  $2 \times 10^{-6}$  cm/s (63 cm/yr). These results were similar to Hydrologic Evaluation of Landfill Performance (HELP) model simulations that predicted permeabilities ranging from  $2.1 \times 10^{-6}$  cm/s (66 cm/yr) for minus 2.5 inches of glacial till down to  $5.2 \times 10^{-10}$  cm/s (0.02 cm/yr) for glacial till mixed with 5% bentonite. The model simulations also indicated that the major water loss would be to evapotranspiration. Infiltration was predicted to be reduced by 34% for glacial till cover, and could be decreased by up to 80% if the cover materials were compacted. Actual field test results showed that infiltration was reduced by 60%, 88%, 89%, and 97% for glacial till screened to minus 2.5 inches, till screened to minus 0.5 inches, till screened to minus 2.5 inches mixed with 5% bentonite, and the PVC membrane, respectively.

##### 4.2.5.2. Potential for Capillary Barriers

Cover systems designed to retain moisture within their layers (e.g. capillary barriers), appear to be well-suited to the climate in northern Minnesota, where most reactive mine wastes are found. Northern Minnesota receives approximately twenty eight inches of precipitation each year, two thirds of which falls between May and September (MN DNR, 2000). This amount of rainfall

should be sufficient to maintain a moisture-retaining layer within a capillary barrier. Moisture-retaining layers not only divert infiltration, but also inhibit oxygen transport to the underlying waste. These characteristics help minimize the extent of drainage acidification. Laboratory and field scale studies world-wide have demonstrated the effectiveness of capillary barriers under similar climatic conditions.

There are several potential drawbacks to capillary barrier cover systems. First, most operational demonstrations of capillary barriers have been on tailings within a basin, where side slopes were not an issue. Methods to cover side slopes without additional expense must be developed before extensive use of capillary barriers on stockpiles is feasible. In practice, capillary barriers may be better suited for tailings within a basin or covering waste material disposed of in an open pit rather than large stockpiles.

Second, the cost of constructing a capillary barrier cover system may be prohibitive. Since several layers of different cover materials must be purchased, transported, and put in place, capillary barriers are generally more expensive to construct than simple cover systems. In practice, costs are minimized by using locally available materials. For example, local glacial till is frequently used to construct capillary barriers. However, similar materials are not readily available in northern Minnesota, the third potential drawback. Local alternative materials are likely to include low or desulfurized tailings. One particularly attractive alternative is taconite tailings, which generally contain very little sulfur and several percent carbonate minerals. It is important to note however, that the availability of taconite tailings or any other alternative material will be highly site specific.

#### 4.2.5.3. Potential for Organic Cover Systems

Oxygen-consuming organic covers also show potential for preventing acidic drainage from reactive mine wastes in Minnesota. However, several questions must be addressed before organic cover systems can be used extensively. Two methods of applying organic material to reactive mine wastes have been described in the literature. The first method involves the application of successive layers of organic material over reactive wastes throughout the life of the disposal operation. Ultimately, this practice results in layering and/or blending of organic and mine waste materials. The second method assumes that operations have ceased and that the organic material represents the final cover. At this time, it is not entirely clear which of these methods would be best suited for Minnesota mine wastes.

Numerous organic materials have been tested in laboratory and field cover system demonstrations conducted world-wide. Listed among these materials are forestry wastes, peat, paper residue, and various municipal wastes. Based on demonstrated results reported in the literature and local availability, paper recycling residues, and possibly municipal wastes, appear to be the most promising organic cover materials in northern Minnesota. Both of these options are also attractive in that they use one waste material to contain another.

Unfortunately, both paper recycling residues and municipal wastes contain metals, nutrients, and bioorganic contaminants. As these materials oxidize and break down, these contaminants could be released into surface and ground water, presenting additional environmental toxicity issues. In order to avoid trading one environmental problem for another, laboratory and field leaching tests of any potential organic cover material should be conducted.

#### 4.3. Water Cover Systems

This section reviews the use of water cover systems to physically isolate reactive mine wastes from the surrounding environment, and in doing so, preventing the generation of problematic drainages. During the literature search for this document, it was discovered that a consistent terminology used to describe cover systems did not exist. Consequently, it is important to define the term “water cover system.” Section 4.3 describes water cover systems, in which water is used to submerge reactive mine wastes. This is distinct from “dry” cover systems which utilize solid-phase materials to cover mine wastes.

##### 4.3.1. Principles of Water Cover Systems

As stated previously, the rate of sulfide oxidation can be minimized, by limiting exposure of mine waste materials to water and/or oxygen (equations 1 and 2). Water cover systems are designed to chemically isolate reactive mine wastes by inhibiting oxygen diffusion through a water column (i.e. in a pit or lake) or through interstitial pore spaces (i.e. in a tailings basin). This is due to the marginal solubility of oxygen in water. Dissolved oxygen concentrations are approximately 8.6 mg/L at 25EC, whereas oxygen concentrations in air are 285 mg/L (21%) at 25EC (Table 8). Furthermore, the oxygen diffusion coefficient is four orders of magnitude slower in water than in air (Table 8). As a result, the oxygen flux into a waste material at the sediment interface decreases by a factor of at least 3100 compared to air (Dave, 1992).

Table 8. Oxygen solubility/concentration and diffusion coefficient in air and water at 25EC (adapted from Dave, 1992).

Parameter	Air	Water	Air:Water
concentration (mg/L)	285	8.6	33
molecular diffusion coefficient, $D_o$ ( $m^2/s$ )	$1.78 \times 10^{-5}$	$2 \times 10^{-9}$	8900
effective diffusion coefficient, $D_e$ ( $m^2/s$ )	$1.43 \times 10^{-6}$	$1.6 \times 10^{-10}$	8900
flux at the interface ( $g\ m^2/s$ )	$3.5 \times 10^{-4}$	$1.1 \times 10^{-7}$	3100

In a theoretical, ideal situation, sulfide oxidation rates can be slowed by placing reactive wastes beneath a water cover. Although it is unlikely that sulfide oxidation rates would decrease to such an extreme extent at an operational scale, the amount of alkalinity required to neutralize any acid produced due to sulfide oxidation will be lower beneath a water cover than in air. Depending on specific conditions on site, the required alkalinity may be supplied by neutralizing minerals in the waste itself, surrounding rocks, or surface and/or ground water inputs to the system. The presence of organic materials (e.g. dissolved organic matter, algae, etc.) is also beneficial in that they may further limit the rate of sulfide oxidation by consuming dissolved oxygen.

#### 4.3.2. Water Cover System Designs

There are two general types of water cover systems: underwater disposal and elevated water tables. **Underwater disposal** assumes a permanent water cover above the waste material. Simple underwater disposal consists of depositing reactive mine wastes at the bottom of an existing pit or natural lake. If the pit or lake is deep enough, the water column may become chemically and/or thermally stratified, resulting in a deep layer of anaerobic water above the waste. This stratification further isolates waste materials from oxygen.

In some situations, it may be beneficial to place a surface barrier layer of fine-grained, nonreactive material (clay, sand, soil, till etc.) above the submerged reactive waste. The surface barrier layer further inhibits diffusion of oxygen to and may reduce metal release from reactive wastes. Alternatively, organic materials may be used as a surface barrier layer, which promotes sulfate reduction and metal sulfide precipitation.

If ground water outflow from the system is a concern, a ground water barrier layer can be placed below the reactive waste material. Ground water barriers either block ground water flow (e.g. clay liners), create preferential low resistance flow paths for ground water (e.g. porous envelopes), or remediate outflow prior to entering ground water (e.g. reactive walls or anoxic limestone drains). In some situations, surface and ground water barriers may be used to isolate reactive wastes under water.

**Elevated water tables** involve raising the water table to saturate reactive wastes placed in a pit or tailings basin. In this situation, a ground water gradient often exists. Consequently, infiltrating precipitation and ground water can be expected to flow through the reactive waste as directed by the gradient. Additional surface and ground water barriers can be incorporated into an elevated water table design. Generally speaking, these options are the same as described for the case of simple underwater disposal. However, surface barriers designed to reduce infiltration of precipitation may also be considered.

### 4.3.3. Assessment of Water Cover Systems

#### 4.3.3.1. Variables Affecting the Design of a Water Cover System

Many of the variables affecting water cover design, such as the physical, chemical and mineralogical properties of the wastes, climatic conditions, local regulations, and economics, are identical to those that must be considered when constructing a dry cover. However, water cover designs must also take into account local hydrology, pit, lake, or tailings pond dimensions and orientation, storage volume requirements, minimum water depth for bed stability, and structural engineering requirements (Table 9). Each of these variables is site-specific. Consequently, a water cover system designed for one location will not be suitable for any other.

#### 4.3.3.2. Modeling Water Cover Systems

Frequently, a water cover system will be tested for potential success under several different sets of conditions using hydrologic and/or geochemical predictive models. Predictive models are frequently used to predict the long-term effectiveness of a water cover, since this cannot be determined prior to implementation. Predictive hydrologic models are typically used to establish steady-state flow into or out of the system and future water levels (e.g. in an open pit after closure). Predictive geochemical models are used to determine oxygen transport through the cover, sulfide oxidation rates, metal release, and geochemical equilibria in the water column. The use of models in this way is an inexpensive method to evaluate multiple scenarios (e.g. diversion of surface water to or from a water cover) and their long-term impacts on local surface and ground water quality in a very short amount of time.

#### 4.3.3.3. Effectiveness of a Water Cover System

The effectiveness of a water cover system is typically measured in terms of water chemistry and trace metal release into pore waters, the overlying water layer, and surface and ground water outflow from the system (if applicable). The ultimate water quality overlying a reactive waste will ultimately depend upon sediment transport and wave effects, oxygen flux at the water-waste interface, the nature of waste-water interaction, rate of sulfide oxidation, diffusion of reaction products into the water cover, and dilution or evaporation due to climatic conditions. Each of these factors can be estimated using relatively simple numeric models, laboratory experiments, and field observations. Laboratory experiments are generally designed to determine the physical and/or chemical interactions between reactive wastes and the water cover. They have also been used to test the effectiveness of various barrier layers (e.g. sand, peat, etc.) at reducing oxidation in reactive wastes placed beneath a water cover. These results are often used as input for predictive models (see previous section). Field lysimeter studies are conducted to verify laboratory studies and to demonstrate the effectiveness of these methods under environmental conditions.

Table 9. A wide array of factors should be assessed prior to disposing of reactive waste rock beneath a water cover.

Factor		Tailings Basin	Open Pit	Underground Mine	Natural Lake
Biological	aquatic resources	U	U	U	U
	biota inventory				U
Economic	cost estimate	U	U	U	U
Engineering	dam safety	U			
	extreme climate events (drought, earthquake, flood)	U	U	U	U
	structural integrity	U		U	
Geochemical	ground water chemistry	U	U	U	U
	pore water chemistry	U	U	U	U
	surface water chemistry	U	U		U
	waste chemistry	U	U	U	U
	waste mineralogy	U	U	U	U
	waste particle density	U		U	U
	waste particle size	U		U	U
	waste type (fresh or oxidized)	U	U	U	U
Hydrologic	climate	U	U		U
	local topography	U	U		U
	water balances	U	U	U	U
Physical	bathymetry		U		U
	discharge rate	U	U	U	U
	minimum water depth	U	U		U
	orientation	U	U		U
	storage volume	U	U	U	U
	surface dimensions	U	U		U
	wave action	U			U

#### 4.3.4. Cost Analysis of Water Cover Systems

The expense associated with a water cover system increases as the cover system becomes more complex. Construction costs may become an issue if a bottom rock drain or dams must be included in the disposal design. Maintaining dams around flooded tailings basins will also increase long-term costs. Even relatively simple disposal of waste materials in an open pit or lake involves transportation of the material to the disposal site. Due to the wide range of water cover system options and site-specific variables, the actual expense of a water cover system must be evaluated in terms of the needs and requirements of individual operations.

#### 4.3.5. Water Cover Studies and Potential Use in Minnesota

##### 4.3.5.1. MN DNR Subaqueous Disposal Studies

Water cover systems represent another promising method for preventing acidic drainage from reactive mine wastes in Minnesota. The MN DNR, Division of Lands and Minerals has conducted five laboratory experiments and one field experiment to examine the effectiveness of subaqueous disposal and subaqueous disposal with various barrier layers for controlling the oxidation of sulfide minerals present in mine wastes (Lapakko et al., 1998a):

1. Subaqueous disposal tests in the absence of oxygen consuming layers or oxygen diffusion barriers were initially conducted in 2-L Erlenmeyer flasks;
2. Subsequent tests were conducted in 2-in.-diameter columns to describe the release of sulfate as a function of sulfidic mine waste depth for two different mine waste particle sizes;
3. In a second phase of the Erlenmeyer flask experiment (#1), composted yard waste was added to some of these flasks to determine if organic amendments would further inhibit oxidation;
4. Composted yard waste and fresh rock, as well as controls for compost and rock, were subjected to subaqueous dissolution in the laboratory;
5. A column experiment was conducted to determine the effectiveness of taconite tailings, yard waste compost/tailings mixtures, and limestone layers as potential oxygen diffusion barriers above sulfidic mine wastes; and
6. Barriers of tailings, a yard waste compost/tailings mixture, and limestone are presently under examination in small-scale field tests.

Subaqueous disposal tests in the absence of oxygen consuming layers or oxygen diffusion barriers were initially conducted in flasks (#1 above) using Virginia Formation hornfels rock containing 14% pyrrhotite and in columns (#2 above) using sulfidic Duluth Complex rock (Lapakko et al., 1998). These two experiments revealed that separating sulfidic mine waste from the atmosphere with a layer of water would reduce the rate of oxidation of sulfide minerals present. However, oxidation would continue and possibly produce acidic waters.

Modified subaqueous disposal experiments with barrier layers were conducted in flasks (#3 above), beakers (#4 above), and columns (#5 above; Lapakko et al., 1998a). Composted yard waste was used as an oxygen consuming layer between sulfidic rock and the overlying water in the flask and beaker tests. Tailings, tailings/yard waste compost mixtures, and limestone were used as barrier layers in the column tests. In general, the inorganic barrier layers were more effective at decreasing sulfide oxidation rates than the compost barriers (MN DNR Hibbing laboratory, unpublished data). However, it was noted that experimentally determined rates of acid production averaged roughly 500% higher than theoretical values based on oxygen diffusion through a water column. This is a considerable difference, and should be considered a result of the weekly addition of fresh, oxygen saturated water. The amount of water removed for sampling and replaced with new water was nearly 5% of the entire column water volume for each sample. Therefore, the rates of acid production due to pyrrhotite oxidation in this experiment are not controlled by the rate of oxygen diffusion, but by the addition of oxygenated water to the rock bed.

A field test (#6 above) examining the effectiveness of limestone, tailings and a tailings compost mixture above sulfidic rock in a subaqueous environment is presently in progress at the MN DNR Research Site in Hibbing, MN (Lapakko et al., 1998a). The subaqueous tests are being conducted in large cylindrical tanks (d = 46-in., h = 11-ft.), fitted with a 30-mil PVC liner. The tanks were filled to a depth of 48 inches with minus 3/4 inch Duluth Complex rock with sulfur contents ranging from 0.63 % to 0.69 %. Initial calculations have yielded negative acid production rates (MN DNR Hibbing laboratory, unpublished data). The most likely explanation for this is that the amount of sulfate released due to sulfide oxidation is very small relative to sulfate concentrations already present in the water cover, and therefore, cannot be measured accurately.

#### 4.3.5.2. MN DNR Study to Create Wetlands on Reactive Tailings

In addition to subaqueous disposal, the MN DNR, Division of Lands and Minerals has been investigating various strategies to create wetlands on acid and non-acid generating tailings (Eger et al., 2000a,b). This discussion will focus on small-scale field studies conducted on acid-generating tailings at the MN DNR field research facility in Hibbing, MN (Eger et al., 2000a). Acid-generating tailings from a massive zinc sulfide deposit in Winston Lake, Ontario, were placed in small cylindrical tanks. Two uncovered controls were established in addition to five treatments: 61-cm cover of wetland soil, 61-cm of glacial till, 61-cm of tailings, and two tanks with 71-cm of water. One of the water covers included the aquatic macrophytes: *Elodea canadensis*, *Potamogeton sp.*, and *Ceratophyllum demersum*. Cattails (*Typha sp.*) were planted in the tanks with substrate cover. Overall, the wetlands created over acid generating tailings were effective at preventing acid conditions (pH = 6.0-6.5) and minimizing metal release (Zn = 0.02 - 0.1 mg/L) relative to the “on-land” controls (pH = 3.1-3.2; Zn = 6400-7400 mg/L; Eger et al., 2000a). The authors suggested that successful creation of wetlands in large tailings basins would require a 60-cm soil cover along the shoreline and establishment of submergent vegetation in the deeper water areas.

#### 4.3.5.3. Potential Use of Water Covers in Minnesota

In summary, both of these strategies show promise as mitigative strategies for reactive mine wastes in Minnesota. Subaqueous disposal in abandoned open mine pits represents one alternative for water covers. Decades of iron mining in northern Minnesota has left a large number of abandoned open mine pits across the Mesabi and Cuyuna Iron Ranges. One option may be to backfill reactive mine wastes into selected pit lakes. The advantage of this alternative is that these pits lie below local ground water tables and fill with surface and ground water naturally.

However, backfilling open pit lakes with reactive mine waste should be approached cautiously. As open pits fill with water, local ground water gradients can reverse, resulting in ground water outflow into local aquifers. Any contamination of water within the pit due to oxidation of reactive mine waste will be carried into the aquifer, potentially resulting in contamination. This would be particularly disadvantageous, since a large number of communities in northern Minnesota draw drinking water from these aquifers. Consequently, any plan to dispose of reactive mine wastes within an existing open pit should involve extensive geochemical and hydrologic characterization as well as ground water monitoring to ensure that local water supplies are protected.

Elevating the local water table and establishing wetlands on reactive tailings represent a second alternative water cover system. Wetlands have several advantages including the creation of an anoxic zone above reactive wastes, accumulation of decaying organic matter provides additional substrate to improve growth, and fits with reclamation goals aimed at restoring wetlands in mining areas.

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## CONTROL STRATEGIES

Whether or not a mine waste produces problematic drainage depends on a wide array of variables that describe the physical, chemical, and mineralogical characteristics of the waste, pre-existing site conditions, climate, and government regulations. Consequently, there is no single, correct approach to environmental mine waste management. In some cases, it may be acceptable to limit generation of problematic drainages rather than strict prevention. Referred to as “control strategies,” these methods generally utilize amendments to stabilize mine wastes and inhibit formation of problematic drainages. This section will review the use of alkaline amendments, microencapsulation techniques, and bactericides to control drainage chemistry.

### 5. Chemical Stabilization of Reactive Mine Wastes

#### 5.1. Alkaline Amendments

##### 5.1.1. The Goals of Alkaline Amendments

One approach to controlling the acid release from reactive mine waste is to amend reactive wastes with alkaline materials. While the most commonly used alkaline amendment, limestone, is a natural geologic material, industrial materials and waste products have also been used in mining applications (Table 10).

Table 10. Natural geologic, industrial, and waste products have been used as alkaline amendments to neutralize acidity produced by reactive mine wastes.

Natural Geologic	Industrial	Waste Products
limestone/calcite	lime	alkaline waste rock or tailings
dolomite	caustic soda (NaOH)	active treatment sludges
phosphate/apatite	Bauxsol	lime kiln dust
	sodium carbonate	coal combustion by-products
		steel making slag
		lime stabilized sewage sludge

Alkaline solids neutralize acid produced by the oxidation of sulfide minerals. This neutralization has three secondary effects. First, the elevated pH yields an environment which is unsuitable

for *Thiobacillus ferrooxidans*, a strain of bacteria which catalyzes sulfide mineral oxidation. The elimination of these bacteria limits the rate of sulfide oxidation, and therefore, the rate of acid production. Second, the elevated pH enhances the oxidation of ferrous iron and the subsequent precipitation of ferric oxyhydroxides. If the pH is elevated in the immediate neighborhood of iron sulfide mineral surfaces, precipitates will form on the mineral surface. This would impede chemical transport to and from the iron sulfide mineral surface, and consequently, inhibit iron sulfide oxidation and the attendant acid production. Third, as pH increases the equilibrium concentrations of trace metals decrease. The decrease in trace metal concentrations is due to increased precipitation (as hydroxides, oxides, and/or carbonates) and adsorption.

An important factor in the balance between acid production and acid neutralization is the relative masses of the iron sulfide and calcium and magnesium carbonate minerals. The alkaline dosage requirement can be estimated based on theory or empirical evidence. The theoretical alkalinity requirement can be calculated assuming that each mole of sulfur produces two moles of acid and that each mole of calcium carbonate equivalent consumes two moles of acid. Calculation of the acid-producing sulfur content should be based on sulfur associated with iron sulfide minerals.

#### 5.1.2. Assessment of Alkaline Amendment Effectiveness

The effectiveness of an alkaline amendment is measured in terms of the quality of water draining from the mine waste. This is usually determined in terms of drainage pH, acidity, and chemical release (e.g. sulfate and trace metals). In other words, a successful amendment will result in near-neutral drainage pH with low levels of sulfate and trace metals.

Laboratory studies on alkaline amendments are typically conducted in columns, where reactive waste rock or tailings are mixed with alkaline material (e.g. limestone, lime, fly ash, etc.). Alkaline loadings are geochemically determined by the acid producing potential (AP) of the waste material and the neutralization potential (NP) of the alkaline amendment. Although a wide range of NP to AP ratios have been tested, those in the range of 1:1 to 2:1 tend to temporarily neutralize acid produced due to sulfide oxidation (Day, 1994; Stewart et al., 1994; Lapakko et al., 1997, 2000).

#### 5.1.3. Cost Analysis of Adding Alkaline Materials to Reactive Mine Wastes

The actual cost involved with amending reactive mine wastes with alkaline material varies widely depending on the type of alkaline material used, application ratio, and accessibility of the location. Chemical costs tend to be low, ranging from zero to tens of dollars per ton (Table 11). Waste alkaline materials, such as fly ash, are the least expensive and are usually free.

Regardless of the type of alkaline material used, transportation constitutes the majority of the expense involved with this strategy. Estimated overland hauling costs for a driver and 20-ton capacity truck range from \$60 to \$80 per hour. Assuming an average hauling speed of 40-mph, overland hauling costs would be \$0.10 per mile per ton.

Table 11. Prices for alkaline materials vary widely. These values do not include transportation costs.

Alkaline Material	Suppliers <sup>1</sup>	Cost of Material (\$/ton)
limestone	Presque Isle, Alpena, MI; Great Lakes Calcium; Bryan Rock, Bayport, MN; Northland Constructors, Duluth, MN	\$8-60
manufactured sand (limestone product)	Camas, Shiely Division, Grey Cloud Island, St Paul, MN	\$7.50
fly ash	Archer Daniels Midland, Mankato, MN; NSP <sup>2</sup> , Riverside & Black Dog plants, MN	0-\$2
lime kiln dust	Cutler-Magner, Duluth, MN	free

<sup>1</sup>Suppliers of alkaline materials used by the MN DNR for past laboratory and field experiments.

<sup>2</sup>Northern States Power now owned by Excel Energy

#### 5.1.4. Alkaline Amendments and Reactive Mine Wastes in Minnesota

While a few successes have been reported in laboratory studies, field demonstrations of alkaline amendments have been less successful at neutralizing acidic drainage (Evans and Rose, 1995). In rare cases, marginal, short-term results have been claimed as mitigative successes (Davis et al., 1999). The differences in results between laboratory and field studies is not particularly surprising. Incomplete mixing of the reactive waste and alkaline material, as well as the formation of preferential flow paths, are cited as limitations to the effectiveness of field treatments (Mehling et al., 1997). Furthermore, the reactivity of limestone, in particular, is limited when particle size is large. This situation may be improved when alkaline amendments are used to treat fine-grained mine wastes such as tailings. With finer mine wastes and alkaline solids a homogeneous mixture can be more readily attained and, due to more uniform particle sizing, flow tends to be more uniform than in the wide range of particle sizes in waste rock piles.

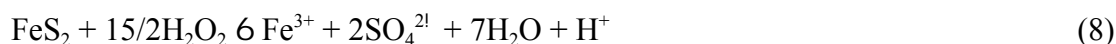
Recent analyses of alkaline amendment strategies indicates that these physical and hydrologic limitations may have a greater impact on mitigative success than the geochemical characteristics of the waste and alkaline materials (Kempton et al., 1997; Morin and Hutt, 2000). The latter publication indicated that waste rock drainage acidity is dependent on the flow path length within acid neutralizing rock separating zones of acid generating rock. Analysis indicated that waste rock with a bulk NP:AP ratio of 300:1 could release acidic drainage if appropriate neutralizing rock flow path length was not attained (Morin and Hutt, 2000).

## 5.2. Microencapsulation of Reactive Wastes

### 5.2.1. Mechanisms of Microencapsulation

Generation of problematic drainage can be controlled by isolating reactive surfaces from oxidizing agents (i.e.  $O_2$ ,  $Fe^{3+}$ ). This can be accomplished by chemically precipitating a ferric coating on the surface of the waste material. This process, often called microencapsulation, prevents further oxidation of sulfide minerals by blocking the transport of oxidants to the sulfide surface and consuming ferric iron before it can become an oxidant.

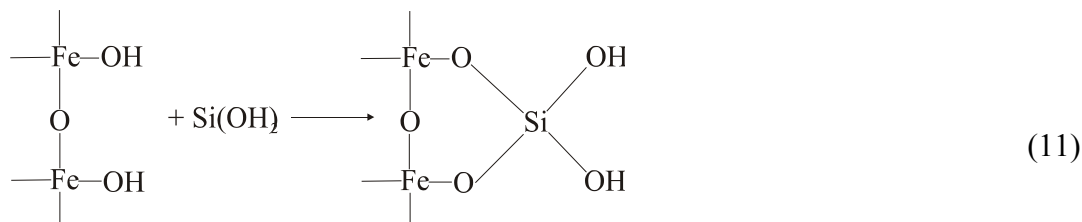
The coating is produced by reacting sulfidic material with low concentrations of an oxidizing agent in the presence of soluble phosphate or silica in a buffered solution. Hydrogen peroxide or calcium hypochlorite are typically used as oxidizing agents. The oxidizing agent reacts with the sulfide to produce ferric ions:



Sodium acetate is used to buffer the solution at a pH of 5 to 6. At this pH, dissolved ferric iron is unstable and precipitates as ferric hydroxide. If dissolved phosphate is present, it will scavenge ferric ions and ferric phosphate will precipitate:



If silicic acid is present in the solution, it will react with the ferric hydroxides, producing an insoluble ferric silicate precipitate that is chemically stable at low pH (Evangelou, 1996):



Silicate coatings are generally considered more “environmentally friendly” than phosphate coatings. This is because the addition of excess dissolved phosphate species to the mine waste may result in contamination of local surface and/or ground waters. Silicate coatings are more resistant to degradation, especially under acidic conditions.

In addition to phosphate and silicate coating materials, organic derivatives have been used to create a thin surface coating on reactive mine wastes. The exact composition of these materials are frequently vague but, some examples include epoxy resins, dust suppressants, and latex coatings.

#### 5.2.2. Assessment of Microencapsulation Effectiveness

Approximately half of the studies cited here dealt with the feasibility of chemically producing coatings on reactive mine wastes and tailings. Reasonably successful coatings were reported in laboratory studies using phosphates, silicates, and various organic materials. Generally, the presence of coatings on rock and tailing surfaces were confirmed using SEM, although decreased iron levels in drainages was often cited as evidence of coating formation.

Twelve laboratory and field studies addressed issues of drainage quality from treated samples of waste rock and tailings (Table 12). Seven studies reported drainage pH, which was usually in the range of 6 to 7. However, alkaline materials were used during two of these tests, which would have influenced the results. Finally, laboratory and field tests have reported that induced coatings inhibited the release of arsenic and heavy metals (e.g. lead, zinc, and cadmium) from mine waste materials (Table 12).

#### 5.2.3. Cost Analysis of Microencapsulation Techniques

Microencapsulation is a fairly new technology and few field-scale applications exist. Most field applications are proprietary demonstrations that did not include cost analyses. Only two estimates were found in the literature. In 1995, microencapsulation of a waste material containing 30% pyrrhotite was estimated to be \$4-5/ton of waste (Georgopoulou et al., 1995). A more recent estimate for one acre of soil treated to a depth of 50-cm was \$20,000 - 30,000 (Conca et al., 1999). Since application of microencapsulation solutions requires no specialized equipment, the major expense is that of chemicals.

Table 12. Summary of drainage quality results from microencapsulation laboratory tests and field demonstrations.

Reference	Coating Type	Drainage Quality Results		Test Duration
		pH > 6	Reduced Metals	
Laboratory Studies:				
Fytas et al., 1999	silicate	U	U	80 days
Fytas and Evangelou, 1998	phosphate	U	U	5 weeks
Fytas et al., 2000	phosphate	U <sup>1</sup>	U	101, 227, 318 days
Vandiviere & Evangelou, 1998	phosphate & silicate	U <sup>2</sup>	nd <sup>3</sup>	16, 18, 19, 28 months
Field Studies:				
Conca et al., 1999	phosphate	nd	U	nr <sup>4</sup>
Littlepage et al., 2000	phosphate	nd	U	nr
Jensen et al., 1999	phosphate	nd	U	nr
US EPA, 1999	phosphate	nd	U	nr
Williams et al., 1999	phosphate & silicate	U	U	nr
Mitchell and Anderson, 2000	silicate	U	U	nr
Chatham and Svee, 1996	silicate	U	U	nr
KEECO	silicate	nd	U	nr

<sup>1</sup>all but one column experiment maintained pH >6

<sup>2</sup>mixed with CaCO<sub>3</sub>

<sup>3</sup>nd = not determined

<sup>4</sup>nr = not reported

#### 5.2.4. Potential Use of Microencapsulation Technology to Address Mine Waste Issues in Minnesota

Application of microencapsulation techniques are likely to be very site-specific based on individual rock types. It has been suggested that physical parameters such as particle size and shape may determine the extent to which an effective microencapsulation coating can be induced (Evangelou, 2000). Furthermore, mineralogic characteristics of the waste material may also affect microencapsulation effectiveness. A coating solution that inhibited pyrite oxidation in a field demonstration may not effectively encapsulate pyrrhotite-bearing waste material (i.e. from the Duluth Complex). Therefore, laboratory tests should be conducted on specific rock types to optimize the method of microencapsulation prior to implementation of a field-scale project.

Field demonstrations of microencapsulation technology suggest that a wide range of metals can be immobilized in soils and tailings using a phosphate- or silicate-type coating (Chatham and Svee, 1996; Conca et al., 1999; Jensen et al., 1999; US EPA, 1999; Williams et al., 1999; Littlepage et al., 2000; Mitchell and Anderson, 2000). While these results look promising, the exact method of application on an operational scale is unclear at this time. Several questions still need to be answered:

- How are microencapsulation coatings induced under field conditions?
- How are optimized coating solutions determined in the laboratory scaled up for use in the field?
- Most of these studies used pyritic tailings or finely crushed waste rock. Can a uniform coating on larger particle size material (i.e. 1-2 inches) be achieved?
- What is the long-term stability of these coatings?

#### 5.3. Organic Amendments

The following information was excerpted from Gusek et al. (in progress). Oxygen is one of the key reactants in the generation of acidic drainage from reactive mine wastes (reactions 1 and 2). If the presence of oxygen within a stockpile, tailings basin, or waste rock dump can be limited, the rate of acid generation can be minimized. One approach is to intermix organic material with the waste. As the organic material decomposes, it consumes oxygen (reaction 12):



where,  $\text{CH}_2\text{O}$  represents a generic organic material. If adequate organic material is present, the rate of oxygen consumption will exceed the rate of oxygen diffusion, preventing oxygen from reacting with reactive minerals. Ultimately, organic materials may begin to ferment, producing reducing conditions within the waste. Under reducing conditions the rate of sulfide oxidation is extremely low. Reducing conditions also favor sulfate reducing bacteria, which help increase drainage pH and remove metals from pore waters.

Although there are several case studies where organic material has been placed on top of tailings (Appendix 3), there is little information on incorporation of organic material into mine waste. Greenworld Science has several patents on the use of this technology and have injected soluble organics at metal mining sites to remediate contaminated groundwater (Greenworld Science).

#### 5.4. References on the Chemical Stabilization of Reactive Mine Wastes

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## 6. Biological Stabilization of Reactive Mine Wastes

### 6.1. Inhibition of Biological Sulfide Oxidation Using Bactericides

Bacteria, such as *Thiobacillus ferrooxidans*, have been shown to catalyze the oxidation of ferrous iron in reactive mine waste effluents below a pH of approximately 4.5. Bacterial oxidation produces acid, which may result in highly acidic drainage. **Surfactants** have been shown to interrupt bacterial activity, reducing the rate of oxidation, and therefore, controlling the rate of drainage acidification.

Surfactants can be obtained as a powder or in a controlled release pellet. Powdered surfactants can be slurried with water and sprayed directly onto the mine waste material, or incorporated with the mine waste material as it is deposited. Drawbacks to powdered surfactants are that they are easily flushed through the waste material and that they degrade over time. Consequently, surfactants require frequent re-application. Re-application rates are estimated at three to six months (Patterson, 1987; Delaney et al., 1997).

Control release pellets incorporate the surfactant in a rubberized matrix. The surfactant is gradually released through a diffusion mechanism. Since pellets are not likely to be flushed from the waste material, their bactericidal properties can last for several years.

Recently, a second class of bactericides has been investigated. **Thiol-blocking agents** have been shown to block the oxidation thiosulfate and sulfur to sulfate (i.e. sulfuric acid). Most of these chemicals are toxic and unsuitable for environmental applications. However, one class of thiol-blocking agents, heterocyclic mercaptans, are considered “relatively non-toxic.” Heterocyclic mercaptans, have been reported to inhibit the activity of *Thiobacillus thioparus*, a neutrophilic sulfur-oxidizing bacteria, in laboratory column and field tests (Stichbury et al., 1995; Lortie et al., 1999). As with the surfactants, the affects of heterocyclic mercaptans are temporary and would require re-application.

### 6.2. Assessment of Bactericides as Sulfide Oxidation Inhibitors

#### 6.2.1. Preliminary Evaluation of Bactericides

Prior to application on an operational-scale, physical, chemical, and biological characterization of the waste material should be completed. Laboratory studies are generally designed to determine the acid generation potential in terms of sulfide minerals present as well as bacterial activity (i.e. bacteria counts). This information, used in conjunction with laboratory column tests, can be used to determine an appropriate bactericide application rate for a specific waste rock type (Kleinmann and Erickson, 1988). However, application rates determined in the laboratory do not always extrapolate well to operational-scale, environmental conditions (Patterson, 1987; Delaney et al., 1997).

### 6.2.2. Parameters Used to Assess Bactericide Effectiveness

The effectiveness of a bactericide application can be measured in terms of the quality of water draining from the mine waste. This can be determined in terms of drainage pH, acidity, chemical release (e.g. sulfate, iron, and other metals), and bacteria counts. If the application of bactericide successfully inhibits biological oxidation of sulfides in a mine waste, drainage pH should remain neutral, while acidity, chemical release of sulfate and metals, and bacteria counts remain low.

In laboratory and field studies reported in the literature, the effectiveness of **surfactants** as bactericides was frequently reported in terms of drainage acidity (Tables 6.2.2a and 6.2.2b). Unfortunately, these data did not indicate whether or not the drainage met the environmental water quality standard (pH >6). The fact that acidity was present suggested that these drainages did not meet this standard. This is not an unexpected result because surfactants inhibit the activity of acidophilic bacteria. Acidophilic bacteria do not become active until the pH reaches an acidic range, approximately 4-4.5. Consequently, drainage pH values as low as 4 can be expected when surfactants are used as bactericides.

**Heterocyclic mercaptans**, however, inhibit the activity of neutrophilic sulfide oxidizing bacteria. When heterocyclic mercaptans are used as bactericides, drainage pH values in the neutral range are more likely to occur. Drainage pH did remain elevated in the two studies that have been conducted on heterocyclic mercaptans (Stichbury et al., 1995; Lortie et al., 1999). However, it did not necessarily remain above 6.0 (Tables 13 and 14).

### 6.3. Cost Analysis of Bactericides

Based on case studies presented by Benedetti et al. (1990) and Rastogi et al. (1990), a single application of surfactant to an acre of waste material will cost between \$1000 and \$3000. Note that these are 1990 dollars and have not been adjusted. These cost estimates do not include those associated with waste disposal, neutralization chemicals, grading, or other reclamation measures. No estimates were found for heterocyclic mercaptans.

### 6.4. Potential Use of Bactericides to Inhibit Sulfide Oxidation in Minnesota Mine Wastes

Bactericide proponents claim that bactericides show promise as a preventative strategy against the development of acidic drainage from mine wastes. They are attractive because they are inexpensive and can be applied easily either sprayed on as a slurry or distributed within the waste in pellet form. Bactericides appear to be the most effective at preventing acid mine drainage when they are used on fresh waste material and reapplied frequently throughout operations. They should not be expected to remediate an existing drainage problem.

Table 13. Summary of results from laboratory studies designed to evaluate the effectiveness of bacterial inhibitors at reducing the effects of sulfide oxidation.

Reference	Inhibitor Used	Maintained pH >6	Drainage Quality Results			Inhibited Bacteria
			Reduced Acidity	Sulfate Reduced	Metals Reduced	
Sobek, 1987	surfactant	nd <sup>1</sup>	U	nd	nd	nd
Patterson, 1987	surfactant	nd	nd	nd	U (Cu)	nd
Watzlaf, 1988	surfactant	nd	U	nd	nd	U
Stichbury et al., 1995	heterocyclic mercaptans	no U	nd nd	neg. <sup>2</sup> no	nd nd	nd nd

<sup>1</sup>nd = not determined  
<sup>2</sup>neg. = negligible reduction

Table 14. Summary of results from field tests on the effect of bacterial inhibitors on reactive mine waste drainage quality.

Reference	Inhibitor Used	Maintained pH >6	Drainage Quality Results			Inhibited Bacteria
			Acidity Reduced	Sulfate	Metals	
Patterson, 1987 & Delaney et al., 1997	surfactant	no	nd <sup>1</sup>	nd	nd	no
Parisi et al., 1994	surfactant	no	U	U	nd	nd
Lortie et al., 1999	heterocyclic mercaptan	no	nd	U	U (Fe, Cu, Ni, Zn)	U
Lortie et al., 1999	heterocyclic mercaptan	no	nd	U	U (Fe)	U

<sup>1</sup>nd = not determined

It is important to remember that bactericides represent a short-term mitigative strategy that will require the use of another, longer-term mitigative strategy for long-term control. The most frequently reported objective of bactericide usage has been to prevent acidification long enough for vegetation to become established on a reclaimed surface. In these cases, reclamation efforts are usually expected to minimize the amount of water that must be collected and treated prior to discharge into the environment. Bactericides used in this manner are considered to be a cost effective method to minimize treatment costs. Alternatively, establishment of a healthy vegetative cover may, in some situations, be sufficient to prevent future acidification of drainage. Although reported less frequently, it seems logical that bactericides could be used to prevent acidification of mine wastes during operations until a dry cover system, including a vegetated surface, can be placed over the mine waste.

There are two major drawbacks to the use of bactericides. First, surfactants cannot be relied upon to maintain drainage pH above 6.0. This pH may be sufficient to maintain a healthy vegetative cover in some instances. However, it generally does not meet acceptable environmental standards. As a result, the use of surfactants alone is not a viable option for preventing acidic drainage problems in Minnesota.

Second, the “relatively non-toxic” nature of heterocyclic mercaptans has not been defined in the literature. It is unclear at this time whether or not heterocyclic mercaptans can or should be released to the environment. Furthermore, neither of the two heterocyclic mercaptans described in the literature could be found in the Minnesota Pollution Control Agency (MN PCA) database (Clark, 2000). Based on information found on Material Safety Data Sheets for these two chemicals, aquatic toxicity has not yet been determined. Since the MN PCA does not have adequate toxicity information on either of the two heterocyclic mercaptans, no standard exists to govern their usage. Consequently, the MN PCA recommends conducting biological toxicity tests on drainages containing these compounds prior to extensive use (Clark, 2000).

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## TREATMENT SYSTEMS

Whether or not a mine waste produces problematic drainage depends on a wide array of variables that describe the physical, chemical, and mineralogical characteristics of the waste, pre-existing site conditions, climate, and government regulations. Consequently, there is no single, correct approach to environmental mine waste management. Oftentimes, prevention and control strategies are insufficient to mitigate problematic drainages satisfactorily. In these cases, it is necessary to collect and treat the drainage prior to discharge to the surrounding environment. Both passive and active approaches have been taken to treat mine drainages. This section will review the application of constructed wetlands, permeable reactive barriers, and several active treatment options for the treatment of problematic mine drainages.

### 7. Passive Treatment Systems

Passive treatment systems take advantage of naturally occurring biological, physical, and chemical processes to remove contaminants from surface and ground waters (Table 15). Removal of metals from waste rock stockpile drainage as it flowed through a white cedar swamp in northern Minnesota has been well documented (Eger et al., 1980; Eger and Lapakko, 1988). These and similar observations in other natural wetlands (Wieder and Land, 1982; Samuel et al., 1988; Sencindiver and Bhumbla, 1988; Sobolewski, 1997; Skousen et al., 1998; Wildeman and Pavlik, 2000) are the basis for engineering systems with conditions that promote metal removal and/or elevate alkalinity in mine drainages without continuous chemical or operational requirements.

Table 15. Summary of naturally occurring removal mechanisms for species commonly found in acidic mine waste drainages.

Removal Mechanism	Elements Affected
sulfate reduction	SO <sub>4</sub> , acidity
sulfide precipitation	As, Cd, Cu, Fe, Mn, Ni, Pb, Zn
oxyhydroxide precipitation	Al, Fe, Mn, Zn
carbonate precipitation	trace metals
adsorption	trace metals and As
ion exchange/chelation	trace metals
filtration	suspended and colloidal material
uptake by plants	metals

## 7.1. Constructed Wetlands

Although the design of wetland treatment systems can vary substantially, there are two basic types; aerobic (overland or surface flow) wetlands, and organic substrate (vertical flow) wetlands (Eger, 2000). Settling ponds and alkaline producing systems (i.e. SAPS and open limestone channels) are often incorporated into passive treatment system designs to improve effluent metal and alkalinity levels. Constructed wetlands require little operational attention or maintenance, and therefore, may be favorable over conventional active treatment systems. Disadvantages of constructed wetland systems include the large area of open land required for adequate treatment, the relatively low flow requirements, and diminished performance in cold climates.

### 7.1.1. Aerobic Wetlands

Aerobic wetland systems promote metal removal via oxidation reactions in the surface water or upper portion of the substrate where oxygen is present in the highest concentrations. Iron removal occurs primarily through precipitation of ferric oxyhydroxides. Trace metals are removed by adsorption, chelation, or ion exchange processes associated with the organic component of the substrate. Since these reactions are favored at high pH, aerobic wetlands are most effective when the mine drainage is net alkaline.

Aerobic wetlands can be used for acidic drainages following adequate pretreatment using an anoxic limestone drain (ALD). ALDs are typically constructed in an area with seeps, where the water will contain little to no dissolved oxygen. The presence of even a small amount of dissolved oxygen will result in precipitation of iron oxyhydroxides, which can coat limestone rendering it ineffective and/or plug the drain. Thus, oxygen must be excluded from the drain.

An ALD usually consists of a trench constructed to intercept the contaminated ground water and filled with two to three inch limestone. The goal of the drain is to provide sufficient alkalinity to the drainage so that when the water surfaces, there is sufficient alkalinity to maintain a neutral pH. Outflow from the drain collects in an aerobic wetland or settling pond to remove iron and other metals.

### 7.1.2. Organic Substrate Wetlands

Organic substrate wetlands are anaerobic systems that promote bacterial sulfate reduction, which produces dissolved hydrogen sulfide and bicarbonate alkalinity (reaction 13),



where  $\text{CH}_2\text{O}$  represents a low molecular weight organic substance such as lactate or acetate.

The wetland is constructed with organic material (e.g. compost) that is usually supplemented with limestone. Trace metal removal occurs primarily through the precipitation of metal

sulfides, hydroxides, and carbonates. Acid is neutralized both through sulfate reduction processes and limestone dissolution. Organic substrate wetlands are most effective when the mine drainage is acidic.

### 7.1.3. Assessment of Constructed Wetland Treatment System Designs and Size

Wetland treatment systems are evaluated primarily by the water quality at the outflow of the wetland (Hedin et al., 1994). In order to produce an effluent which meets water quality standards, wetlands must be designed properly. For detailed information on wetland treatment design considerations, a reasonably comprehensive overview was compiled by Wildeman et al. (1993).

Important factors in the design are the drainage pH and total metal load (Figure 4). Net alkaline waters can be treated using an aerobic wetland. Net acidic waters must either be pretreated using an ALD or treated using an organic substrate wetland. If the raw water contains high levels of dissolved oxygen and iron, an organic substrate wetland is the preferred treatment strategy. Oftentimes, a combined system is the most effective. Several wetland cells may be connected to optimize metal removal or to treat large flow volumes.

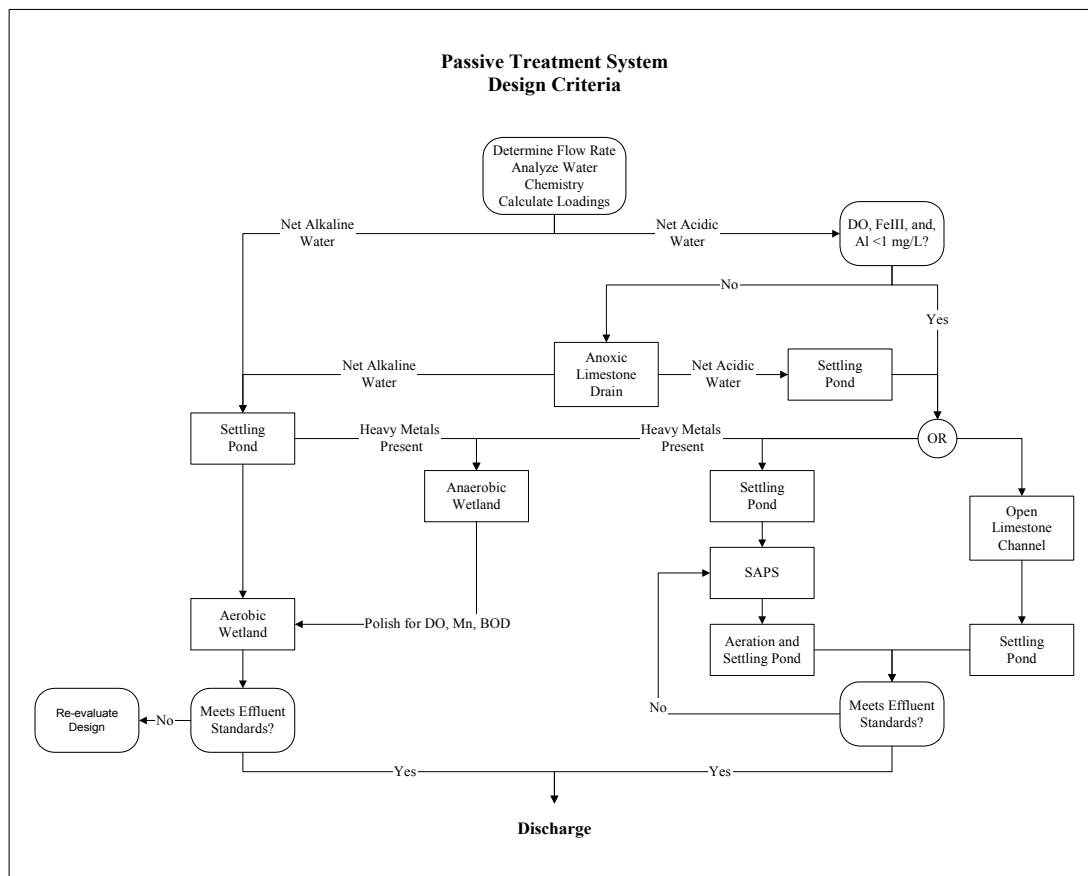


Figure 4. Constructed wetland design depends upon the chemistry of water to be treated. Modified Gusek et al., 2001.

Wetland treatment systems require large areas of space in order to be effective. Inadequate metal removal and poor performance will result if the size of the system is underestimated. Hedin et al. (1994) developed sizing criteria for constructed wetlands based on previous cases. In order to consistently meet regulatory guidelines for an alkaline drainage, sizing factors were 10 g/m<sup>2</sup>d and 0.5 g/m<sup>2</sup>d for iron and manganese, respectively. The acidity sizing factor for a net acid drainage was 3.5 g/m<sup>2</sup>d. It is important to note that these “compliance criteria” were generated for coal mine drainages in Appalachia and may not be appropriate for metal mining operations or other regions. However, Wildeman et al. (1993) considered these sizing factors reasonable for wetland treatment of metal mine drainages in Colorado.

#### 7.1.4. Cost Analysis of Constructed Wetland Treatment Systems

Costs for these systems can vary depending on the system design and complexity. Construction costs for liners, trenches, and spillways will increase as the number of treatment cells increase. Since constructed wetland treatment system designs are site-specific, average construction costs are difficult to determine. Costs estimates include a wide variety of capital, construction, and operation costs (Table 16). It has been estimated that constructed wetland installation costs range from \$5 to \$32/m<sup>2</sup> (Wildeman et al., 1993). Installation of constructed wetlands at the Dunka Mine site in northern Minnesota varied from \$19 - \$31/m<sup>2</sup> (Eger, 2000).

Table 16. Summary of major cost components for constructed wetlands (modified from Wildeman et al., 1993).

Capital	Construction	Operation/Maintenance
engineering (design, testing, & specifications)	personnel	inspections
environmental studies & permitting	transportation	monitoring
land acquisition	site preparation	cleaning of conveyances
access (rights of way)	source control /diversion	flow adjustments
	earthwork	major overhauls (i.e. substrate rejuvenation/replacement)
	basin lining	
	concrete	
	plumbing	
	substrate conditioning /installation	
	vegetation	

#### 7.1.5. Constructed Wetland Treatment of Problematic Drainages in Minnesota

The majority of constructed wetlands have been applied to coal mine drainage issues. However, they do have some potential for addressing hard rock mining issues. A survey of 128 coal mine drainages found that coal mine drainages can have copper, nickel, and zinc concentrations as high as 2.49, 10, and 146 mg/L, respectively (Hyman and Watzlaf, 1996). These levels are comparable to those expected for hard rock mine drainage.

The MN DNR has conducted considerable research on the topic of constructed wetlands to treat problematic drainages on a laboratory-, field-, and operational-scale. Field studies have also addressed issues related to establishing wetlands on sulfidic tailings as a preventative technique. A bibliography of MN DNR publications related to these studies is provided here (Appendix 6).

Currently in Minnesota, there are five operating constructed wetland treatment systems at the Dunka Mine site near Babbitt, and one proposed sulfate reduction system at the Soudan State Park (Eger, 2000). The sulfate reduction system proposed to treat mine drainage at the Soudan State park will probably be built in the spring of 2001.

The five operating systems were designed to treat drainage from covered stockpiles (section 4.2.5). The drainage is generally near-neutral, although pretreatment systems help increase drainage pH from W1-D and Seep 1 prior to wetland treatment. These drainages also contain elevated levels of nickel, copper, cobalt, and zinc (Eger and Lapakko, 1991; Eger and Melchert, 1992). These systems are basically overland flow wetlands with some subsurface flow occurs through limestone drains which were built as part of the berms within the wetlands. Treatment efficiency varies for each of the five wetland systems due to the fact that each system treats drainage from a different stockpile area (Table 17). In general, these wetlands have decreased metal loads in the stockpile drainage (Eger et al., 1996, 1997, 2000b; MN DNR, unpublished data).

Based on results from the Dunka Mine site, constructed wetland treatment of near-neutral drainage appears to be a viable treatment system throughout much of the year. However, it is important to note that the wetlands at Dunka are part of an integrated mine waste management system involving dry covers on the stockpiles (section 4.2.5) and a neutralization treatment plant (section 8.8.2). Wetland treatment alone has not been able to consistently meet regulatory discharge limits at Dunka.

Two climatic conditions common to northern Minnesota limit the extent to which constructed wetlands can be used to treat problematic mine drainages. First, cold temperatures typical during winter months severely limit biological activity. Second, constructed wetland treatment failures are common when flow rates fluctuate widely or are too high. Consequently, elevated flow rates in the spring (i.e. snow melt) occurring immediately after an extended period of cold winter temperatures may render constructed wetlands ineffective for a substantial period of time.

Due

Table 17. Five wetland treatment systems located at the Dunka Mine site near Babbitt, MN have treated near-neutral drainage from stockpiles with variable results.

Wetland Treatment System	Parameter of Concern	Influent (mg/L)	Effluent (mg/L)
W2D-3D <sup>1</sup>	pH	7.1	7.0
	Cu	0.06	<0.005
	Ni	1.94	0.06
	Co	0.02	<0.005
	Zn	0.06	<0.005
W1D <sup>2,4</sup>	pH	7.2	7.4
	Cu	0.03	<0.005
	Ni	0.90	0.13
	Co	0.01	0.02
	Zn	0.02	<0.005
Seep 1 <sup>2,4</sup>	pH	6.9	7.2
	Cu	0.27	0.06
	Ni	7.24	3.34
	Co	0.19	0.06
	Zn	0.94	0.41
Seep X <sup>2</sup>	pH	7.1	7.2
	Cu	0.41	0.14
	Ni	1.82	0.99
	Co	0.09	0.03
	Zn	0.57	0.35
EM-8 <sup>3</sup>	pH	7.2	7.1
	Cu	0.03	0.01
	Ni	1.62	1.04
	Co	0.01	<0.005
	Zn	0.05	0.04

<sup>1</sup>average influent concentrations monitored from 1992-1994, average effluent concentrations for 1992-1998

<sup>2</sup> average concentrations from July 1995 (wetland brought online) through 1999

<sup>3</sup> average concentrations from 1998 (wetland brought online) through 1999

<sup>4</sup> "Influent" monitoring stations represent the effluent from limestone pretreatment cells intended to raise drainage pH prior to entering the W1D and Seep 1 wetlands.

to these climatic factors, a constructed wetland system in northern Minnesota may only be effective for six months out of the year. Thus, constructed wetland systems should be used as one part of an integrated system, as at Dunka.

## 7.2. Permeable Reactive Barriers

Permeable reactive barriers represent an alternative method for treating contaminated ground waters. In the most general sense, a permeable reactive barrier consists of a reactive material installed within an aquifer in such a way that contaminated ground water flows through it (Figure 5). The reactive material induces chemical reactions, which result in contaminant removal from the ground water. Two detailed reports have been published that summarize permeable reactive barrier technology (US EPA, 1998; Yin and Allen, 1999).

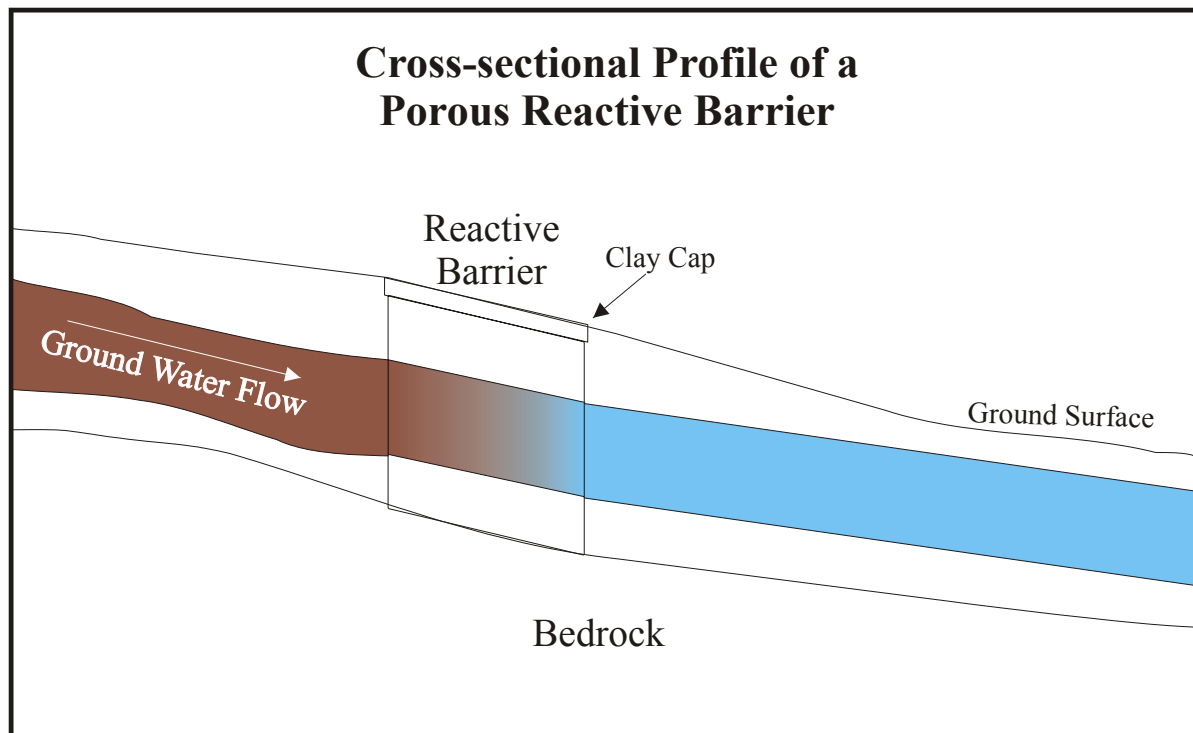


Figure 5. Cross-sectional view of a permeable reactive wall used to passively treat contaminated ground waters. Color is used to represent gradual improvement in contaminated (brown) ground water quality.

The mechanisms by which permeable reactive barriers remove contaminants vary depending upon the type of application. Acidic drainage from mine wastes is usually treated using a biological sulfate reduction system. Sulfate reducing bacteria consume dissolved sulfate, converting it to hydrogen sulfide (reaction 14). Metal removal occurs when the biologically produced sulfide reacts with metals in solution to form insoluble metal sulfides (reaction 15):



where,  $\text{CH}_2\text{O}$  generically refers to an organic carbon source and  $\text{Me}^{2+}$  represents a dissolved metal.

Adsorption is another mechanism by which metals can be removed from ground waters contaminated by mining activities. In addition to adsorption to organic substrates in sulfate reducing barriers, laboratory-scale tests have also been conducted on the ability of various inorganic absorbents to remove metals from contaminated water. Column experiments have shown that arsenic removal can be achieved using basic oxygen furnace oxide (a by-product of steel manufacturing; McRae et al., 1999), zero valent iron (McRae et al., 1999; Su and Puls, 2001), activated alumina (McRae et al., 1999), and hydrous ferric oxide (Martin and Kempton, 1999). Hydrous ferric oxide (Martin and Kempton, 1999) and zero valent iron (Astrup et al., 2000; Phillips et al., 2000) were also able to remove hexavalent chromium from solution. Recent research in the area of mesoporous molecular sieves (nanostructures with high adsorptive capacities and specificity) may also have an application in permeable reactive barriers for a wide variety of heavy metals and mercury (Mercier et al., 1999).

Finally, Conca et al. (1999) suggest that phosphate induced metal stabilization can be used in a permeable reactive barrier system to passively remove metals from ground waters. The removal mechanism in this system involves the secondary precipitation of metal phosphate phases in the presence of a poorly crystalline apatite mineral referred to as Apatite II. This system is particularly effective at removing U, Pb, Cd, and Zn from solution.

#### 7.2.1. Factors Affecting Permeable Reactive Barrier Performance

Several site-specific factors affect the performance of a permeable reactive barrier. Drainage chemistry, ground water flow patterns and velocity, and site accessibility must be considered when designing a permeable reactive barrier. Careful selection of reactive materials is another critical factor in reactive barrier performance. Selected materials must be reactive enough to handle contaminant loads. However, they must not react so quickly that they are rapidly consumed. Materials used in a reactive barrier must also be permeable enough to transmit ground water flow at the site for an extended period of time. Finally, materials must be readily available and affordable. Although several different reactive barrier technologies exist, subsequent discussion of these factors will focus on treatment of acid rock drainage using sulfate reduction barriers.

Success of a sulfate reducing barrier depends on establishment of sulfate reducing bacteria within the barrier. Sulfate reducing bacteria require an organic substrate, nutrients, near neutral pH, and anoxic conditions for optimum growth. The ideal organic substrate should be fresh and contain short carbon chains with single bonds. Numerous organic materials have been tested. Those that have shown some success include leaf mulch (Waybrant et al., 1995, 1997; Benner et al., 1997, 1999, 2000; McGregor, 1999, 2000), wood wastes (Blowes et al., 1995; Waybrant et al., 1995, 1997; Benner et al., 1997, 1999, 2000), animal by-products (Waybrant et al., 1995; Canty, 1999a,b, 2000), and municipal wastes (Benner et al., 1997, 1999, 2000; Waybrant et al., 1995, 1997;). The use of these particular organic materials is primarily based on laboratory experiments conducted by Waybrant et al. (1997). Partially decomposed straw has also been used as an organic substrate for sulfate reducing bacteria in a reactive barrier with limited success (Bechard et al., 1991; 1995).

One problem inherent to sulfate reducing reactive barrier systems is that the permeability of the barrier decreases over time as the organic substrate decomposes. This loss of permeability may restrict ground water flow to the point where it is forced around the reactive barrier rather than through it. To avoid this problem, the organic substrate must be thoroughly mixed with a material with a relatively high hydraulic conductivity, such as pea gravel. Any layering of hydraulic conductivities within the reactive barrier may result in channeling, which reduces residence time and diminishes effectiveness.

The overall hydraulic conductivity of the reactive mixture may be very sensitive to the substrate-gravel ratio. Therefore, permeability tests should be conducted prior to installation to determine the gravel requirement for a specific substrate and ground water flow. Hydrologic modeling can be used to determine appropriate target hydraulic conductivities for the reactive mixture. Benner et al. (1997) found that the hydraulic conductivity of the wall must be an order of magnitude greater than that of the surrounding aquifer to achieve good flow distribution within a reactive barrier.

Strongly acidic environments typical of acid rock drainage do not favor sulfate reducing bacteria. To avoid this problem, limestone is usually added to the organic substrate to elevate the solution pH within the barrier. Oxygen will also inhibit the activity of sulfate reducing bacteria. Consequently, a protective cap (e.g. clay) should be placed across the upper section of the barrier to minimize oxygen ingress.

### 7.2.2. Evaluation of Reactive Barrier Systems

Field evaluation of reactive barrier systems typically involves the use of monitoring wells and piezometers. Contaminant removal can be qualitatively estimated by the water chemistry of samples collected from monitoring wells placed both up and down gradient from the reactive barrier. Monitoring wells and piezometers may also be installed within the barrier or at different depths. Careful planning and installation makes it possible to track ground water flow and identify zones of maximum contaminant removal relative to barrier placement. These data are often used as input for geochemical models.

Geochemical models are used to describe geochemical interactions and mass transport occurring within a reactive barrier. Application of reactive transport models is used to meet two important objectives. First, models can be used as a design tool. Quantification of potential loss or accumulation of mass within the barrier and estimation of long-term permeability of the substrate are critical factors to consider during the reactive material selection phase of barrier design. Second, modeling is used to evaluate reactive barrier performance and to provide insight into geochemical mechanisms controlling contaminant removal.

### 7.2.3. Construction Costs Associated with Reactive Barriers

Construction costs for permeable reactive barriers will vary widely depending on the size of the barrier needed, barrier design, and reactive materials used. Very few operational-scale examples of permeable reactive barriers used to treat mining impacted ground water were found in the literature. Probably the most relevant example was the reactive barrier installed at the Nickel Rim Mine tailings impoundment near Sudbury, Ontario. Materials and installation for this reactive barrier were estimated to cost US \$30,000 (Yin and Allen, 1999). No other cost estimates were found in the literature.

### 7.2.4. Potential Application of Reactive Barrier Technology in Minnesota

Results from laboratory and field studies suggest that permeable reactive barrier technology can be used to remove sulfate (Waybrant et al., 1995, 1997), metals (Blowes and Ptacek, 1992; Waybrant et al., 1995, 1997; Blowes et al., 1998; Conca et al., 1999; Martin and Kempton, 1999; McRae et al., 1999; Mercier et al., 1999), halogenated organics (Gillham and O'Hannesin, 1992; 1994; Yabusake et al., 2001), and phosphate (Ptacek et al., 1994; Baker et al., 1996, 1997, 1998) from contaminated ground waters. In addition to these, field-scale studies have demonstrated the use of permeable reactive barriers to treat ground water contaminated with acid rock drainage (Bechard et al., 1991, 1995; Blowes and Ptacek, 1994; Blowes et al., 1994, 1995; Benner et al., 1997, 1999, 2000; Canty, 1999a, b, 2000; McGregor et al., 1999, 2000) and seepage from landfills (Greenwald et al., 1999). This technology can also be incorporated into septic system designs to prevent nitrate contamination of local ground water (Robertson and Cherry, 1995).

Permeable reactive barriers using sulfate reduction technology may be an attractive strategy to treat certain ground waters contaminated with acid rock drainage. One of the advantages of this strategy is that the reactive wall is underground, eliminating the need to create anaerobic conditions for sulfate reducing bacteria. The in-ground location of a permeable reactive barrier system makes it less visible from the surface, and therefore, more aesthetic than other treatment systems. Finally, it is less expensive than typical ground water “pump and treat” strategies, because the reactive barrier is installed within the ground water flow path. These advantages make application of permeable reactive barriers particularly attractive at abandoned or inactive mine sites and possibly the toe of tailings impoundments.

However, there are two main disadvantages to this strategy that must be addressed. Permeable reactive barrier systems are not feasible at all sites. In order for a permeable reactive barrier system to be successful, the contaminated ground water must flow through the barrier. Local hydrogeology must facilitate movement of ground water through the barrier at a rate at which it can be treated. If subsurface features allows ground water to flow around the barrier, this strategy will be ineffective. Consequently, detailed knowledge of the local geohydrologic setting is required if this strategy is to be considered.

This problem may prove critical for mining applications in northern Minnesota. Bedrock in northern Minnesota is shallow, with only a few feet of overburden (i.e. glacial till) present in most areas (Olsen and Mossler, 1982). Correct placement of a permeable reactive barrier in this situation would be difficult, and in areas with significant outcropping, impossible without blasting and excavation.

The second disadvantage is that the long-term effectiveness of permeable sulfate reducing barriers has not been demonstrated. There are only a few operational-scale demonstrations reported in the literature. Only three mining-related demonstrations have been cited in this document. Of these three, only one demonstrated the ability to remove metals from ground water for more than three years (Table 18). This is most likely due to the gradual degradation of the substrate, resulting in a loss of reactivity and/or permeability. Since substrate degradation is inevitable, permeable reactive barriers will have a definite, finite lifetime. For example, the lifetime of the Nickel Rim reactive barrier was calculated to be only fifteen years. This is a rather short period of time relative to the potential lifetime of water quality problems associated with acid mine drainage (i.e. hundreds of years). Maintenance costs associated with replacing or overhauling reactive barriers may actually be cost prohibitive in some cases. However, a comparison of long-term costs associated with periodic maintenance of a permeable reactive barrier system to active ground water treatment has not been conducted.

Table 18. The long-term success of permeable reactive barriers have not been established.

Site	Duration of Study	Reference
Nickel Rim Mine, Sudbury, Ontario, Canada	38 months - documented declining reactivity over time.	Benner et al. (2000) Benner et al. (1999) Benner et al. (1997) Blowes et al. (1995) (test cell)
Halifax International Airport, Nova Scotia, Canada	The cells were not very effective. Best results were observed between 252 and 336 days out of >500 days monitored.	Bechard et al. (1995) Bechard et al. (1991)
Ore transfer location, Greater Vancouver Area, Canada	21 months - Acid neutralization and metal precipitation documented	McGregor et al. (2000) McGregor et al. (1999)

### 7.3. Passive Treatment References

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## 8. Active Treatment Systems

Active treatment is an effective strategy to ameliorate problematic drainages, however capital and operational costs make active treatment an expensive alternative. It may also require long-term liability, even beyond the life of the operation. In this section, several active treatment strategies will be reviewed.

The most commonly utilized active treatment system for problematic mine drainages in the United States is a neutralization system. Acid neutralization systems are generally moderately expensive, relatively simple to operate, require no pretreatment or specialized chemicals, and address a wide variety of water quality issues. In addition, acid neutralization systems can be enhanced with chemical precipitation circuits for additional metals removal.

Ion exchange and reverse osmosis systems tend to be used in cases where conventional chemical precipitation fails to meet regulatory discharge limits. These technologies are rarely used as stand alone treatment systems. Rather, circuits are added to the existing systems to enhance removal of target dissolved species. In most cases, there are only one or two dissolved species that cannot be removed using conventional methods.

Biological treatment systems are less expensive than the aforementioned systems. However, they require a high level of technical expertise to operate and maintain. They are particularly beneficial when sulfate concentrations must be reduced to relatively low levels to meet regulatory discharge requirements.

### 8.1 Acid Neutralization Treatment Systems

Active treatment of problematic drainage in the United States typically involves neutralization of acid followed by oxidation and precipitation of iron oxyhydroxides, sedimentation/dewatering, and sludge disposal. Much of the information presented in this section was summarized in a design manual published by the US EPA (1983).

Lime, limestone, caustic soda, and soda ash are examples of chemicals that have been used to neutralize acidic drainage in active treatment systems. Each of these chemicals has specific advantages and disadvantages (Table 19) for use in active treatment systems. Lime neutralization is used the most frequently.

Neutralization is usually followed by aeration to promote precipitation of iron oxyhydroxides. Ferric iron is much less soluble than ferrous iron in a neutral pH range. Therefore, iron removal will be more efficient if ferrous iron present in acidic drainages can be oxidized to ferric iron. This can be accomplished using mechanical or chemical aeration systems. Mechanical aeration is designed to introduce large amounts of oxygen into the drainage and keep iron oxyhydroxide precipitates suspended until reaching a settling basin. Chemical oxidation can be achieved

Table 19. Summary of the advantages and disadvantages of some common neutralization chemicals used for active treatment of acidic drainages.

Neutralization Chemical	Advantages	Disadvantages
Lime	<ol style="list-style-type: none"> <li>1. available in bulk</li> <li>2. low cost/gram acidity neutralized</li> <li>3. efficient neutralization</li> <li>4. raises pH high enough to remove Mn</li> <li>5. simple handling and storage</li> <li>6. useful in high flow situations</li> </ol>	<ol style="list-style-type: none"> <li>1. quicklime must be slaked which increases capital costs and requires daily monitoring</li> <li>2. may absorb moisture during storage</li> <li>3. may contain "grit"</li> <li>4. sludges settle slowly</li> <li>5. large initial capital expenditure</li> </ol>
Limestone	<ol style="list-style-type: none"> <li>1. generally inexpensive</li> <li>2. higher density sludge than lime</li> <li>3. safe and easy handling</li> </ol>	<ol style="list-style-type: none"> <li>1. low solubility</li> <li>2. slow reactivity</li> <li>3. adequate reactivity requires high quality limestone (high Ca, -325 mesh, high specific surface area)</li> <li>4. difficult to achieve an adequate pH for Fe and Mn removal</li> <li>5. easily armored, especially if [Fe]&gt;5mg/L</li> </ol>
Caustic Soda	<ol style="list-style-type: none"> <li>1. highly reactive</li> <li>2. liquid, can be pumped or fed by gravity</li> <li>3. mixes easily with drainage</li> <li>4. raises pH high enough to remove Mn</li> <li>5. produces effluent with low suspended solids, turbidity, and Fe content</li> <li>6. sludges have "acceptable" settling properties</li> </ol>	<ol style="list-style-type: none"> <li>1. expensive</li> <li>2. dangerous to handle</li> <li>3. 50% solution freezes at 54 °F</li> </ol>
Soda Ash	<ol style="list-style-type: none"> <li>1. briquettes easy to handle</li> <li>2. dissolution rate ensures continual treatment</li> <li>3. treatment cells can be arranged in series or parallel to suit needs</li> <li>4. sludge settles and compacts comparable to lime</li> </ol>	<ol style="list-style-type: none"> <li>1. expensive</li> <li>2. limited availability</li> <li>3. only effective for low flow drainages</li> <li>4. only effective for low Fe drainages, since briquettes may become coated with iron</li> </ol>

through the addition of oxidants such as ozone (Sato and Robbins, 2000), hydrogen peroxide, and potassium permanganate.

Acid neutralization treatment methods produce large volumes of sludge that must be handled and disposed of in an environmentally appropriate manner. Sludges are composed of iron and other metal oxyhydroxides, gypsum and other sulfate salts, and unused neutralization chemicals (e.g. lime, calcium carbonate etc.). Settling and dewatering of sludges can be difficult. However, the addition of polyelectrolytes (water soluble, high molecular weight, charged organic polymers) improves sludge settling rates. Settling units vary from large earthen basins to mechanical clarifiers to filtration units (Table 20). Sludge handling and dewatering tend to be the most costly aspects of neutralization treatment systems. Engineered basins offer a low cost alternative. However, mechanical clarifiers and filtration units offer more control over the final sludge characteristics.

Sludge must be disposed of in a basin or landfill. On-site disposal is less expensive, however it requires a large amount of open space. It is, therefore, advantageous to produce sludges with high solids contents. Numerous conventional and patented processes are used to produce high density sludges (Kuyucak, et al., 1995; Murcock et al., 1995; Orava et al., 1995; Aube and Payant, 1997; Poirier and Roy, 1997; Aube, 1999; Aube and Zinck, 1999; Zinck and Aube, 2000; Zinck and Griffith, 2000). Although each process may have a unique component, they all incorporate a circuit for recycling the sludge into a neutralization reactor. The recycled sludge provides alkalinity to neutralize AMD as well as solid surfaces to promote precipitation of metal oxyhydroxides.

Table 20. Summary of the sludge dewatering methods used in coal mining (adapted from US EPA, 1983).

Method	Sludge Density (weight % solids)
Centrifugation	8 - 64
Vacuum filtration	9 - 35
Pressure filtration	9 - 26
Porous drying beds	15 - 25
Drying lagoon	12 - 20
Mechanical clarifier	1 - 7
Lagoon or settling pond	0.5 - 4.5

An alternative method to produce a more innocuous sludge is known as the “Ferrite Process” (Choung et al., 2000; McKinnon et al., 2000). Ferrites are magnetic iron oxide minerals that incorporate divalent metals into their crystal lattice,  $MFe_2O_4$ . They can be produced by heating aerated sludges to approximately 65 EC. The sludge must have a ferric to ferrous iron ratio of approximately 1.75, which can be attained by adding  $FeSO_4 \cdot 7H_2O$ . The optimum pH is between 9 and 11.

Ferrites containing trace metals can be magnetically separated from the sludge, resulting in a less hazardous sludge. A potential drawback to the Ferrite Process is that high levels of calcium in the sludge will result in coprecipitation of calcite and/or gypsum. These precipitates interfere with ferrite crystal growth. Consequently, pH adjustments are made using NaOH and excess calcium must be removed prior to ferrite production. Calcium removal involves additional expense. However, the commercial value of ferrites may offset the cost.

## 8.2. Chemical Precipitation Systems

In some systems, the lime treatment system may be followed by an additional circuit to precipitate dissolved species as gypsum, sulfides, or ferric iron co-precipitates. In high sulfate drainages, the lime-treated effluent tends to be supersaturated with respect to gypsum ( $CaSO_4$ ). Excess sulfate can be removed from solution by adding seed crystals of gypsum, which provide active surfaces to catalyze precipitation (Smit, 2000). Precipitated gypsum is thickened, filtered, and removed as a waste product, or recycled as seed crystals.

Additional metal removal can be accomplished by taking advantage of the relatively low solubility of metal sulfides. The addition of sodium sulfide to the lime-treated effluent promotes metal sulfide precipitation. As with the lime treatment circuit, a polymer can be added to improve settling of fine-grained sulfides, which are usually removed using gravity filtration.

Ferric iron co-precipitation typically occurs prior to the neutralization step (Lapakko, 1993; Ericksen et al., 1999; Aube and Stroiazzo, 2000). This process can be used to remove dissolved species (i.e. molybdenum and arsenic) that do not precipitate well using standard neutralization methods. Ferric iron is added to the raw drainage in the form of ferric sulfate. The pH of the solution is adjusted to 4.5 to optimize co-precipitation. Solids are removed by standard techniques and the clarified water can be sent on to the neutralization circuit for additional metals removal and polishing before discharge to the environment.

Finally, a unique chemical precipitation treatment system, in which heavy metals are encapsulated in a silica structure, is being developed (Mitchell and Atkinson, 1995; Mitchell and Wheaton, 1999; Rybock et al., 1999; Mitchell et al., 2000). The primary reagent is a proprietary calcium/silica-based powder, called KB-1<sup>TM</sup>. It can be applied dry or in a slurry, similar to typical neutralization reagents. The removal mechanism takes place in phases initiated by different reagent components. One component raises the pH to precipitate metals. The second component polymerizes, incorporating metals into a silica matrix. Within this silica matrix,

metals precipitate as relatively coarse, dense particles, creating a sludge that is chemically stable under normal environmental conditions.

### 8.3. Ion Exchange Technology

Ion exchange involves the interchange of ions between a solid medium and aqueous solution. The solid medium typically consists of monovalent ions electrostatically attracted to charged functional groups on an inert, polymeric substrate (i.e. resin). Di- and trivalent ions in solution are more strongly attracted to the charged functional groups than monovalent ions. Consequently, di- and trivalent ions replace monovalent ions on the resin, releasing the monovalent species to solution. Water softening is the most common example of ion exchange, where dissolved calcium and magnesium ions exchange with sodium ions on the ion exchange resin.

The long-term effectiveness of an ion exchange resin is determined by the density of functional groups, and therefore the concentration of monovalent ions, on the resin. Once all the monovalent ions on the resin have been replaced, the ion exchange resin will be rendered ineffective. However, ion exchange resins are easily regenerated by periodically back flushing the system with a strong acid, base, or salt solution. Waste water produced by regeneration must also be treated and disposed of properly. The higher the dissolved solids concentration, the more frequently the exchange resins will need to be regenerated.

Ion exchange resins have been shown to remove trace metals and anions from problematic mine drainages. Dissolved di- and trivalent metal ions are removed from solution and exchanged for monovalent ions, typically sodium or hydrogen. Dissolved anions, such as sulfate and arsenate, can also be removed using anion exchange resins. However, high calcium and sulfate concentrations typical of most mine drainages result in the need for frequent resin regeneration. This potential drawback has been avoided in more complicated treatment systems, where a pretreatment step removes some of the dissolved solid load prior to the ion exchange step.

#### 8.3.1. The GYP-CIX Process

Several ion exchange processes have been developed to treat mine drainages. The GYP-CIX process has been particularly promising in the treatment of drainages with elevated calcium and sulfate levels (Bowell, 2000). It uses a series of different ion exchange resins to remove both cations and anions from low acidity mine water without pretreatment (Robertson and Rohrs, 1995). In the first stage, calcium is removed by a strong acid cation exchange bed. This results in a strongly acidic solution, which is passed through an anion exchange unit to remove sulfate and arsenate. The anion exchange unit releases hydroxide ions, which neutralizes the solution. The cation exchange bed can be regenerated using sulfuric acid, which is neutralized with lime to precipitate gypsum and metal hydroxides. Similarly, the anion resin is regenerated with a lime solution, which also precipitates gypsum. These gypsum sludges are similar to those produced by typical neutralization treatment systems and can be disposed of in the same manner.

The GYP-CIX process has been used to treat water from the Berkeley Pit, Butte, MT with encouraging results (Table 21; Bowell, 2000).

### 8.3.2. BIO-FIX Beads

In the late 1980s, the US Bureau of Mines developed a porous polymeric bead to remove metals from mining-related waste waters (Bennett and Jeffers, 1990; USBM, 1990; Bennett et al., 1991; Jeffers et al., 1992). Nonliving biomass (i.e. sphagnum moss) was immobilized in a polysulfone matrix to create a biomass-foam immobilized extractant (BIO-FIX) bead. Laboratory and field tests showed that the nonliving biomass in BIO-FIX beads effectively adsorbed heavy metals commonly found in mine drainages (Bennet and Jeffers, 1990; USBM, 1990; Bennett et al., 1991; Jeffers et al., 1992).

One of the advantages of BIO-FIX beads is that they are easy to handle and can be used in conventional processing equipment without producing sludge. They are less cumbersome compared to living biomass treatment systems because they do not require additional nutrients or special conditions. Since the biomass is incorporated into a synthetic matrix (i.e. bead), separating the metal-laden biomass from the treated water can be accomplished easily.

Laboratory tests have established the stability of BIO-FIX beads for a wide range of pH values, flow rates, and treatment volumes (Bennet and Jeffers, 1990; Bennett et al., 1991). High metal removal efficiencies were demonstrated in four field-scale tests using two different treatment approaches (Table 22). While maintenance requirements for these tests were generally low, high suspended solid loads resulted in plugging problems (Jeffers et al., 1992). Although the laboratory and field tests have demonstrated effective metal removal in the pH range of 3 - 8, it is not certain how effective metal removal would be in a strongly acidic mine drainage (Bennett et al., 1991; Parker and Robertson, 1998). It should also be noted that BIO-FIX bead treatment does not provide alkalinity or increase drainage pH.

### 8.3.3. Zeolites and Selective Chelating Resins

Not all ion exchange materials consist of polymeric resins. Naturally occurring zeolites also exhibit cation exchange characteristics (Leppert, 1990; Gussmann et al., 1991; Vos and O'Hearn, 1993; Schultze et al., 1994; Riveros, 1995; Gilbert et al., 1999). The crystal structure of natural zeolites produce negative surface charges that attract cations. Structural differences between zeolites result in variable affinities for different metal ions. Therefore, zeolites should be selected based on site-specific drainage water chemistry and metal removal needs. Advantages of using zeolites as ion exchangers include high physical integrity and loading capacity. However, substantial desorption of sodium, potassium, calcium or other metals into solution may prove to be a major disadvantage. Another disadvantage of natural zeolites is that in practice they cannot be regenerated and must be replaced regularly.

Table 21. The GYP-CIX process resulted in elevated pH and decreased levels of Ca, Mg, Fe, Mn, Cu, Zn, and SO<sub>4</sub> in waters from the Berkeley Pit, Butte, MT (from Bowell, 2000).

Concentrations (mg/L, unless noted)	Untreated	GYP-CIX
pH (su)	2.7	8.0
Ca	490	50
Mg	420	20
Na	70	50
Fe	1100	<0.1
Mn	182	<0.1
Cu	186	<0.1
Zn	550	<0.1
SO <sub>4</sub>	8000	200

Table 22. Two different treatment approaches have been used to demonstrate metal removal by BIO-FIX beads in four field-scale tests (Bennet and Jeffers, 1990; USBM, 1990; Bennett et al., 1991).

Parameter	Percent Removal	Treatment Approach	Met Discharge Limits?
Cd	90	low-maintenance troughs	U
Co	94	3-column circuit	U
Cu	99	3-column circuit; low-maintenance troughs	U
Fe	>83	3-column circuit; low-maintenance troughs	in 2 out of 3 field tests
Mn	>97	3-column circuit	U
Ni	96	3-column circuit	U
Pb	18	low-maintenance troughs	

Zn	>77	3-column circuit; low-maintenance troughs
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Selective chelating resins and synthetic zeolites have also been investigated as ion exchange materials. However, these materials tend to be degraded at low pH typical of most mine drainages. Furthermore, iron interferes with their ion exchange capacities. One example of a chelating resin that may have application to mine drainage treatment is the Octolig<sup>®</sup> treatment system (Metre-General Inc., 2000). Metal immobilization occurs via chelation by ligands bound to silica gel (resin). These ligands are reported to be highly selective for metals without retaining “benign” ions such as sodium, potassium, or calcium. This selectivity results in effluent metal concentrations in the parts per billion level.

The Octolig<sup>®</sup> resin is insoluble and stable at a range of pH values from 0.5 to 10.5 and temperatures from 0EC to 80EC (Metre-General Inc., 2000). It comes in packed columns from 12 to 42 inches in diameter. The 42 inch columns can treat up to 25 gpm of water. For larger flows, the columns are arranged in parallel. Pretreatment precipitation or filtration circuits can also be added, particularly if the drainage contains high levels of heavy metals. Field demonstrations of the Octolig<sup>®</sup> treatment system have taken place at a Colorado gold mine, the Berkeley Pit in Montana, the Summitville mine in Colorado, and the Leadville drainage tunnel in Colorado (Metre-General Inc., 2000; Appendix 7).

#### 8.4. Membrane Filtration

Reverse osmosis can be applied to a wide variety of water purification needs. Commonly, it is used to desalinate seawater and brackish water to produce potable water. It has also been used to treat problematic mine drainages (Sastri and Ashbrook, 1976; Blackshaw et al., 1983; Awadalla and Hazlett, 1992; Stewart et al., 1997; Berg and Arthur, 1999; Bowell, 2000; HW Process, 2000). Membrane filtration techniques purify water by forcing water under pressure through membranes that are impervious to certain dissolved species. Several membrane filtration techniques, as determined by membrane pore size, have been used world-wide. The exact technique used depends upon water usage requirements (Table 23).

Table 23. Summary of membrane filtration techniques based on operation parameters and water usage.

Filtration Technique	Pore Radius (nm)	Operating Pressure (MPa)	Potential Applications for Problematic Mine Drainages
Reverse osmosis	<1.5	2.7 - 10.3	removal of mono- and divalent salts
Nanofiltration	1.5 - 2.5	1.0 - 2.7	removal of divalent salts
Ultrafiltration	2.5 - 50.0	0.35 - 1.4	removal of colloids and fine particles

Microfiltration	100 - 10000	0.04 - 0.35	filtration of fine particles
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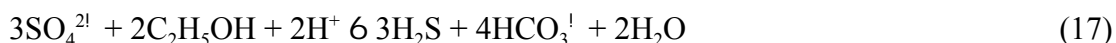
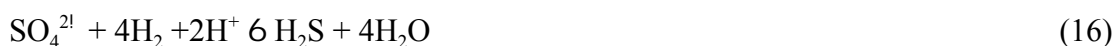
Reverse osmosis separates mono- and divalent salts from water through the application of pressure that exceeds the osmotic pressure of the drainage. Flow through the membrane will be proportional to the net applied pressure and the natural osmotic pressure. Dissolved solids are retained in a brine, which requires special disposal (e.g. lime neutralization).

Advantages of reverse osmosis include removal of both dissolved inorganic and organic contaminants, a simple modular configuration, minimal space requirement, and continuous operation. However, water chemistry typical of most mine drainages may lead to several problems. High levels of dissolved species (i.e. iron, calcium, and sulfate) can result in; (1) high osmotic pressures, which lower the effective driving force for separation, (2) deterioration of product water quality, (3) shortened membrane life, and (4) precipitation of sparingly soluble salts (i.e. gypsum, metal hydroxides), which can lead to membrane fouling and increased resistance to permeation.

Although the lower pH typical of many mining effluents may improve efficiency, it also contributes to the deterioration of the membrane. Reverse osmosis is also an expensive technology relative to more conventional chemical precipitation techniques (section 8.6). Consequently, reverse osmosis is usually only used as a pre- or post-treatment step to remove contaminants not removed by more conventional means.

### 8.5. Biological Sulfate Reduction Treatment Systems

Biological sulfate reduction systems represent one alternative to typical neutralization systems used to treat contaminated ground waters (Nakamura, 1988; Dvorak et al., 1991; Boreck et al., 1995; de Vegt et al., 1997; Diaz et al., 1997; Rowley et al., 1997; de Vegt et al., 1998; Dijkman et al., 1999; Kolmert et al., 1999). These systems utilize sulfate reducing bacteria to consume dissolved sulfate and acid, producing hydrogen sulfide gas and alkalinity. In order for sulfate reduction to occur, however, a dissolved species capable of donating electrons (i.e. energy source) must be available. Both hydrogen gas (reaction 16) and simple organic compounds (reaction 17) can be used for this purpose and both types of systems have been used to treat mining effluents (Boreck et al., 1995; Diaz et al., 1997; Dijkman et al., 1999).



Hydrogen is generally used in systems with high sulfate loads. However, hydrogen must be produced on site by cracking methanol or using a natural gas steam reformer. Organic electron donors are preferred for smaller sulfate loads.

Contaminated ground water must be pumped to a surface biological reactor, where sulfate reduction takes place. Although systems vary, they generally proceed in a series of steps beginning with sulfate reduction. Alkalinity generated during the reaction helps consume acidity in the waste stream, while biologically produced hydrogen sulfide reacts with dissolved metals (reaction 18):

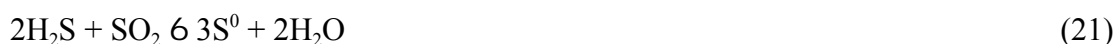


where,  $\text{Me}^{2+}$  represents a dissolved metal. Subsequent steps allow metal sulfide precipitates to settle out of solution. Some systems utilize a “chemical loop,” which enables selective precipitation of metals (Nakamura, 1988; Dvorak et al., 1991; de Vegt et al., 1997, 1998).

As a final step, some biological treatment systems recover elemental sulfide as a saleable product. This is accomplished by oxidizing either gaseous or dissolved hydrogen sulfide to elemental sulfur. One documented method utilizes aerobic sulfur bacteria to oxidize excess dissolved hydrogen sulfide to elemental sulfur (reaction 19):



A second method utilizes the two step Claus Process (reactions 20 and 21),:



where hydrogen sulfide is stripped from metal sulfides in the sludge.

Biological sulfate reducing treatment systems have the potential to effectively remove sulfate and metals from ground waters contaminated by mining wastes. Biological treatment systems are advantageous over traditional neutralization systems because; 1) lower levels of sulfate, iron, and other metal can be achieved, 2) it produces a relatively innocuous sludge that settles rapidly, and 3) organic waste materials (e.g. municipal sludge) can be used as the electron donor. Sulfate reducing treatment systems have an additional advantage in that saleable elemental sulfur can be recovered, making these systems more economical.

Despite these advantages, biological sulfate reducing treatment systems can be fragile. Sulfate reducing bacteria require an anaerobic and pH neutral environment to effectively reduce dissolved sulfate to hydrogen sulfide. Most ground waters contaminated by mining wastes are anaerobic, satisfying the first requirement. However, these waters tend to be acidic rather than neutral. Acidic influents require neutralization prior to entering the sulfate reducing reactors. The addition of neutralization circuits to treatment system design increases capital and operating costs, as well as creating additional sludge disposal issues.

Although ground waters are generally anaerobic, they must be pumped to the surface for treatment in biological reactors. Once contaminated ground waters reach the surface, it can be difficult to maintain anaerobicity, particularly in large reactors. Oxygen leaks into the system will lead to decreased effectiveness, and possibly, system failure. Additional problems can be encountered when excess hydrogen sulfide builds up within the system. Hydrogen sulfide gas trapped within the treatment bed may result in gas-lock and diminished flow. It may be possible to bleed hydrogen sulfide out of the system. However, the odor may be problematic, especially in populated areas.

Successful biological treatment of contaminated ground water produces an effluent with low sulfate and metal loads. However, this effluent may not be suitable for discharge into the environment. Low dissolved oxygen levels and high biological oxygen demand (BOD) in treated waters can have detrimental effects on streams and other receiving waters. Therefore, effluents should be aerated prior to discharge. Dissolved contaminants leached from waste materials (e.g. paper sludges) used as organic substrates in the treatment system also present potential problems in the effluent. These contaminants may not be suitable for discharge into the environment and additional treatment steps will be required for their removal. Materials used for organic substrates should be evaluated carefully, since aeration may be an ineffective method of removing contamination from waste materials.

#### 8.6. Assessment of Active Treatment Systems

The goal of an active treatment system is to increase drainage pH and decrease metal concentrations to meet local and national regulatory discharge limits. Active treatment system requirements are almost always site-specific. Drainage chemistry is one of the most important variables that must be considered when choosing an active treatment system. Factors such as pH, sulfate concentration, dissolved metals, and oxygen levels are used to determine which treatment system will best fit the needs for a particular drainage (Table 24).

Table 24. Optimum drainage chemistry for various active treatment systems.

Treatment Option	pH	Sulfate (mg/L) <sup>1</sup>	Dissolved Metals	Oxygen Levels
Acid neutralization/ chemical precipitation	acidic	> 10,000	low or high	high
Ion exchange	acidic to neutral	500 - 10,000	low	
Reverse osmosis	acidic to neutral	500 - 10,000	low	
Biological sulfate reduction	near-neutral	500 - 10,000	high	anoxic

Other variables include site accessibility, the amount of available space for process equipment and sludge disposal, climate, and local discharge limits. In some situations, it may be necessary to use a combination of active treatment systems to optimize treatment and/or meet discharge limits. Finally, sludge produced as a result of metals precipitation must meet Toxic Contaminant Leaching Protocol (TCLP) standards in order to be disposed of in a municipal landfill. If sludges do not meet these standards, they must be treated as hazardous waste, greatly increasing the cost of handling and disposal.

#### 8.7. Cost Comparison of Different Active Treatment Systems

Operational costs of active treatment systems vary depending on the choice of system, chemicals and materials required, sludge handling and disposal, and site-specific considerations such as transportation costs. Additional precipitation circuits to remove metals also add to construction and operational costs. However, these costs can be reduced if an economically valuable product, such as elemental sulfur, pyrite, or a specific metal, can be recovered from the waste stream.

Based on information collected from several North American treatment plants (Appendix 6), operational costs for acid neutralization plants are on the order of several dollars per thousand gallons treated (Table 25). One operation reported that chemicals accounted for approximately two thirds of their operational costs (Aziz and Ferguson, 1997). Very little information was available for design, capital, and construction costs for neutralization plants. Based on information provided for a single operation, the Argo Tunnel, CO, these costs are estimated at \$5.8 million dollars.

Ultimately, the costs associated with an ion exchange treatment system depend upon the dissolved solids load in the drainage relative to the drainage quality that must be achieved (Table 26). Due to the regeneration frequency required to remain effective, ion exchange systems tend to be cost prohibitive as a lone treatment for problematic mine drainage. However, in certain situations, ion exchange systems may be cost effective. For example, ion exchange may be cost effective if the drainage in question contains high levels of metals that are difficult to remove by neutralization alone. In this case, an ion exchange circuit placed prior to neutralization can reduce the quantity of neutralization chemicals and associated expense that would otherwise be required to meet discharge limits. Ion exchange should also be considered when very low water quality standards must be met. In this situation, ion exchange is used as a polishing step after neutralization but prior to discharge. Finally, extraction of economically valuable metals may make ion exchange a viable and efficient treatment option.

As with ion exchange systems, the cost associated with a membrane filtration treatment system (i.e. reverse osmosis) depends upon the drainage chemistry relative to the quality of water that must be achieved to meet discharge limits (Table 26). No current mine drainage applications of reverse osmosis were identified during the course of this study. Past applications were used

to treat a specific mine drainage issue and were discontinued, presumably due to the high cost of

Table 25. Average operational costs of selected North American acid neutralization treatment operations (see Appendix 6 for references and contact information).

Operation	Process	Average Flow (gpm)	Estimated Annual Operation Costs (\$ 10 <sup>6</sup> /year)	Operation Cost Estimate (\$/1,000 gal)
Argo Tunnel, CO	caustic soda	320	1.2	7.13
California Gulch Superfund Site, Leadville drainage, CO	caustic soda	1600 <sup>1</sup>	1.0	1.61
California Gulch Superfund Site, Yak Tunnel, CO	caustic soda	400 <sup>2</sup>	nr <sup>3</sup>	nr
Eagle Mine, CO	lime	160	nr	nr
Summitville, CO	hydrated lime	1300 <sup>1</sup>	2.5	9.51
Equity Silver, British Columbia	quicklime	nr	1.1	4.73

<sup>1</sup> Assumes operational period of approximately nine months, excluding winter

<sup>2</sup> Plant operates three to four days a week

<sup>3</sup> nr = not reported

Table 26. Several active treatment system requirements increase operating costs. Qualitative comparison of these variables indicates that ion exchange and reverse osmosis systems are the most expensive systems to operate.

Variable	Acid Neutralization/ Chemical Precipitation	Ion Exchange	Reverse Osmosis	Sulfate Reduction
Pretreatment for solids	na <sup>1</sup>	optional	required	required
Reagent consumption	high	moderate	low	low
Nutrient consumption	na	na	na	high
Energy consumption	moderate	moderate	high	low
Sludge disposal	high	na	na	low
Brine disposal or treatment	na	high	high	na

<sup>1</sup> na = not applicable

operation. High costs are largely due to high energy consumption and the need to dispose of brine wastes.

In general, biological treatment systems are the least expensive, but the most complicated systems to operate (Table 26). Only a few operational-scale examples of biological treatment systems for mining impacted ground water are reported here. Projected capital and annual operating costs for the “Biosulphide Process” treatment system installed at the Britannia Copper Mine, British Columbia were \$2.2 million and \$1 million, respectively (Rowley et al., 1997). The more recently applied THIOPAQ Process, used at Budelco Zinc Refinery, The Netherlands and a pilot plant at Kennecott’s Bingham Canyon, Utah Copper Mine, is estimated to cost US \$340/t of sulfate removed. Depending on the current market value, operating costs can be decreased by adding a circuit to recover elemental sulfur for resale.

Cost comparisons between the various active treatment strategies are difficult to quantify. This is due, in part, to a lack of adequate cost analysis data. However, the validity of any such comparison would be questionable on the basis that these systems are employed to meet site-specific treatment needs. Comparisons are further complicated by the fact that one or more of these systems may be used in series to optimize conditions for a particular system or to meet local discharge requirements.

Qualitative comparisons can be made based on the various system requirements (Table 26). Pretreatment circuits and high energy consumption often result in high operational costs (Table 27). Consequently, reverse osmosis and ion exchange systems tend to be more expensive than acid neutralization alone. Sulfate reduction circuits are generally the least expensive because they require very little energy input and reagents can be regenerated on-site. In relative terms, annual operating costs for ion exchange and reverse osmosis systems may be as much as 30 - 50% higher than a conventional acid neutralization system, while operation of a sulfate reduction system may cost as much as 50% less.

## 8.8. Application of Active Treatment Systems to Problematic Mine Drainages in Minnesota

### 8.8.1. AMAX/Kennecott Test Site

In 1976, AMAX Exploration, Inc. began construction of an exploration shaft near Babbitt, Minnesota. Six small test piles containing 62,000 tons of low grade copper-nickel Duluth Complex rock taken from an exploration shaft, were constructed. In 1982, Kennecott Mining Company took over state leases for the prospect. In 1998 Kennecott reclaimed the site with the exception of the six test piles and the settling basin. The MN DNR assumed liability for the test piles and as part of a NPDES permit had to establish an acceptable active treatment system for the effluent from the piles. Treatment objectives were to develop a low maintenance, low cost treatment system that would reduce all metals in the drainage to less than 1 mg/L (Eger et al., 1993).

Table 27. Qualitative cost analysis information indicate that ion exchange and reverse osmosis are the most expensive active treatment options, while sulfate reduction is the least expensive.

Treatment System	Relative Capital Cost	Relative Operating Costs	Estimated Operating Costs (\$/1000 gal)
Acid neutralization	high	1	1.6-10 <sup>1</sup> 0.3-0.7 <sup>3</sup>
Ion exchange	moderate	1.3 - 1.5	0.3-0.8 <sup>3</sup>
Reverse osmosis	high	1.3 - 1.5	1-4.6 <sup>2,3</sup>
Sulfate reduction	low	0.5	highly variable

<sup>1</sup>see Table 25

<sup>2</sup> information obtained from HW Process Technologies, Inc.

<sup>3</sup>Federal Remediation Technology Roundtable (excludes sludge disposal)

Laboratory tests were conducted by the MN DNR to compare the effectiveness of lime, magnesium hydroxide, and sodium hydroxide at neutralizing acid and removing trace metals (Eger et al. 1993). Tests were designed to determine contact time requirements, sludge production, and dosage requirements for each neutralizing agent. Magnesium hydroxide was selected for field tests based on the following results:

- Easy application,
- Safe handling,
- Maximum pH of 9.5, reducing the possibility of over-treatment,
- Lower mass and volume of sludge per dosage unit,
- Produced the highest density sludge,
- Residual neutralizing capacity available in the sludge,
- Non-hazardous sludge produced.

At the field site, a treatment system designed to be compatible with the existing water collection system was constructed (Eger et al., 1993). The system consisted of a central 3785 L collection sump with a float activated pump. As the sump filled, the float activated pump engaged and discharged about 750 L of treated water to a settling basin. Roughly 570 L remained in the sump after the pump shut off. To maximize contact time between the leachate and the magnesium hydroxide slurry (58% by weight), the alkaline material was added immediately after the sump emptied.

Based on the laboratory results, an initial magnesium hydroxide dose between 1.0 - 2.0 g/L was used (Eger et al., 1993). By the end of the field season, this dose was increased to 2.2 g/L to provide better treatment at high flow rates. Field results generally agreed with the laboratory results. When dosage and contact time (>6hr) were adequate, pH generally exceeded 8.0 and suitable treatment was achieved. However, treatment efficiency decreased, particularly in terms of nickel removal, as flow rates increased above 3.7 L/min. Overall, it was concluded that magnesium hydroxide could effectively neutralize acid and remove metals from problematic mine drainages (Eger et al., 1993).

#### 8.8.2. MN DNR Field Research Site, Hibbing, MN

In 1996, the magnesium hydroxide treatment technology developed at the AMAX/Kennecott site was applied to drainage from field experiments conducted at the MN DNR field research site in Hibbing, Minnesota (Figure 6). Construction cost for the building and treatment system were approximately \$95,000. The treatment plant operates continuously and is designed for a daily maximum flow of 1.5 gpm (Table 28). Approximately 60,000-80,000 gallons of waste water is treated annually at a cost of approximately \$12,000 (MN DNR, unpublished data).

Drainage from field experiments is generally acidic, containing elevated levels of copper, nickel, cobalt, and zinc. Since effluent from the treatment plant discharges into the Hibbing sewer system, it must meet local discharge limits (Table 29). Sludge produced during the treatment process must also meet TCLP limits for landfill disposal.

#### 8.8.3. Dunka Mine Site

Active treatment of problematic mine drainages is currently used at LTV Steel Mining Company's Dunka Mine Site near Babbitt, Minnesota. More than 50 million tons of sulfide bearing rock is stockpiled at this site. The most recent report indicates total flow from the stockpiles ranges from 50 to 200 million gallons per year. Drainage seeping from the stockpiles is generally near-neutral, however, pH values as low as 4.5 have been reported. Despite the neutral drainage pH, copper, nickel, cobalt, and zinc concentrations exceed ambient levels by 10 to 10,000 times.

At present, drainage from these stockpiles is collected and discharged into wetland treatment systems. Outflow from the wetland treatment systems can be pumped to a collection basin for lime precipitation treatment to further reduce trace metal concentrations prior to discharge into Unnamed Creek, and ultimately, Birch Lake. Each year, this system treats approximately three million liters of drainage at a cost of \$200,000 (see A6.2 for description of the process).

Early in 2000, LTV Steel Mining Company announced that they would be closing their operation. In July 2000, LTV proposed re-issuance of their National Pollution Discharge Elimination System (NPDES) permit to allow them to shut down the lime precipitation plant at Dunka. In response, the MN DNR requested that the facilities remain in such a condition that

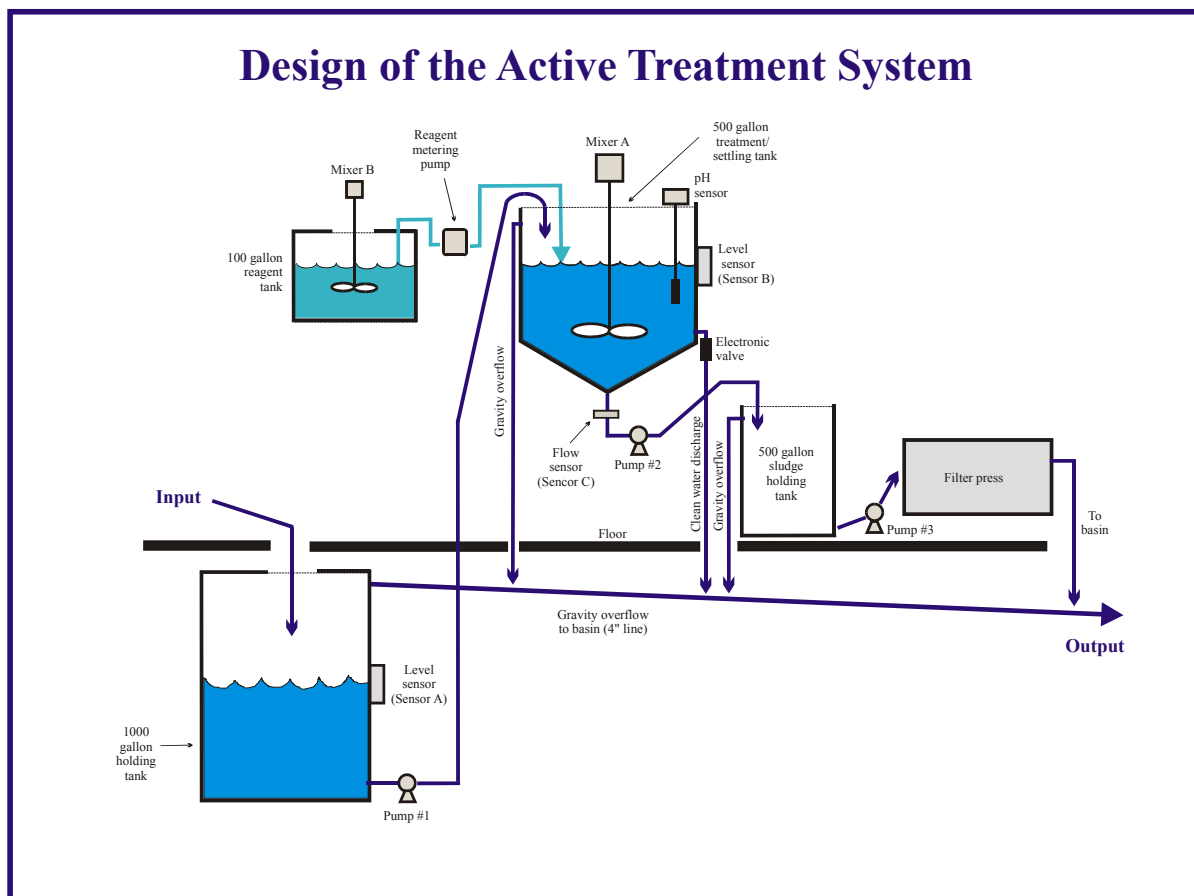


Figure 6. Drainage from field experiments conducted at the MN DNR Hibbing research facility is treated with magnesium hydroxide to neutralize acid and precipitate metal oxyhydroxides.

Table 28. MN DNR Hibbing research site treatment plant specifications.

Source of waste water	mine waste drainage
Flow rate	1.5 gpm
Operating time	24 hours/day, 7 days/week
Mg(OH) <sub>2</sub> dosage	2 .2 g/L
Leachate volume/batch	380 gal
Sludge volume after settling	2% or 8 gal
Sludge mass / 400 gal batch	7.3 lbs or 3.33 kg

Table 29. Leachate treatment specifications for the MN DNR research site treatment plant, Hibbing, Minnesota.

Parameter	Concentration Range in Drainage (ppm)	Concentration Range in Effluent (ppm)	Required Discharge Limit
pH	4.0	7.25-9.0	6.5-8.5
Cu	50 - 150	0.001-0.05	0.5
Ni	100 - 300	0.137-2.1	3.0
Co	10 - 30	0.001-0.1	1.0
Zn	10 - 30	0.001-1.34	4.0

operation of the treatment plant could resume on short notice should the wetland treatment systems fail or if water quality standards in Unnamed Creek are exceeded. Both requests were granted, and the Minnesota Pollution Control Agency reissued LTV's NPDES permit in August 2000 (Permit MN 0042579).

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## SUMMARY AND CONCLUSIONS

### 9. Integrated Case Studies

Whether or not a mine waste produces problematic drainage depends on a wide array of variables that describe the physical, chemical, and mineralogical characteristics of the waste, pre-existing site conditions, climate, and government regulations. Consequently, there is no single, correct approach to environmental mine waste management. A comprehensive mine waste management plan usually involves multiple mitigative strategies to achieve the desired results. Integrated systems, including contingencies to treat impacted waters, are necessary to meet regulatory standards.

Numerous examples of integrated environmental mine waste management approaches could be cited here. However, this section will focus on one example from the Dunka Mine in Minnesota, one existing and one proposed mine in Wisconsin, and a South Dakota mine. The Wisconsin and South Dakota case studies were selected based on geographic, climatic, and regulatory similarities to northern Minnesota. Finally, a list summarizing additional case studies from Canada can be found in Appendix 7.

#### 9.1. Dunka Mine, Babbitt, Minnesota

Although the Dunka Mine near Babbitt, Minnesota is listed in this section, the majority of the information regarding mine waste management strategies employed at Dunka can be found throughout this document (Table 30).

Table 30. Summary of environmental mine waste management activities and research projects at the Dunka Mine, Babbitt, MN.

Activity or Research Project	Section in this document
Limestone treatment beds	1.1.1
Stockpile capping program	4.2.5
Co-disposal of hornfels rock in LTV's taconite tailings basin	3.2.4
Wetland treatment systems	1.1.2, 7.1.4, 7.1.5
Active treatment plant	8.7.1, 8.7.2

Beginning in the 1960's, Duluth Complex rock was excavated and stockpiled at the Dunka site in order to access the underlying iron formation. In the mid-1970s it was discovered that trace metal concentrations were elevated in Unnamed Creek, which drains the mining watershed (Figure 7). The metal sulfide minerals present in the stockpiled Duluth Complex rock were determined to be the source of the metals in the creek. Taconite mining from the Dunka Mine ceased on August 26, 1994. The location of environmental mine waste management activities and research reviewed in this document can be found in Table 30. Additional information is available in Appendix 7.1 and cited references.

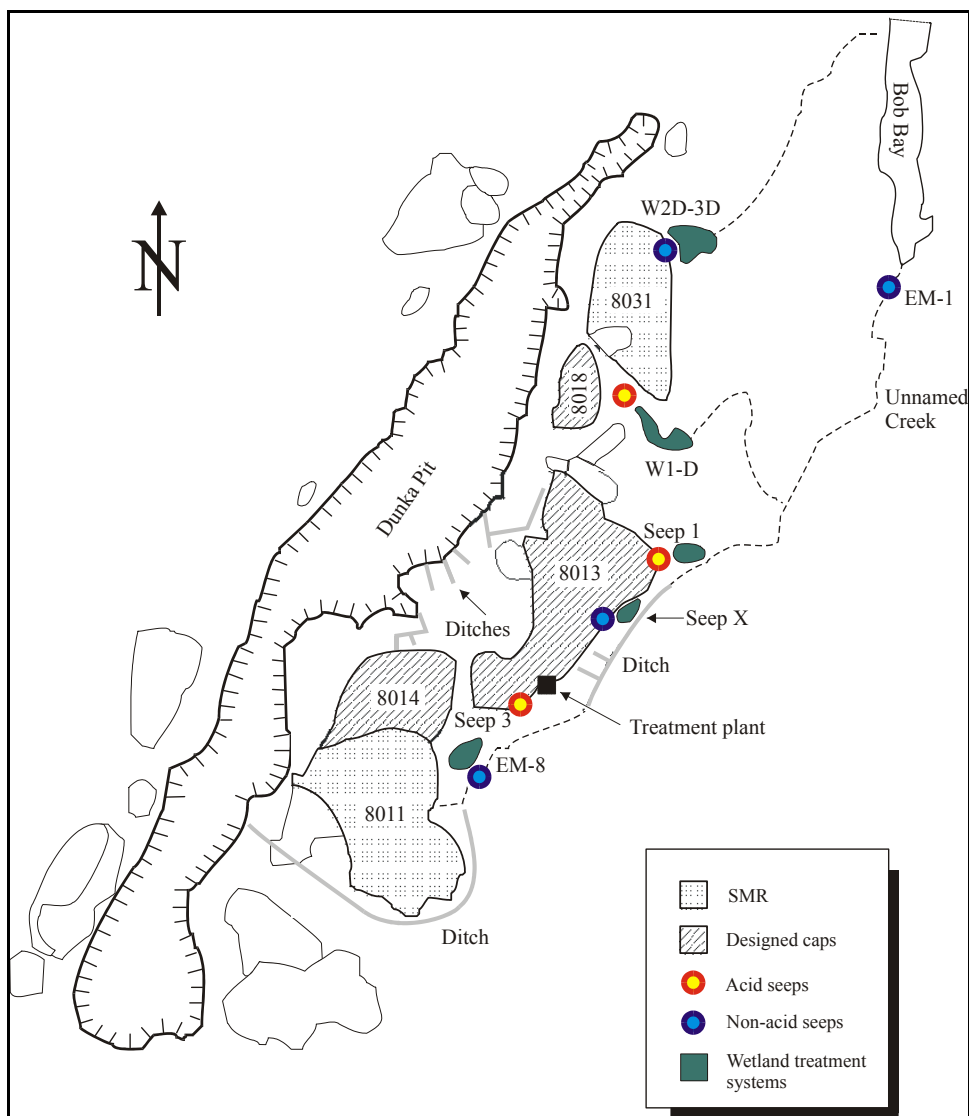


Figure 7. Area map of the Dunka Mine, near Babbitt, MN.

## 9.2. Flambeau Mine, near Ladysmith, Wisconsin

In January 1991, permits were granted to the Flambeau Mining Company to develop a copper/gold/silver mine near Ladysmith, Wisconsin. Approximately 1.9 million tons of ore was recovered from the mine between 1993 and 1997, resulting in a 32 acre open pit (Table 31). The mine plan called for on-site storage of nearly 5 million yd<sup>3</sup> of overburden and potentially reactive waste rock during operations. Upon closure, these materials were backfilled into the open pit and the site was returned to its approximate pre-mining state.

Over time, the backfilled waste rock will be saturated by in-flowing groundwater. Eventually groundwater levels over the pit will return to approximate pre-mining conditions with groundwater flow occurring mainly in the overburden material at average depth to groundwater expected to be less than 20 feet. Similarly, the direction of groundwater flow will return to pre-mining conditions so that flow is predominantly from east to west, toward the Flambeau River which lies about 140 feet from the edge of the pit (Figure 8).

### 9.2.1. Mine Waste Characterization and Storage

During the Mining operation, nearly 5 million yd<sup>3</sup> of waste material was removed from the Flambeau pit. Topsoil was scraped from the surface and stored in a separate stockpile at the northeast edge of the pit. Till, sandstone, and saprolite were stripped and stored in separate cells of the "Type I" stockpile. Site-specific waste rock characterization tests indicated that waste rock should be sorted based on its potential for acid generation as determined by sulfur content. The operational limit for non-acid and acid generating rock was 1% sulfur. That is, waste rock containing less than 1% sulfur was not expected to generate acidic drainage. This was considered to be a conservative estimate.

Table 31. Physical characteristics of the Flambeau open mine pit and operation (Sevick et al., 1998; WDNR, 2001).

Parameter	Magnitude
Extent of mine operation	181 ac
Extent of the pit	32 ac
Pit trend	NE-SW
Pit length	- 2600 ft
Pit width	- 550 ft
Maximum pit depth	- 220 ft
Minimum distance to Flambeau River	- 140 ft



Figure 8. Aerial view of the Flambeau pit seen from the northeast (WDNR, 2001). The Flambeau River can be seen along the upper edge of the photo.

Approximately, 2.7 million yd<sup>3</sup> material was stockpiled in the “Type I” stockpile, which contained roughly 0.63 million yd<sup>3</sup> low sulfur waste rock (Table 32). Surface runoff from the Type I stockpile was collected in settling ponds and treated prior to discharge to the Flambeau River. “Type II” material (> 1% sulfur) was placed in a surface storage facility underlain by a liner and leachate collection system (Table 32). Leachate from the high sulfur stockpile, along with water pumped from the open pit and surface runoff from the crusher/loading area, was treated at the on-site wastewater treatment facility before discharge to the river.

Table 32. Overburden and waste rock were classified based on sulfur content and placed in separate surface storage facilities during operation of the Flambeau Mine (Chapman et al., 1998; Sevick et al., 1998; WDNR, 2001).

Stockpile	Type I	Type II
Amount of material	2.7 million yd <sup>3</sup>	2.1 million yd <sup>3</sup>
Amount of waste rock	0.63 million yd <sup>3</sup>	2.1 million yd <sup>3</sup>
S-content	< 1%	> 1%
Storage area	40 ac	27 ac
Storage facility	unlined surface storage	1' fine grained till 60 mil HDPE liner
Drainage collection	drain to lined settling ponds	lined conveyance to surge ponds
Wastewater treatment	evaporation or pumped to treatment plant	treatment plant: lime neutralization sulfide precipitation filtration

### 9.2.2. Backfill Operation

The backfill strategy was to place the high-sulfide material in the bottom of the pit followed by the low-sulfur waste rock and then replacing the unconsolidated overburden in its approximate original stratigraphic position. Potentially acid generating material returned to the bottom of the pit was expected to be below the recovered water table. By refilling the pit in this manner, and allowing the ground water table to recover, reactive waste rock would be placed in anoxic conditions similar to those that existed before mining began.

Backfilling of the Flambeau Pit began in fall 1996 and continued through 1997 (Foth & Van Dyke, 1997, 1998). Between August and November 1996, high sulfur waste rock was placed in the bottom of the pit in three foot, compacted lifts. Crushed limestone was applied to high sulfur rock to neutralize stored acidity accumulated while the material was stockpiled, as well as potential acidity generated as the ground water table recovers. Laboratory and field studies and oxygen transport calculations were used to determine alkaline addition rates required to neutralize stored and future acidity, respectively (Foth & Van Dyke, 1997; Chapman et al., 1998).

The rate of limestone addition for high sulfur rock that was excavated from the west end of the pit and immediately deposited in the east end ranged from 4.0 - 4.9 lb/ton. Application rates of 20 lb/ton were used for stockpiled high sulfur rock backfill. In practice, limestone was applied to the surface of each waste rock lift prior to its removal from the stockpile (Foth & Van Dyke, 1998). Approximately 1.5 million pounds of crushed limestone was applied to the high sulfur rock during this portion of the backfill operation (Foth & Van Dyke, 1998). After placement of the high sulfur rock and limestone in the pit was completed, backfilling continued with low sulfur waste rock amended with a limited amount of limestone, saprolite, sandstone, and till.

Surface reclamation of the site began in 1998 by regrading the land surface to its approximate original contour. Stockpiled topsoil was spread over the site and vegetated to create grasslands, woodlands, and wetlands. Reclamation activities through 2000 included seeding, plug planting, tree planting, erosion control, mowing, invasive species control, trail construction, and prescribed burning (Flambeau Mining Co., 2001).

### 9.2.3. Environmental and Ground Water Monitoring

According to permit requirements, the Flambeau Mining Company must regularly monitor ground water, backfill pore water, wetland, and Flambeau River water levels and quality in addition to aquatic ecology characteristics (WDNR, 2001). As part of the permitting process, both hydrologic and geochemical modeling were used to characterize the potential impacts of the backfilled pit on the Flambeau River (Prickett et al., 1996; Foth and Van Dyke, 1997; Eng. Tech. Assoc. Inc., 1998). Hydrologic modeling indicated that it would take fifteen years for ground water levels in the backfilled pit to recover (Eng. Tech. Assoc. Inc., 1998). Furthermore, calculations showed that the flux of backfill pore water to the river would be  $6.5 \times 10^5$  times lower than the ten year, seven day, low flow for the river (Flambeau Mining Company, 2001). Consequently, it was assumed that backfill pore water quality would not have a measurable affect on surface water quality. The October 2000 assessment concluded that regional ground water flow, including backfill pore water, is indeed moving toward the Flambeau River as expected (Flambeau Mining Company, 2001).

Ground water quality monitoring in nineteen wells has been conducted on a regular basis. Trends in backfill pore water chemistry indicate a calcium-magnesium-sulfate-carbonate dominated water in equilibrium with calcite and gypsum (Flambeau Mining Company, 2001). This equilibrium condition suggests that stored acidity in the backfill has been neutralized by limestone. Major elements, pH, and alkalinity concentrations have been stable, although iron, manganese, and carbon dioxide levels are higher than anticipated (Flambeau Mining Company, 2001). Concentrations of other metals, including copper, are similar to predicted values. In the October 2000 assessment, it was concluded that backfill pore water chemistry had stabilized and was not affecting Flambeau River water quality.

### 9.3. Richmond Hill Gold Mine, South Dakota

In 1988, South Dakota granted permits to develop the 400 acre Richmond Hill surface gold mine near Lead, South Dakota (Figure 9; Durkin, 1995; Durkin et al., 1998; Duex, 2000). Approximately 8.2 million tons of ore was recovered from a single pit between 1988 and 1993. Gold and silver were produced from three heap leach pads until 1995. During operation, approximately 3.5 million tons of waste rock was placed in the Spruce Gulch waste rock dump.

In 1992, the South Dakota Department of Environment and Natural Resources (DENR) discovered a 200,000 ton stockpile of sulfide ore on top of the Spruce Gulch waste rock dump (Durkin, 1995). Subsequent investigation identified acidic drainage containing elevated metal concentrations coming from the toe of the waste rock pile (Durkin, 1995; Duex, 2000). This led to a series of events including mine shutdown, increased bonds, fines for permit and water quality standard violations, and development of a reclamation plan.

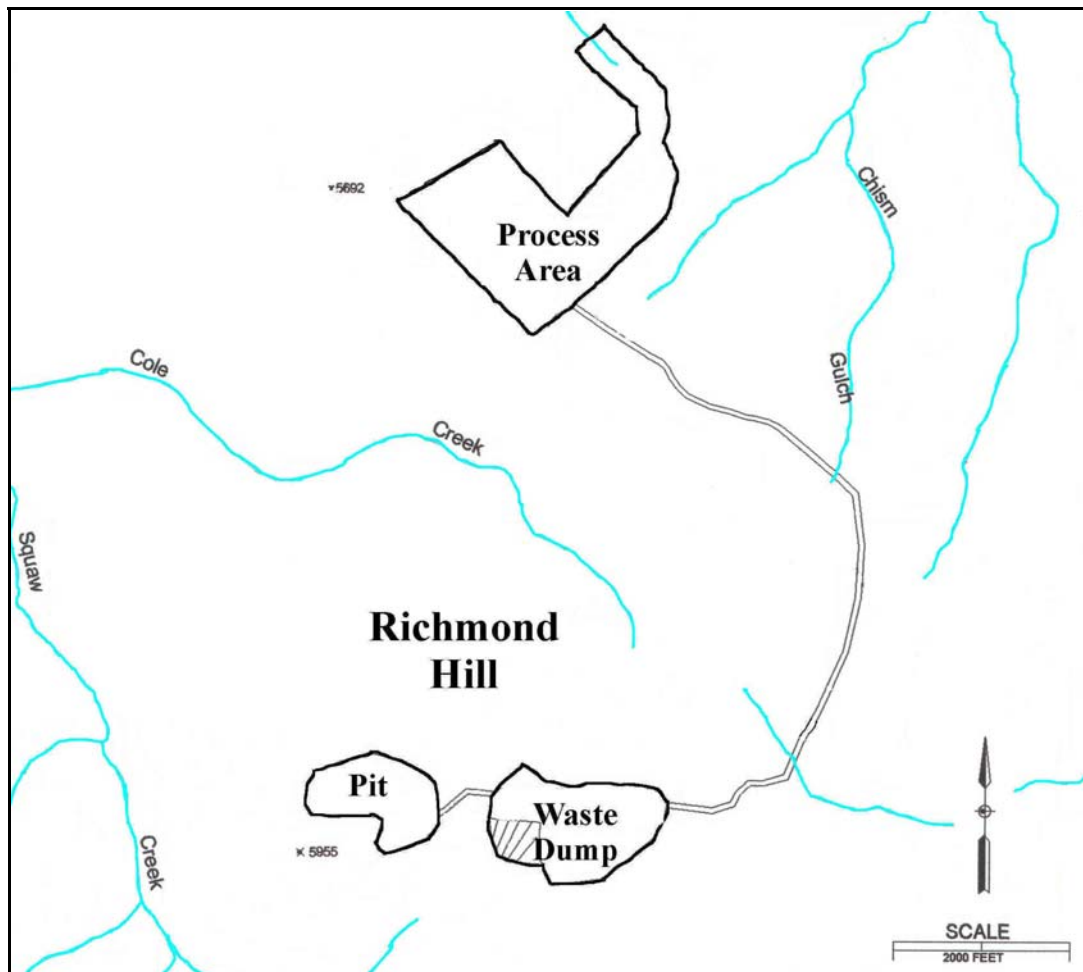


Figure 9. Schematic diagram of the Richmond Hill Mine complex, near Lead, South Dakota. Reactive wastes from the waste dump and process areas were relocated in the mine pit and covered.

### 9.3.1. Short-term Mitigation Measures

Short-term mitigation measures focused on collection and treatment of impacted waters (Durkin, 1995; Durkin et al., 1998). Beginning in the spring of 1992, treatment and retention ponds were constructed to collect drainage from the waste rock depository. An anoxic limestone drain was used for the initial treatment. However, it was quickly rendered ineffective by ferric hydroxide armoring of the limestone. Subsequent treatment in the ponds utilized caustic soda to neutralization acid and precipitate metal hydroxides. After partial treatment, this water was pumped to an 80 million gallon storm water pond for storage, and ultimately, treatment in the waste water treatment plant. A system of pumping wells was constructed within the retention pond area to collect local impacted ground water within a near surface alluvial/colluvial aquifer.

Several other measures were taken to contain the reactive sulfide ore and inhibit further oxidation. In order to contain further drainage problems, the sulfide ore stockpile was moved to the heap leach pads. Diversion ditches were constructed to prevent unimpacted surface water from entering the depository and becoming contaminated. Since it was likely that some reactive waste rock remained in the depository, lime, limestone, and fly ash were applied to key areas for additional neutralization. Finally, the surface of the depository was sprayed with a semi-sealant to minimize infiltration of precipitation.

### 9.3.2. Closure Plans

The primary goal of the closure plan was to isolate and stabilize the reactive sulfide ore to reduce the potential for long-term environmental risk to surface and ground waters (Durkin, 1995; Durkin et al., 1998; Duex, 2000). To achieve this goal, 2.7 million tons of reactive material from the waste rock depository and heap leach pads was relocated to the former open pit (Figure 9). The remaining, non-acid generating, waste rock from the depository was incorporated into a multilayer cover system designed to minimize exposure to precipitation and oxygen. The backfilled and covered pit is now referred to as the “Engineered Pit Backfill Facility.”

Because the pit lies above the regional ground water table, there was little concern from contamination of deep ground water (Duex, 2000). However, shallow water entered the pit through seeps in the highwalls. Chimney drains were constructed near these seeps to divert shallow ground water around the backfill waste.

The backfill operation was accomplished by placing reactive waste rock in the bottom of the pit in three foot, compacted lifts (Durkin, 1995; Duex, 2000). Before emplacing the cover system, the backfill surface was graded to 3:1 to 6:1 slopes. The cover system, from bottom to top, consisted of 6 inches crushed limestone, an 18 inch low permeability layer, a 54 inch thermal barrier/drainage layer made of non-acid generating waste rock and limestone, and 6 inches of topsoil and vegetation. The low permeability layer, constructed by mixing non-acid generating waste rock crushed to minus ½ inch with high sodium bentonite clay, was placed in two nine

inch lifts. Reported permeabilities ranged from  $1 \times 10^{-9}$  to  $8 \times 10^{-9}$  cm/s, which met the target maximum permeability of  $1 \times 10^{-7}$  cm/s (Duex, 2000).

Performance monitoring focuses mainly on water movement into and out of the backfilled waste (Durkin et al., 1998; Duex, 2000). However, pore gas (i.e. oxygen and carbon dioxide) levels, settling, temperature, ground and surface water quality, and aquatic ecology characteristics are also monitored. Monitoring results through 1999 indicate that the multilayer cover system does limit infiltration of precipitation and oxygen into the backfilled waste (DENR, 2000; Duex, 2000). Furthermore, surface and ground water quality and aquatic ecological monitoring suggest that the Engineered Pit Backfill Facility has not adversely impacted the surrounding environment (Durkin et al., 1998; Holm et al., 1999; DENR, 2000).

Water quality at the toe of the former waste dump was markedly improved by removal of the waste rock, although soluble acid salts remain in the colluvium and near-surface fractured bedrock of Spruce Gulch. Consequently, water treatment is still required prior to discharge. It is anticipated that these soluble contaminants will rinse from the impacted colluvium, and that long-term water treatment will not be required in this area.

Reclamation of the three heap leach pads was approved by the DENR in June 1996 (DENR, 1996). Closure issues related to the leach pads included acid drainage from leach pad three, and near neutral pH contaminant mobility from leach pads one and two. Remediation at leach pad three included blending limestone with the spent ore. The limestone was mined on-site, crushed to minus 1/4 inch, and blended at a ratio of 3:1 neutralization potential:acid potential, including an additional factor for the effective calcium carbonate efficiency due to the grain size distribution of the limestone. All three pads were capped with a low-permeability clay cap similar to the Engineered Pit Backfill Facility. Double liner, leak detection, collection, and recover systems are required by South Dakota state law below all heap leach pads. The spent ore from the site was reclaimed in place on the existing leach pad liners to limit potential migration of pad effluent solutions to ground water.

Intermittent water treatment is currently required at the site. Water is collected year-round from the three leach pads and several sources in the former waste dump area and stored in a lined pond. The water treatment plant is operated on an intermittent basis as required by the site water balance.

### 9.3.3. Environmental and Ground Water Monitoring

According to permit requirements, the backfilled pit must be monitored for water movement, pore gas content, and temperature as well as surface and ground water quality and aquatic health (i.e. Squaw Creek). Performance monitoring as of 2000 indicate that the cover system is stable and effective (Holm et al., 1999; DENR, 2000; LAC Minerals, 2001).

- Neutron probe measurements indicate that the clay cap has limited infiltration of water into the underlying waste, although some infiltration has occurred.

- Oxygen levels in the waste pore spaces remain elevated in local ore zones. However, this is thought to be due to diffusive or convective transport from the base of the backfill along contacts rather than through the cap.
- Temperature profiles were relatively stable and did not indicate the occurrence of accelerated sulfide oxidation.
- Vegetation has become well established on the covered surface, and no cracks or serious erosion problems were visible.

A series of ground water monitoring wells were installed around the former pit, waste dump, and process area. Surface water monitoring sites were established at the toe of the waste dump and at several points along Squaw Creek. Monitoring results indicate that ground water quality trends are improving and collection systems are adequate to prevent impacts to surface water (Duex, 2000). Metal concentrations in the pad effluent are low enough that passive treatment may become a feasible long-term treatment option. A pilot-scale passive treatment system was installed in 1999 (DENR, 2000). Brief summaries of surface and ground water quality appear in Appendix 7.3. All monitoring data is considered public and is available from the DENR (Durkin et al., 1998; Nelson, 2001).

#### 9.4. Crandon Deposit (Proposed Mine), Crandon, Wisconsin

The Crandon Deposit is a massive zinc-copper ore body located near Crandon, Wisconsin. The Nicolet Minerals Company has submitted permit applications to develop an underground mine to remove approximately 55 million tons of sulfidic ore (WDNR, 2001). The WDNR is in the process of reviewing the project. Since the project is only in the proposal stage, a detailed review in this document is not yet warranted. However, a few aspects of the proposed mine waste management plan are listed below (Hockley et al., 1998; Sevick et al., 1998; WI DNR, 2001).

##### Tailings Management

- De-pyritization of the tailings
- Underground disposal of pyrite tailings
  - incorporation of pyrite in a cement-based paste backfill
  - submergence of the paste backfill upon closure
- Placement of depyritized tailings in the Tailings Management Area
  - base liner (described from bottom to top)
    - fine soil layer (1 ft thick)
    - GCL
    - 60-mil HDPE geomembrane
  - leachate collection system
  - dry cover system (described from bottom to top)
    - site soils placed directly over waste
    - fine soil layer (1 ft thick)
    - GCL
    - 60-mil HDPE geomembrane

- drainage layer
- vegetative layer

#### Water Management

- Grouting of underground mine workings to minimize ground water inflow
- Reflooding underground mine workings upon closure
- Active treatment of contaminated ground water from the mine
  - lime neutralization and precipitation
  - sulfide precipitation
  - filtration
  - reverse osmosis
  - evaporation
  - discharge to a soil adsorption system
- Recycling of water from the Tailings Management Area to the mill

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## **Volume 2 of 2**



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Final Report to the Minerals Coordinating Committee  
June 30, 2001**

**Minnesota Department of Natural Resources  
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June 30, 2001

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A1.3. Summary of US, Canadian, Australian, and Swiss efforts to compile mitigative strategies for environmental metal mine waste management.

Table A1.3.1. Summary of methods to prevent or control the generation of acidic drainage.

Table A1.3.2. Summary of treatment methods for acidic drainage

## APPENDIX 2

### Selective Handling of Reactive Mine Wastes

Table A2.1.	Bibliographic annotations of literature regarding the backfilling of open pits using tailings and waste rock .....	A2.3
Table A2.2.	Bibliographic annotations of literature regarding the backfilling of underground mine workings .....	A2.4
Table A2.3.	Bibliographic annotations of literature regarding the co-disposal of tailings and waste rock .....	A2.5
Table A2.4.	Selected Monitoring Well Data Collected by LTV Steel Mining Company .....	A2.6
Table A2.5.	Selected Tailings Basin Seepage Data Collected at LTV Steel Mining Company .....	A2.12



Table A2.1. Bibliographic annotations of literature regarding the backfilling of open pits using tailings and waste rock. Italicized numbers refer to reference numbers for the Reclamation Section's literature database (Papyrus).

Reference	Study Scale <sup>1</sup>	Waste Material		Treatment or Amendment Strategy			
		Waste Rock	Tailings	Alkaline Addition	Flooded	Dry Cover	Paste or HDS
Hill & Benson (1999)	F	U					
Cravotta et al. (1994)	O	U		U			
Hockley et al. (1997)	O	U		U	U		
Chapman et al. (1998)	R, O	U		U	U		
Sevick et al. (1998)	M, L, O	U		U	U		
Dorey et al. (1999)	O	U		U	U		
Morin & Hutt (2000)	O	U		U	U		
Flambeau Mining Co. (2001)	O	U		U	U		
Peterson et al. (1998)	M	U			U		
Wickham et al. (2001)	M	U			U		
Lewis-Russ et al. (1997)	O	U			U	U	
Durkin (1995)	O	U		U		U	
Kerr et al. (1999)	O	U		U		U	
Duex (2000)	O	U		U		U	
Aiken et al. (1999)	M, L, F	U		U	U	U	
Jakubick et al. (1997)	O	U		U	U	U	
Orava et al. (1997)	R	U	U	U	U		
Cincilla et al. (1997)	R		U				U
Dahlstrom (1997)	R, F		U				U
Verburg et al. (2000)	L		U		U		U

HDS = high density solids

<sup>1</sup> R= review, L= laboratory, M= modeling, F= field, O= operation

Table A2.2. Bibliographic annotations of literature regarding the backfilling of underground mine workings. Italicized numbers refer to reference numbers for the Reclamation Section's literature database (Papyrus).

Reference	Study Scale <sup>1</sup>	Issue Addressed	
		Problematic Drainage	Stability
Dahlstrom (1997)	R		
Luppnow & Dorman (1998)	R		
Hockley et al. (1998)	M	U	
Bertrand et al. (2000)	L	U	
Verburg et al. (2000)	L	U	
Walker (1993) #1174	R		U
Zou (1997)	L		U
Gay & Constantiner (1998)	L		U
Lord & Liu (1998)	L, F		U
Ouellet et al. (1998)	L, O		U
Bernier et al. (1999)	L		U
Chen et al. (1999)	L		U
Qiu et al. (1999)	L		U
Amaratunga (2000)	R		U
Landriault et al. (2000)	R, O		U
Kump (2001) #3911	O		U
Burnett et al. (1995)	O	U	U
Helms & Heinrich (1997)	L, O	U	U
Scheetz & Schantz (1999)	L	U	U

<sup>1</sup> R= review, L= laboratory, M=modeling, F= field, O= operation

Table A2.3. Bibliographic annotations of literature regarding the co-disposal of tailings and waste rock. *Italicized numbers refer to reference numbers for the Reclamation Section's literature database (Papyrus).*

Reference	Study Scale <sup>1</sup>	Hydrology	Removal Mechanism	Disposal Methods	Economic Aspects
Nicholson et al. (1989) <i>#1131</i>		U			
Nichol et al. (2000)	F	U			
Lefebvre & Gelinas (1995)	O	U			
Wilson et al. (2000b)	O	U			
Lapakko & Eger (1981)	L		U		
Iwasaki et al. (1982) <i>#621</i>	L		U		
Lapakko et al. (1983)	L,F		U		
Eger et al. (1984)	L,F		U		
Lapakko et al. (1985) <i>#3875</i>	L,F		U		
Lapakko et al. (1986a) <i>#2849</i>	L,F		U		
Lapakko et al. (1986b) <i>#2838</i>			U		
LTV, 1996. Closure Plan	O			U	
Mehling et al. (1997)	O			U	
Williams (1997)				U	U
Sellgren & Addie (1998)				U	U
Johnson et al. (1995)	L				
Fortin et al. (2000)	L				
Lamontagne et al. (2000)	L				
Poulin et al. (1996) <i>#3896</i>	L				
Wilson et al. (2000a)	O				

<sup>1</sup> R= review, L= laboratory, M=modeling, F= field, O= operation

Table A2.4. Selected Monitoring Well Data Collected by LTV Steel Mining Company (Clark, personal communication).

### HORNFELS WELL MONITORING

H-1	Well at E side of hornfels right side of hornfels as viewed from dike								
Date	pH (s.u.)	Cu (µg/L)	Ni (µg/L)	Co (µg/L)	Zn (µg/L)	SO <sub>4</sub> (mg/L)	Ca (mg/L)	Mg (mg/L)	T. Fe (mg/L)
29-Mar-1995	7.7	1	17	2	20	153	37	44	5.1
17-May-1995	7.6	2	8	2	20	190	60	50	14.0
16-Aug-1995	7.4	4	15	9	20	186	61	45	7.0
14-Nov-1995	7.5	11	25	12	70	177	48	40	5.7
22-Mar-1996	7.6	<1	4	9	27	160	70	45	3.4
20-Jun-1996	7.5	13.7	16.4	20.6	121	138	65.5	40	16.0
15-Aug-1996	9.8	3.8	14.2	9.7	16	152	7.8	12.6	4.4
5-Dec-1996	ns								
18-Mar-1997	ns								
11-Jun-1997	ns								
7-Aug-1997	ns								
20-Nov-1997	ns								
28-May-1998	ns <sup>1</sup>								
2-Sep-1998	7.3	6.2	40.6	10.1	52	162	96.2	67.5	5.3
20-Nov-1998	8.2	26.4	23.4	5.7	<10	85.1	101	98.1	41.1
25-Feb-1999	7.6	24.9	17.6	6.2	68	135	61.4	35.1	23.0
11-May-1999	9.50	96.4	19.6	2.2	41	72.4	6.4	9.7	11.0
16-Aug-1999	7.21	<1	5.1	2.7	23	200	77.5	46.2	4.3
15-Nov-1999	7.20	<1	<2	<1	<10	171	52.2	59.5	0.20
14-Feb-2000	7.50	6.5	9.9	1.2	<10	138	3.5	2.0	7.94

H-2	Well at S end of hornfels	1706.16'	center (south) of hornfels as viewed from dike
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## HORNFELS WELL MONITORING

Date	pH (s.u.)	T. Cu (µg/L)	T. Ni (µg/L)	T. Co (µg/L)	T. Zn (µg/L)	SO4 (mg/L)	Ca (mg/L)	Mg (mg/L)	T. Fe (mg/L)
29-Mar-1995	8.8	1	13	2	10	66	14	40	3.8
17-May-1995	8.9	5	17	4	30	74	20	40	10.0
16-Aug-1995	7.8	6	12	2	10	112	39	34	14.0
14-Nov-1995	7.8	9	14	2	30	99	28	28	21.0
22-Mar-1996	8.4	2	2	2	20	79	23	33	10.0
20-Jun-1996	7.5	12.4	13.1	2.1	31	166	67.6	36	15.0
15-Aug-1996	9.7	<1	<2	2.6	<10	151	38.6	33.6	1.74
5-Dec-1996	9.1	7.3	10.4	<1	11	155	41.7	27.7	9.25
18-Mar-1997	ns								
11-Jun-1997	8.9	2	6.8	3.8	14.7	89.1	21.0	26.5	16.8
7-Aug-1997	9.2	10.6	243	3.2	15	77.5	6.4	20.6	25.2
3-Dec-1997	9.4	2.5	<2	<1	19	61.0	<1	11.7	30.7
28-May-1998	9.9	<1	8.6	4.4	60	81.6	3.2	6.4	11.7
2-Sep-1998	7.4	2	50.5	17	181	162.0	56.3	47.6	13.2
20-Nov-1998	8.7	23.8	22.3	5.4	<10	83.8	101.0	98.7	30.6
25-Feb-1999	10.0	9.7	5.5	2.3	29	67.7	3.7	12.3	15.3
11-May-1999	9.38	125	25.3	2.5	17	85.8	6.7	5.5	13.5
16-Aug-1999	7.33	<1	<2	<1	<10	95.9	14.0	33.6	0.69
15-Nov-1999	7.60	<1	<2	<1	<10	177.0	50.9	58.4	0.23
14-Feb-2000	7.61	3.4	<2	2.1	<10	176.0	41.8	46.8	0.14

H-3	Well at W side of hornfels				left side of hornfels (as viewed from dike)				
Date	pH (s.u.)	T. Cu (µg/L)	T. Ni (µg/L)	T. Co (µg/L)	T. Zn (µg/L)	SO4 (mg/L)	Ca (mg/L)	Mg (mg/L)	T. Fe (mg/L)
29-Mar-1995	ns								
17-May-1995	7.0	3	19	3	30	729	120	190	25.0
16-Aug-1995	7.1	2	40	7	40	825	225	118	11.0
14-Nov-1995	ns								
22-Mar-1996	7.6	<1	14	3	32	140	90	48	10
20-Jun-1996	ns								
15-Aug-1996	ns								
5-Dec-1996	ns								
18-Mar-1997	ns								
11-Jun-1997	ns								
7-Aug-1997	ns								
20-Nov-1997	ns								
28-May-1998	ns <sup>1</sup>								
2-Sep-1998	9.9	80.8	29.6	2	20	108	15.8	8.8	9.24
20-Nov-1998	8.8	<1	23.0	5.7	41	83.8	13.2	34.6	30.6
25-Feb-1999	10.0	18.8	11.5	1.6	24	114	6.0	4.5	5.1
11-May-1999	9.45	4.3	<2	1.2	17	67	3.4	3.8	3.75
16-Aug-1999	7.95	2.7	<2	<1	<10	102	43.0	21	12.1
15-Nov-1999	7.60	<1	<2	<1	<10	188	50.2	57.4	0.23
14-Feb-2000	7.50	15.9	21.2	<1	<10	54.5	4.9	<0.5	2.78

## HORNFELS WELL MONITORING

SE-1 Seep at south end of basin									
Date	pH (s.u.)	T. Cu (µg/L)	T. Ni (µg/L)	T. Co (µg/L)	T. Zn (µg/L)	SO4 (mg/L)	Ca (mg/L)	Mg (mg/L)	T. Fe (mg/L)
29-Mar-1995	7.3	1	1	1	<10	230	76	75	1.6
17-May-1995	7.3	<1	<1	6	10	336	90	90	0.6
16-Aug-1995	7.3	1	6	2	<10	259	75	98	1.0
14-Nov-1995	7.3	1	1	2	<10	220	56	68	1.6
22-Mar-1996	7.3	<1	<1	5	17	296	115	109	2.4
20-Jun-1996	7.2	12.6	<1	4.4	12	277	106	110	2.3
15-Aug-1996	7.3	<1	<2	7.8	<10	279	98.6	101	1.84
5-Dec-1996	7.5	<1	<2	1.3	<10	179	62.0	55.8	1.57
18-Mar-1997	7.6	<2	9	2.2	<10	175	48.4	37.1	0.08
13-May-1997	7.6	<1	<2	2.3	<10	162	45.0	36.5	0.07
7-Aug-1997	7.8	<1	<2	<1	<10	132	32.6	34.4	<0.03
20-Nov-1997	7.7	<1	<2	<1	<10	154	41.6	37.3	0.07
28-May-1998	7.6	3	<2	3.8	12	151	39.4	37.8	0.15
19-Aug-1998	7.6	<1	<2	2.5	13	133	40.7	39.6	0.08
9-Nov-1998	7.7	<1	13.6	2.6	<10	147	39.9	38.8	0.10
17-Mar-1999	8.2	<1	<2	1.5	19	188	38.4	40.5	0.06
14-May-1999	7.95	<1	<2	<1	17	249	90.6	109	1.73
25-Aug-1999	7.46	8.5	<2	3	13	234	88.3	105	1.59
10-Nov-1999	7.10	<1	<2	2.7	<10	225	79.5	89.1	0.98
17-Feb-2000	7.10	<1	<2	6.6	<10	261	80.9	93.7	1.35

SE-2 Seep at south end of basin									
Date	pH (s.u.)	T. Cu (µg/L)	T. Ni (µg/L)	T. Co (µg/L)	T. Zn (µg/L)	SO4 (mg/L)	Ca (mg/L)	Mg (mg/L)	T. Fe (mg/L)
29-Mar-1995	7.9	<1	2	<1	<10	227	61	68	0.1
17-May-1995	8.0	1	<1	1	<10	202	60	50	0.1
16-Aug-1995	7.9	2	7	1	<10	182	53	44	0.4
14-Nov-1995	7.9	2	1	<1	1	159	34	40	0.1
22-Mar-1996	8.1	2	<1	4	15	185	80	60	13.0
20-Jun-1996	7.9	12.0	<1	1.6	<10	203	71	64	0.35
15-Aug-1996	8.2	<1	<2	2.8	<10	234	70.7	71.0	0.23
5-Dec-1996	7.7	<1	4.1	1.3	<10	199	59.1	68.6	0.46
18-Mar-1997	8.2	<2	10	1.4	<10	330	83.0	91.5	5.05
13-May-1997	8.2	<1	<2	<1	<10	273	74.8	95.5	0.23
7-Aug-1997	8.2	<1	3.2	<1	<10	226	65.9	70.8	0.22
20-Nov-1997	8.3	<1	4.4	<1	<10	172	58.9	59.2	<0.03
28-May-1998	8.2	2.5	<2	1	<10	177	52.6	51.9	0.27
19-Aug-1998	8.0	<1	<2	<1	13	170	58.4	57.0	0.55
9-Nov-1998	7.9	<1	<2	<1	<10	164	53.0	53.7	2.38
17-Mar-1999	8.0	<1	<2	1	<10	162	44.3	42.7	0.35
14-May-1999	7.92	<1	<2	<1	14	155	43.7	43.4	0.08
25-Aug-1999	7.64	2.5	<2	<1	<10	156	43.9	49.7	0.14
10-Nov-1999	7.50	<1	<2	<1	<10	149	47.1	51.8	0.13
17-Feb-2000	7.20	<1	<2	2.6	<10	154	44.3	48.0	0.19

## HORNFELS WELL MONITORING

SE-3	Seep at south end of basin								
Date	pH (s.u.)	T. Cu (µg/L)	T. Ni (µg/L)	T. Co (µg/L)	T. Zn (µg/L)	T. SO4 (mg/L)	Ca (mg/L)	Mg (mg/L)	T. Fe (mg/L)
29-Mar-1995	8.2	<1	<1	1	<10	230	58	63	<0.1
17-May-1995	8.4	<1	<1	1	<10	192	150	140	<0.1
16-Aug-1995	8.4	1	<1	1	<10	193	54	46	<0.4
14-Nov-1995	8.3	1	1	<1	1	171	38	48	<0.1
22-Mar-1996	8.3	<1	<1	3	14	200	80	65	5.0
20-Jun-1996	8.5	13.6	<1	1.6	<10	213	74	67	2.30
15-Aug-1996	8.4	5.8	<2	3.9	<10	229	68.4	70.0	0.91
5-Dec-1996	7.5	<1	2.7	<1	<10	408	81.8	125.0	0.07
18-Mar-1997	8.3	<2	9.5	2.5	32	260	70.5	75.2	1.41
13-May-1997	8.3	3.2	<2	<1	<10	235	64.1	69.8	0.15
7-Aug-1997	8.2	<1	<2	<1	<10	184	56.3	56.7	0.06
20-Nov-1997	8.2	<1	<2	<1	<10	187	61.2	57.9	<0.03
28-May-1998	8.2	<1	<2	3.8	<10	162	44.8	43.1	2.81
19-Aug-1998	7.8	1.9	<2	<1	15	141	48.2	46.9	4.24
9-Nov-1998	7.9	<1	<2	<1	<10	157	45.9	46.5	0.13
17-Mar-1999	7.9	<1	<2	1	<10	191	46.6	47.4	0.52
14-May-1999	7.90	<1	<2	<1	15	164	48.2	49.1	0.05
25-Aug-1999	7.70	3	<2	<1	<10	174	48.9	57.5	0.15
10-Nov-1999	8.30	<1	<2	<1	<10	176	49.2	56.7	0.11
17-Feb-2000	7.80	<1	<2	2.5	<10	166	44.6	52.5	0.28

## HORNFELS WELL MONITORING

NE Seepage Recovery Sump									
Date	pH (s.u.)	T. Cu (µg/L)	T. Ni (µg/L)	T. Co (µg/L)	T. Zn (µg/L)	SO4 (mg/L)	Ca (mg/L)	Mg (mg/L)	T. Fe (mg/L)
9-Jun-1993		2	1	<1	<10				
29-Aug-1993		<1	<1	2	<10				
4-Nov-1993		<1	<1	<1	<10				
8-Feb-1994		1	1	<1	<10				
7-May-1994		1	<1	<1	<10				
16-Aug-1994		7	2	<1	<10				
22-Feb-1995		1	<1	1	<10				
16-May-1995		2	<1	<1	<10				
16-Aug-1995		2	8	<1	10				
31-Oct-1995		1	10	<1	10				
26-Mar-1996		<1	3	1	15				
20-Jun-1996		13	1	<1	<10				
15-Aug-1996		5	<2	3	<10				
12-Nov-1996		<1	6	<1	241				
18-Mar-1997	7.9	6.2	10.3	<1	<10	228	67.3	58.1	0.26
13-May-1997	7.9	2.7	<2	<1	<10	238	69.4	68.3	0.39
7-Aug-1997	7.8	<1	<2	<1	<10	277	79.6	99.9	0.36
20-Nov-1997	7.9	<1	<2	<1	<10	358	94.1	104	<0.03
28-May-1998	8.2	<1	<2	<1	11	241	67.3	71	0.28
19-Aug-1998	7.8	<1	<2	<1	13	200	66.3	71.4	0.25
9-Nov-1998	7.7	<1	<2	<1	<10	211	65.5	62.2	0.30
25-Mar-1999	8.2	<1	<2	<1	19	213	72.3	71.1	0.95
14-May-1999	8.40	<1	<2	<1	19	233	74.4	75.5	0.95
25-Aug-1999	7.79	<1	<2	<1	<10	170	67.7	75.1	0.38
10-Nov-1999	7.8	<1	<2	<1	<10	196	67.9	68.7	0.30
17-Feb-2000	7.3	<1	<2	2.4	<10	207	71.0	76.3	0.48

## HORNFELS WELL MONITORING

NW Recovery Sump									
Date	pH (s.u.)	T. Cu (µg/L)	T. Ni (µg/L)	T. Co (µg/L)	T. Zn (µg/L)	SO4 (mg/L)	Ca (mg/L)	Mg (mg/L)	T. Fe (mg/L)
9-Jun-1993		<1	<1	<1	<10				
29-Aug-1993		<1	<1	<1	<10				
4-Nov-1993		2	<1	<1	<10				
8-Feb-1994		4	5	<1	10				
7-May-1994		1	<1	<1	20				
16-Aug-1994		8	2	<1	<10				
22-Feb-1995		4	1	<1	<10				
16-May-1995		<1	<1	<1	<10				
16-Aug-1995		2	6	<1	10				
31-Oct-1995		1	6	<1	10				
26-Mar-1996		2	<2	<1	20				
20-Jun-1996		1	<1	<1	<10				
15-Aug-1996		<1	<2	2	<10				
12-Nov-1996		<1	3	<1	11				
18-Mar-1997	7.4	<2	9	<1	14	235	60.3	71.0	0.68
13-May-1997	7.9	<1	<2	<1	<10	163	44.8	52.2	0.15
7-Aug-1997	8.3	<1	<2	<1	<10	162	50.5	69.6	0.17
20-Nov-1997	7.4	<1	<2	<1	<10	179	48.5	61.0	0.36
28-May-1998	8.1	3	3.1	1.1	10	178	56.8	65.3	0.18
19-Aug-1998	7.8	<1	<2	<1	12	158	45.4	54.0	0.23
9-Nov-1998	7.9	<1	<2	<1	<10	215	67.6	72.2	0.10
25-Mar-1999	8.7	<1	<2	<1	19	261	67.3	88.0	0.44
14-May-1999	8.03	<1	<2	<1	15	196	49.5	59.5	0.30
25-Aug-1999	7.51	<1	<2	<1	<10	188	54.0	70.9	0.19
10-Nov-1999	7.5	<1	<2	<1	<10	219	50.9	70.7	0.31
17-Feb-2000	7.7	<1	<2	4.9	<10	531	98.9	129.0	5.10

“ns” = no sample

<sup>1</sup> new well installed spring 1998

Table A2.5. Selected Tailings Basin Seepage Data Collected at LTV Steel Mining Company (from Berndt et al. 1999).

<b>LTV-Seep</b> Date	06/13/96	09/11/96	01/22/97	04/09/97	07/16/97	10/09/97	02/04/98	04/08/98	07/23/98	10/07/98	02/10/99
<b>Field Parameters</b>											
Temp (EC)	13	14	11	10	11	14	10	10	10	9	9
Conductivity	1430	1200	1300	1050	1100	1200	1150	900	1350	1100	1250
pH	7.23	7.70	7.30	7.41	7.31	7.45	7.38	7.28	7.22	7.16	7.12
Measured Eh	-48	171	-178	25	2.4	-23	32	117	-38	103	25
Corrected Eh	161	380	33	237	213	186	244	329	174	316	237
<b>majors (ppm)</b>											
Al	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.01	0.01	0.00
Si	11.2	9.3	10.0	10.5	9.3	10.2	10.6	11.1	11.1	12.3	11.4
Fe	2.18	0.09	0.73	1.73	1.20	1.50	2.42	2.49	2.50	2.66	2.70
Mn	1.43	1.23	0.82	0.69	1.00	0.94	0.88	0.91	1.00	0.98	0.99
Sr	0.50	0.24	0.28	0.27	0.33	0.35	0.35	0.38	0.43	0.43	0.40
Ba	0.05	0.02	0.04	0.04	0.04	0.05	0.05	0.05	0.06	0.06	0.05
Ca	96.3	57.0	65.9	50.9	67.3	68.7	65.6	70.6	80.5	82.7	80.6
Mg	109.0	65.4	68.7	49.0	70.9	72.3	69.8	74.1	84.8	87.8	86.8
Na	91	107	140	110	114	107	107	107	102	103	100
K	15.4	16.2	20.9	16.4	17.3	16.3	16.3	16.2	16.2	15.6	15.4
F	1.8	4.4	4.6	4.8	4.8	4.3	4.3	4.2	3.5	3.3	3.3
Cl	31.7	31.9	36.2	35.5	35.7	34.6	32.4	31.4	29.1	28.9	29.1
Br	0.12	0.16	0.16	0.18	0.17	0.16	0.18	0.17	0.15	0.14	0.14
NO3-N	0.18	0.19	0.43	0.47	0.09	1.47	0.11	0.16	0.27	0.35	0.86
SO4	284	207	263	169	237	217	206	220	251	243	234
HCO3-*	530	415	427	342	439	462	450	500	490	524	515
<b>trace (ppb)</b>											
B	383	468	515	457	455	338	453	483	526	527	422
Co	5.1	1.9	3.7	1.6	4.0	2.4	2.0	1.6	2.3	2.0	2.2
Ni	1.9	1.5	0.9	1.7	1.3	1.2	1.7	1.5	3.3	2.7	2.0
Cu	1.2	1.1	0.0	0.0	0.8	0.8	0.9	0.6	1.5	0.5	0.6
Zn	7.8	28.9	30.2	23.7	5.0	17.7	18.1	12.5	15.0	13.9	17.7
As	3.7	2.2	4.8	5.8	4.9	3.7	5.9	4.7	6.1	5.8	5.5
Mo	15	129	123	106	102	86	95	84	67	71	74
W	0.11	0.18	0.75	0.15	0.13	0.12	0.22	0.23	0.81	3.81	1.65
Pb	0.03	0.08	0.48	0.08	0.10	0.00	0.07	0.00	0.09	0.07	0.06
<b>Nutrients (ppm)</b>											
N		<.20	0.22	<0.3	0.28	0.29	0.33	0.46			
NH3-N		0.12	0.32	0.24	0.34	0.27	0.31	0.23			
TP							0.01	0.03			
Notes	*reported as ppm CaCO3										

## APPENDIX 3

### Physical Isolation of Reactive Mine Wastes

Table A3.1.	Bibliographic annotations of research related to dry cover systems for reactive mine wastes . . . . .	A3.3
Table A3.2.	Summary of Operational-Scale Engineered Dry Cover Systems . . . . .	A3.7
	UNITED STATES . . . . .	A3.7
	CANADA . . . . .	A3.9
	AUSTRALIA . . . . .	A3.15
	SWEDEN . . . . .	A3.16
	NORWAY . . . . .	A3.17
Table A3.3.	Summary of Natural Hardpan Formations on Mine Waste Materials . . .	A3.18
Table A3.4.	Bibliographic annotations of research related to wet cover systems for reactive mine wastes . . . . .	A3.20
Table A3.5.	Summary of Operational-Scale Underwater Cover Systems . . . . .	A3.22
	UNITED STATES . . . . .	A3.22
	CANADA . . . . .	A3.23
	SWEDEN . . . . .	A3.29
	NORWAY . . . . .	A3.30



Table A3.1. Bibliographic annotations of research related to dry cover systems for reactive mine wastes. Italicized numbers refer to reference numbers for the Reclamation Section's literature database (Papyrus).

Reference	Study Scale <sup>1</sup>	Barrier Type		Barrier Material					
		Simple	Multi-layer	Soil	Clay	Mine Waste	Organic	Hard pan	Synthetic
Mannheim (1970).#2663	R	U		U					
Shackelford (1991) #2665	L	U		U					
Wilson et al. (1991)	L	U		U					
O'Kane (1993) #1412	M	U		U					
Swanson et al. (1995)	M	U		U					
Williams et al. (1997)	O	U		U					
McLendon et al., (1997)	O	U		U	U				
Rowe et al. #2664	L	U			U				
Bowders et al. (1994)	L	U			U				
Aziz & Ferguson (1997)	F	U			U				
Lindvall et al. (1997)	O	U			U				
Lundgren (1997)	O	U			U				
Kowalewski et al. (1998)	M	U			U				
Woyshner & Swarbrick (1997)	F	U			U	U			
Muller et al. (1998)	M, O	U				U			
Stogran & Wiseman (1995)	M, L	U				U	U		
Elliott et al. (1997) Tailing & Mine Waste	L, F	U					U		
Elliott et al. (1997) ICARD	L, F	U					U		
Chtaini' et al. (1997)	L, F	U					U		
Tasse et al. (1997)	O	U					U		

Reference	Study Scale <sup>1</sup>	Barrier Type		Barrier Material					
		Simple	Multi-layer	Soil	Clay	Mine Waste	Organic	Hard pan	Synthetic
Tasse (1999)	O	U					U		
Tasse (2000)	O	U					U		
Germain et al. (2000)	O	U					U		
Boorman & Watson (1976) #969	O	U							U
Stewart & vonMaubeuge (1997)	R	U							U
Miller & Hornaday (1998)	R	U							U
Brennecke & Corser (1998)	O	U							U
Neukirchner & Lord (1998)	O	U							U
Owaidat & Day (1998)	O	U							U
Shuri et al. (1998)	O	U							U
Sevick et al. (1998)	O	U							U
Lewis et al. (2000)	O	U							U
Udoh (1993) #2866	M, F	U		U	U				U
Rasmuson & Eriksson (1987) #912	R		U	U					
Rasmuson & Collin #1590	M		U	U					
Nicholson et al. (1989) #1131	M		U	U					
Barbour (1990) Can. Geotech. J. 27, 398-401	R		U	U					
Collin & Rasmuson (1990) #1504	M		U	U					
Akindunni et al. (1991)	M		U	U					
Nicholson et al. (1991) #1395	M, L		U	U					
Yanful (1991) #1396	M, L		U	U					

Reference	Study Scale <sup>1</sup>	Barrier Type		Barrier Material					
		Simple	Multi-layer	Soil	Clay	Mine Waste	Organic	Hard pan	Synthetic
Yanful et al. (1993) 578	M, O		U	U					
Yanful et al. (1993) 588	O		U	U					
Bell et al. (1994)	O		U	U					
O’Kane et al. (1995)	L, F		U	U					
Aubertin et al. (1997)	M		U	U					
Wilson et al. (1997),	M, O		U	U					
Woysner et al. (1997)	O		U	U					
Bussiere et al. (2000)	M		U	U					
Yanful & St-Arnaud (1991) #1397	F		U		U				
Yanful (1993)	L, F		U		U				
Yanful et al. (1994)	M, O		U		U				
Strachan & Raabe (1998)	O		U		U				
Duex (2000)	O		U		U				
Timms & Bennett (2000)	O		U		U				
Aachib et al. (1994)	L		U			U			
Bussiere et al. (1995)	L		U			U			
Puro et al. (1995)	O		U			U			
Bussiere et al. (1997)	L		U			U			
Hanton-Fong et al. (1997)	O		U			U			
Ricard et al. (1997)	M, O		U			U			
Benzaazoua et al. (1998)	L		U			U			
O’Kane et al. (1998) #2863	M,L,F		U			U			
Zahn et al. (2001)	M, F		U			U			
Elliott et al. (1997)	L		U			U	U		

Reference	Study Scale <sup>1</sup>	Barrier Type		Barrier Material					
		Simple	Multi-layer	Soil	Clay	Mine Waste	Organic	Hard pan	Synthetic
Cabral et al. (1997) Tailings & Mine Waste	F		U				U		
Cabral et al. (1997) ICARD	F		U				U		
Cabral et al. (1998)	L, O		U				U		
Cabral et al (1999)	O		U				U		
Hanselman & Courtin (1999)	L		U				U		
Blowes et al. (1991) #3860	O		U					U	
Ahmed (1994)	L		U					U	
Ahmed (1995)	L		U					U	
Chermak & Runnells (1997) #2745	L		U					U	
McGregor et al. (1997)	F		U					U	
Lin & Herbert (1997)	O		U					U	
Tasse et al. (1997)	O		U					U	
Ettner & Brasstad (1999)	O		U					U	
Shay & Cellan (2000) #3682	O		U					U	
Aubertin et al. (1997)	F		U	U		U			
Gardiner et al. (1997)	M, O		U	U		U			
Delaney et al. (1997)	R	U	U	U	U		U		U
Swanson et al. (1997)	O	U	U	U	U	U			

<sup>1</sup> R= review, M= model, L= laboratory, F= field, O= operation

Table A3.2. Summary of Operational-Scale Engineered Dry Cover Systems

UNITED STATES

**Dunka Mine, Babbitt, Minnesota**

Waste Material:	50 million tons of waste rock & gabbro (in 8014 and 8018) covering 320 acres
Cover Systems:	SMR (min. 2' till & vegetation): stockpiles with no known water quality problems EMR (min. 2' uncompacted -6" material w/ min. 10% at -200 mesh & vegetation): stockpiles 8011, 8012, 8031 FML (6" sand, 30 mil LDPE, 12" sand & vegetation): stockpile 8018 EMR-FML: stockpile 8013, 8014
Dates:	pile 8011: deposition from 1965-1985, capped 1995 pile 8013: deposition from 1967-1991, capped 1991 pile 8014: deposition from 1967-1993, capped 1995 pile 8018: deposition from 1979-1985, capped 1995 pile 8031: deposition from 1979-1985, capped 1995
Parameters Monitored:	flow, Cu, Ni, Co, Zn
Water Infiltration:	
Oxygen Flux:	
Effluent Quality (mg/L):	
Cost Estimates:	
References:	MN DNR, various published and unpublished reports

**Ridgeway Mine, South Carolina:**

Waste Material:	50 million ton, 300-acre gold tailings impoundment with 62 million ton capacity
Cover System:	preferred plan = 36" saprolite and 7" topsoil and vegetation
Dates:	began Mining in 1987, cover system modeling completed in 1998
Water Infiltration:	
Oxygen Flux:	
Effluent Quality (mg/L):	
Problems Identified:	
References:	Kowalewski et al. (1998) Tailings & Mine Waste pg 369

**Homestake's Santa Fe Mine, west central Nevada:**

Waste Material:	waste rock dump
Cover System:	cap purpose to neutralize existing near surface acidity and block diffusion of acid solutes. 20 t/ac magnesite and brucite applied uniformly over the surface, covered w/ 8-12" topsoil
Dates:	began leaching ore in 1988 and completed ore processing in 1995
Water Infiltration:	
Oxygen Flux:	

Results:	Monitoring data from the site has indicated that the chemical cap has been successful in preventing the diffusion of acidity into cover soil materials and the long-term goal of revegetation success is likely.
Cost Estimates:	cost of grading, purchase, hauling, application, revegetation, & engineering = \$776,000
References:	Shay and Cellan (2000) Proc. 2000 Billings Land Reclamation Symp. CD-ROM

**Richmond Hill Mine, Lawrence Co, South Dakota:**

Waste Material:	waste rock
Cover System:	The capping system= 6" of limestone, 18" clay, 4.5' of thermal barrier/drain layer, and 6" topsoil.
Dates:	Operation 1988-1993, ARD identified 1992, pit backfill 1994, cap completed 1995.
Parameters Monitored:	An extensive program is in place to monitor the effectiveness of the reclamation plan, and includes measurements on moisture content movement, temperature, oxygen content, and groundwater impacts.
Water Infiltration:	water received by the south sump decreased from 1.54 gpd in 1995 to 0.43 pgd in 1998
Oxygen Flux:	generally less than 0.2%, a few readings as high as 4.5%
Effluent Quality (mg/L):	
Cost Estimates:	
References:	Duex, T.A. ICARD, 2, 807-811 (2000)

## CANADA

### **Equity Silver Mine, near Smithers, British Columbia:**

Waste Material:	Southern Tail waste dump= approx 17 million tons gabbro waste rock from the Main Zone Pit was backfilled into the Southern Tail Pit
Cover System:	initially covered w/ 1 m uncompacted till, new cover design= 0.5 m of compacted clay/till overlain by 0.3 m of loose till and then revegetated
Dates:	mine operation 1980 - 1994, large field-scale test of a compacted clay cover evaluated in 1990, compacted clay cover constructed between 1990 and 1994 for the Southern Tail, Main, and Bessemer dumps.
Water Infiltration:	average infiltration has remained below 5% (0.2 - 12%)
Oxygen Flux:	Oxygen concentrations below the cover have decreased to a few percent.
Effluent Quality (mg/L):	no direct data provided, lime consumption at the treatment plant peaked in 1990 and decreased through 1996
Cost Estimates:	Estimated average cost at \$35,000/hectare.
References:	Aziz and Ferguson (1997) ICARD, 1, 183 Wilson et al. (1997) ICARD, 1, 197 O'Kane et al. (1995) Mining & the Environment, 565

### **Cominco Sullivan Mine, Kimberly, British Columbia:**

Waste Material:	90 mt of reactive tailing containing pyrrhotite and pyrite were placed in ponds that cover 373 hectares
Cover System:	Field test: Plot 2= 20-60 cm float rock, 25 cm compacted till and 30 cm non-compacted till; plot 3= 20-60 cm float rock and 45 cm non compacted till
Dates:	close in 2001 after 90 yrs of continuous operation, tailings disposal began 1923, field cover investigation began 1993
Water Infiltration:	non-compacted = 28%, compacted = 6%
Oxygen Flux:	non-compacted = 14-20%, compacted = 2-3% wet season & 12-15% dry season
Effluent Quality:	no direct data, treatment plant lime consumption for non-compacted=482 t, compacted=270 t
Cost Estimates:	\$15 million (\$4.00/m <sup>2</sup> )
References:	Gardiner et al. ICARD, 1, 47 (1997)

### **Kidd Creek tailings, near Timmins, Ontario:**

Waste Material:	thickened tailings impoundment
Cover System:	Test plot 1: 60 cm nonreactive tailings Test plot 2: 15 cm slag under 45 cm nonreactive tailings Test plot 3: 60 cm native clay
Dates:	installed in 1995, monitoring results for 1995 & 1996
Water Infiltration:	tailings beneath the covers retained saturation, 0.85-0.89

Oxygen Flux:

Cover	Annual O <sub>2</sub> Flux (mol/m <sup>2</sup> /yr)	SO <sub>4</sub> Production (kg/m <sup>2</sup> /yr)
NR tailings	36.1	2
slag & NR tailings	73.7	4
clay	13.1	1
none after 1 yr (prediction)	729	40
none after 20 yrs (prediction)	91.1	5

Effluent Quality (mg/L):

Cost Estimates:

References: Woyshner and Swarbrick, ICARD, 3, 1075 (1997)

### East Mine, Falconbridge, Ontario:

Waste Material: tailings impoundment

Cover System: 75 cm over 100 m<sup>2</sup> was excavated and refilled with 26-30 cm reactive waste, the self-sealing/self-healing barrier, 17-23 cm high Fe-S (mill washings) material and filled to the top with the original oxidized tailings that had been excavated.

Dates: tailings deposited prior to 1985, cover installation took place in 1995

Water Infiltration: Testing of cores taken through the barrier determined an average vertical K of  $3.6 \times 10^{-8}$  cm/s for the barrier compared to an average K of  $3.9 \times 10^{-4}$  cm/s for the surrounding tailings.

Oxygen Flux: Pore gas oxygen measurements within the vadose zone of the tailings revealed a marked decrease in O<sub>2</sub> concentration below the barrier (<3.4%) when compared to the O<sub>2</sub> profile of the shallow tailings beside the barrier test plot.

Other Pore Gases: Pore gas measurements also determined an accumulation of pore gas CO<sub>2</sub>, of up to 17% of the pore gas, below the barrier.

Cost Estimates: Estimated cost: \$30-50k/ha.

References: McGregor et al. (1997) ICARD, 3, 1435

### East Sullivan Mine, near Val d'Or, Quebec:

Waste Material: 136 ha tailings impoundment

Cover System: forestry wastes produced by sawmill and presswood industries,

Plot	Organic Cover	Tailings
0	none	weathered
1	none	fresh
2	1 m bark	weathered
3	1 m bark	fresh
4	2 m bark	weathered
5	1 m bark and 30 cm sewage sludge	weathered

Dates: tailings accumulation 1949-1966, forestry wastes laid down between 1984-1996 cover almost 75% of the 136 ha tailings impoundment

Oxygen Flux:  $O_2 < 3\text{-}5\%$

Pore Gases:  $CO_2$  20-30% and  $CH_4$  10--40%

Effluent Quality (mg/L):  $pH > 5.5\text{-}6$ ,  $> 2000$  ppm  $CaCO_3$

Cost Estimates:

References: Tasse et al., ICARD, 4, 1627 (1997)

### **Norebec-Manitou site, near Val d'Or, Quebec:**

Waste Material: sulfidic tailings from the Manitou-Barvue site

Cover System:

Field Test Plot	Reactive Tailings (m)	Sand (m)	Fine Layer (m)	Sand (m)
1	1.6	0.4	0.60 tailings	0.3
2	1.58	0.4	0.60 silt	0.3
3	1.52	0.4	0.30 tailings	0.3
4	1.21	0.4	0.15 tailings & 0.15 tails/bentonite	0.3
5	1.55	0.4	0.90 tailings	0.3
6	1.5	---	---	---

Water Infiltration: The coarse layers maintained saturation ratios of 25-40%, while the fine layers were about 85%.

Oxygen Flux: An effective oxygen barrier is considered to be 80-90% saturated, depending on thickness of the layer.

Cost Estimates: Total construction costs for six cells 70k\$US, and instruments totaled about 45k\$US.

References: Aubertin et al. (1997) ICARD, 2, 715

### **Millenbach Site near Rouyn-Noranda, Quebec:**

Waste Material: moved 17500 m<sup>3</sup> of waste rock and fine material from various locations on site onto 0.66 ha of tailings

Cover System: composite clay cover (30 cm coarse sand, 50 cm compacted clay, 30 cm fine sand, 10-15 cm top soil).

Dates: operation 1971-1980, decommissioned during 1990 and 91

Water Infiltration: detailed calculations given

Oxygen Flux: 0.56 mol/m<sup>2</sup>/yr

Effluent Quality (mg/L): oxidation products = 184 kgO<sub>2</sub>/yr, 89 kgFe<sup>++</sup>/yr, 306 kgSO<sub>4</sub>/yr, 319 kg-acidity as CaCO<sub>3</sub>/yr

Cost Estimates:

References: Woyshner et al., ICARD, 4, 1673 (1997)

**Waite Amulet site, near Rouyn-Noranda, Quebec:**

Waste Material: tailings  
 Cover System: 20 x 20 m test plots (max slope = 3:1): plot 1= control; plot 2 & 3= 30 cm sand, 60 cm compacted, varved clay, 30 cm sand, 10 cm gravel; plot 4= 30 cm sand, 80 mil HDPE, 30 cm sand, 10 cm gravel  
 Dates: project initiated 1990,  
 Water Infiltration: Infiltration through the soil covers = 3.9% of the precipitation during the study.  
 Oxygen: 2.0% beneath the soil covers, 6% beneath HDPE  
 Effluent Quality:

Oxidation Product (g/L)	Uncovered tailings	Covered tailings
Fe	5-20	1-1.5
SO <sub>4</sub>	16-63	5-5.7

Cost Estimates:

References: Yanful et al. (1994) Pitt94, 2, 138

**Les Terains Auriferes Site, Malartic, Quebec:**

Waste Material: Goldfield carbonate tailings: about 10 Mt placed in a 5 m deep layer in a 100 ha tailings pond; Bousquet sulfide tailings: about 8 Mt placed on top of Goldfields tailings  
 Cover System: Zone 1: 0.5 m sand and gravel  
 Zone 2: 0.5m sand and gravel, 0.8 m compacted tailings, 0.3 m sand  
 Dates: closed 1994, construction began 1995  
 Parameters Monitored:  
 Water Infiltration: 6 months of monitoring show an average of 86% saturation on top of the stack and 84% on the slopes.  
 Oxygen Flux: Oxygen consumption measurements show reduced oxygen flux by an average factor of 75, up to 1000.  
 Effluent Quality (mg/L):  
 Cost Estimates: construction = \$65,000/ha, total = \$93,500/ha  
 References: Ricard et al. (1997) ICARD, 4, 1515; Tremblay and Bussiere (1999) MEND 2000 Workshop ME.02

**Clinton site, near Woburn, Quebec:**

Waste Material: waste rock backfill in a 2000 m<sup>2</sup> x 20 m mine shaft  
 Cover System: 5000 m<sup>2</sup> capillary barrier= 30 cm coarse waste rock, 120 cm deinking residues, 50 cm granular material, 10 cm vegetative cover  
 Dates: project began 1997  
 Water Infiltration: k = 1-4 x 10<sup>-6</sup> cm/s  
 Pore Gases: O<sub>2</sub> <2%, CO<sub>2</sub> - 38%, CH<sub>4</sub> - 38%

Effluent Quality (mg/L): pH=7.2, Cu<0.005, Zn=1  
Cost Estimates:  
References: Cabral et al. (1999) Tailings & Mine Waste, 405

**Eustis Mine site, near Sherbrooke, Quebec:**

Waste Material: abandoned tailings, 1.6 ha test plot  
Cover System: deinking residue mixed w/ a “superficial crust” of oxidized tailings, fresh compacted residues, 20 cm compost-residue  
Dates: mine active from 1865-1939, project began Aug. 1995, cell construction completed spring 1996,  
Water Infiltration: degrees of saturation in the range of 76%-85%  
Oxygen: diffusion coefficients in the range of 3.5 m<sup>2</sup>/yr to 110 m<sup>2</sup>/yr  
Pore Gases: Oxygen, carbon dioxide, and methane profiles showed that the oxygen concentration 20 cm below the surface was virtually nil and that the concentrations of carbon dioxide and methane increased with time.

Effluent Quality (mg/L):  
Cost Estimates:  
References: Cabral et al. (1998) Tailings & Mine Waste, 379  
Cabral et al. (1997) Tailings & Mine Waste, 257  
Cabral et al. (1997) ICARD, 3, 1109

**Poirier Site, northern Quebec:**

Waste Material: 5 million tonnes acid generating tailings  
Cover System: geomembrane clay cover = 0.5 m compacted clay on top of a geomembrane liner, overlain by 0.5 - 1.5 m till for vegetation  
Dates: underground Cu/Zn mine operated 1966-1975, property sold in 1985, closure planning began in 1996, implemented in 1998, scheduled for completion in 2000  
Water Infiltration:  
Oxygen:  
Pore Gases:  
Effluent Quality (mg/L):  
Cost Estimates:  
References: Lewis et al. (2000) ICARD, V2, 959; Gallinger (1999) MEND 2000 Workshop ME.02

**Heath Steele Pile 7/12, near Newcastle, New Brunswick:**

Waste Material: 14000 t of mine waste rock and has been producing acidic seepage (pH = 2.1-2.8, Fe = 3.5-13.5 g/L, SO<sub>4</sub> = 12.7-43.4 g/L)  
Cover System: Pile 7/12 was relocated covering 0.25 ha in plan, and has a maximum depth of 5 m (maximum slope of 3:1). The waste rock pile was underlain by a sand layer and an impermeable membrane. Composite cover design= 130

cm thick soil cover (30 cm sand base, 60 cm compacted till, 30 cm granular, 10 cm gravel for erosion protection-58% sand, 40% gravel).  
 Dates: waste relocated 1989, closed 1994  
 Water Infiltration: There has been no noticeable change in the moisture content of the glacial till layer.  
 Oxygen Flux: Reported oxygen concentrations dropped from an initial range of 7.3 - 20.8 % to 8.2 - 14.5% immediately after the cover was in place to 0.2 - 0.7 % after five years.  
 Effluent Quality (mg/L):

(g/L)	1989-90	1992	1993
pH	2.1-2.8	2.3-2.9	3.0-3.2
acidity (as CaCO <sub>3</sub> )	15.8 - 76	15.8 - 54	not available
SO <sub>4</sub>	12.7 - 43	5.1 - 71	10 - 74
Fe	3.5 - 13.8	15.8 - 54	5.0 - 31

Cost Estimates:

References: Yanful et al. Can. Geotech. J., 30, (1993) 578-87.  
 Yanful et al. Can. Geotech. J., 30, (1993) 588-99  
 Bell et al. Pitt94, 2, 113.

## AUSTRALIA

### **Rum Jungle, Northern Territory:**

Waste Material: Whites uranium and copper waste rock dump  
Cover System: 3 layers = 0.225-0.30 m compacted clay, 0.25-0.30 m sandy clay loam, 0.15 m gravely sand  
Dates: mined during 1950's til early 70's, cover was put in place in 1983  
Parameters Monitored:  
Water Infiltration: approx. 2% of precipitation til 1994, increasing to 10% in 1999  
Oxidation Rate: decreased from 0.0466 to 0.137 kg/s  
Effluent Quality (mg/L): SO<sub>4</sub> generation rate decreased from 2520 to 740 t/yr  
Cost Estimates:  
References: Timms and Bennett (2000) ICARD, 2, 813-818.

### **Mt Whaleback:**

Waste Material: 2 billion tons of waste rock over 30 years  
Cover System: The cover system is designed to maximize infiltration during wet periods for subsequent evapotranspiration while minimizing surface runoff.  
Dates: BHP Iron Ore initiated research programs in January of 1995 to develop long term plans for decommissioning of the waste rock material at their Mt. Whaleback operation.  
Results: Monitoring is in progress and will continue for at least two annual wet/dry cycles.  
Water Infiltration:  
Oxygen Flux:  
Effluent Quality (mg/L):  
Cost Estimates:  
References: O'Kane et al. (1998) SME Preprint 98-70

## SWEDEN

### **Bersbo Pilot Project:**

Waste Material:	about 700,000 m <sup>3</sup> of waste rock from an abandoned, historic copper mine
Cover System:	The remediation works included concentration of the waste rock to two piles reducing the exposed area to 50% and dumping 1/3 of the waste volume under the groundwater table in old mine shafts. A sealing layer of compacted clay (0.5 m ) installed on one of the piles and cement stabilized fly ash (0.25 m) on the other pile, both covered by a protective cover of 2 m glacial till.
Dates:	remediation project 1987-1989, follow-up & monitoring since 1989
Water Infiltration:	The water percolation has been significantly reduced to less than 3% (clay liner) and 12% (cementitious liner) of what it was prior to covering.
Oxygen Flux:	The oxygen transport is reduced substantially through the clay liner. The cementitious liner, however, is a less good oxygen barrier, partly due to the higher permeability, partly due to the not fully saturated sealing layer.
Cost Estimates:	
References:	Lundgren (1997) ICARD, 3, 1419

### **Saxberget Mine, near Ludvika:**

Waste Material:	sulfidic tailings producing Zn
Cover System:	0.3 m sealing layer of compacted clayey till applied in two lifts, 1.5 m protection layer of unsorted till, and vegetated
Dates:	began in summer of 1993, completed in autumn of 1995
Water Infiltration:	permeability = $5 \times 10^{-9}$ m/s,
Oxygen Flux:	O <sub>2</sub> <0.5%
Effluent Quality (mg/L):	included in follow up studies
Cost Estimates:	\$14 CND/m <sup>2</sup>
References:	Lindvall et al. (1997) ICARD, 3, 1389-1400

## NORWAY

### **Storwartz Mine, near Roros:**

Waste Material:	tailings impoundment
Cover System:	Four 4m <sup>2</sup> induced hardpan test plots: cell 1=3cm fine and coarse limestone covered by 2 cm oxidized tailings; cell 2= 3cm fine and coarse limestone covered by 2 cm unoxidized tailings; cell 3= control; cell 4= 3 cm quicklime & crushed limestone covered by 2 cm unoxidized tailings
Dates:	issue related remediation & preserving historic Mining areas
Hardpan Description:	am-FeOOH clumps & coatings, jarosite, gypsum
Water Infiltration:	k- $9 \times 10^{-6}$ m/s compared to $1 \times 10^{-5}$ m/s in control
Oxygen Flux:	top 20 cm of tailings=8-18%, decreased to <1% by 40-50 cm depth
Cost Estimates:	
References:	Ettner and Brasstad (1999) Tailings & Mine Waste, 457

Table A3.3. Summary of Natural Hardpan Formations on Mine Waste Materials

**Unknown Location:**

Waste Material:	inactive sulfide-containing mill-tailings deposit
Hardpan Description:	The Fe precipitates consist mainly of goethite, and lesser amounts of lepidocrocite and other Fe hydroxides, and traces of Ca, Cu, Fe, and Zn sulfates.
General Results:	The Fe precipitates fill the intergrain pores, and cement the tailings matrices, leading to a reduction of the tailings porosity. In addition, dissolved elements, e.g. As, Cu, Zn, are retained by Fe oxyhydroxides through adsorption and co-precipitation. As a result, the Fe precipitated layers behave as a barrier to restrict the movement of oxygen, and act as an accumulation zone of heavy metals and As. The extent of sulfide oxidation, and the concentration of dissolved metals, is greatly reduced below the Fe precipitated layers.
Effluent Quality:	below the precipitated layers: pH = 6.2 - 6.5; concentration of dissolved metals = a few ppm
Pore Gases:	Carbon dioxide levels below the cemented layers were recorded at 60% compared to 15% above the layers. The oxygen diffusion coefficient was $1.7 \times 10^{-8} \text{ m}^2/\text{s}$ compared to $2.0 \times 10^{-6} \text{ m}^2/\text{s}$ in uncemented tailings.
References:	Lin and Herbert (1997) Tailings & Mine Waste, 237

**Canadian Malartic Mine, Quebec:**

Waste Material:	The Canadian Malartic impoundment features a layer cake structure with 6.9 Mt of mildly reactive tailings at the base, generated by gold Mining, and 1.2 Mt of strongly reactive, sulfurous, tailings at the top, produced from Ni-Cu extraction.
Dates:	first deposition cycle (Malartic) 1942-1965, second (Marbridge) in the 1960's
Hardpan Description:	Near surface reactive sulfides have already reacted, and oxygen access to fresh sulfides is limited by the presence of a fairly continuous hardpan layer. The surface impoundment was leveled and revegetated with minimal disturbance, keeping the natural drainage pattern developed over the site, and avoiding soil disruptions at depths larger than a few tens of centimeters. Where slope reshaping involved hardpan disturbance, reactive tailings were carried and disposed in a safer location on the site.
Results:	Heavy metals show little mobility within and around the impoundment periphery, as a result of precipitation, coprecipitation, or adsorption. Runoff in direct contact with altered tailings devoid of any significant neutralizing capacities, generates the most contaminated waters, but there are no significant accumulations of such waters.
References:	Tasse et al. (1997) ICARD, 4, 1797.

**Waite Amulet, Quebec:**

Waste Material: Cu-Zn massive sulfide tailings deposited in a 41 ha elevated impoundment (2-15 m deep), 30 wt% sulfide minerals (pyrite, pyrrhotite, sphalerite, and chalcopyrite; traces of galena)

Dates: mill operations from 1929 to 1964

Hardpan Description: 1-5 cm thick, laterally discontinuous (10-100 cm), occurs at the depth of active oxidation, and is characterized by cementation of tailings by Fe(III) minerals

Results:

References: Blowes et al. (1991) *Geochim. Cosmochim. Acta*, 55, 965-978.

**Heath Steele, New Brunswick:**

Waste Material & Dates: Two tailings impoundments, an older one 40 ha where tailings were deposited between 1957 and 1963, and a newer one 200 ha where tailings deposition began in 1965.

Hardpan Description: 10-15 cm thick, occurs 20-30 cm below the depth of active oxidation, is continuous throughout the tailings impoundment, and is characterized by cementation of tailings by gypsum and Fe(II) solid phases, principally melanterite.

Results: Accumulation of gas-phase CO<sub>2</sub>, of up to 60% of the pore gas, occurs below the hardpan. The calculated diffusivity of the hardpan layer is only about 1/100 that of the overlying uncemented tailings. The pore-water chemistry has changed little over a 10-yr period, suggesting that the cemented layer restricts movement of dissolved metals and acts as a zone of metal accumulation.

References: Blowes et al. (1991) *Geochim. Cosmochim. Acta*, 55, 965-978.

Table A3.4. Bibliographic annotations of research related to wet cover systems for reactive mine wastes. Italicized numbers refer to reference numbers for the Reclamation Section's literature database (Papyrus).

Reference	Study Scale <sup>1</sup>	Disposal Setting					
		Tailing s Basin	Constructed Wetland	Open Pit	Under- ground Mine	Natural Lake	Marine
Aubertin et al. (1997)	R						
Dave et al. (1997) #3450	L						
Stogran et al. (1997)	L						
Paktunc & Dave (1999)	L						
Nicholson et al. (1999)	L						
Nicholson & Rinker (2000)	M, L						
Amyot & Vezina (1997)	O	U					
Atkins et al. (1997)	M, F	U					
Kam et al. (1997)	O	U					
Li et al. (1997)	M, O	U					
Ljungberg et al. (1997)	F	U					
Robertson et al. (1997)	F	U					
St-Germain et al. (1997)	F	U					
Beckett et al. (1999)	O	U					
Evans (1999)	L, F	U					
Lindvall et al. (1999)	O	U					
Zou & Huang (1999)	L	U					
Catalan et al. (2000)	O	U					
DeVos et al. (2000)	M, O	U					
Li et al. (2000)	M, F, O	U					
Jones et al. (1997)	M, O		U				
Eger et al. (2000)	F		U				
Atkins et al. (1997)	M, F			U			
Bursey et al. (1997)	M, F			U			

Reference	Study Scale <sup>1</sup>	Disposal Setting					
		Tailing s Basin	Constructed Wetland	Open Pit	Under- ground Mine	Natural Lake	Marine
Hamblin et al. (1997)	M, F			U			
Meyer et al. (1997)	M			U			
Warren et al. (1997)	M			U			
Dorey et al. (1999)	O			U			
Day et al. (2000)	O			U			
Crusius, et al. (2001)	M			U			
Arnesen & Iversen (1997)	O				U		
Neukirchner & Hinrichs (1997)	O				U		
Hockley et al. (1998)	M, O				U		
Quarshie & Rowson (1995)	L					U	
Pedersen et al. (1997)	O					U	
Pedersen et al. (1999)	O					U	
Morin & Hutt (2000)	O					U	
Mugo et al. (1997)	L					U	U
Mugo et al. (1999)	M, L					U	U
Ganguli et al. (2001)	R						U
Orava et al. (1997)	R	U		U			
Arnesen et al. (1997)	M, O	U				U	
Robertson et al. (1997)	O	U				U	

<sup>1</sup> R= review, M= model, L= laboratory, F= field, O= operation

Table A3.5. Summary of Operational-Scale Underwater Cover Systems

## UNITED STATES

### **Eagle Mine, Minturn, Colorado**

Disposal Setting:	inactive Zn-Pb ore body
Waste Material:	pyrite, sphalerite & galena
Dates:	operations discontinued in 1977, flooding began in 1983, water quality monitoring began in 1989
Parameters Monitored:	pH, Zn, Mn, SO <sub>4</sub> inflow
Results:	pH increased from 3 to 6, Zn decreased from 350 mg/L to 20-50 mg/L, Mn decreased from 314-380 mg/L to 40-54 mg/L, SO <sub>4</sub> decreased from >11,000 mg/L to 2600-2970 mg/L, inflow has decreased by about 1/3
References:	Neukirchner & Hinrichs (1997) ICARD, V4, 1469.

### **Kennecott Ridgeway Gold Mine, South Carolina**

Disposal Setting:	flooding backfilled open pits to create deep water lakes, pits will be backfilled at a pace that allows for rapid innundation (<4 months)
Waste Material:	backfilled waste rock and alkaline (lime) amended water
Dates:	reclamation of the South Pit began around 1998, the North Pit was still in production
Parameters Monitored:	
Results:	Water quality in the South Pit meets drinking water standards with minor occurrences of Zn, Cu, & Fe at ppb levels. Algae discovered in the pit water in 1998 suggests continued development of a viable fresh water lake environment.
References:	Dorey et al. (1999) Tailings & Mine Waste, 701

## CANADA

### **Island Copper Mine, Vancouver Island, British Columbia**

Disposal Setting: flood the pit with seawater from Rupert Inlet  
Waste Material: acidic drainage from waste rock injected at bottom of pit  
Dates: operation 1971-1996?  
Parameters Monitored:  
Results:  
References: MEND Report 2.36.1 (1995)

### **Eskay Creek Mine, British Columbia**

Disposal Setting: biologically-barren alpine lake, lime was added to each truckload  
Waste Material: waste rock  
Dates: dump construction began 1990, drainage acidified by 1992, waste rock relocated 1994-1995  
Parameters Monitored: pH, acidity, Zn, Pb,  
Results: in the watershed: 1 year pH = 4.7, Zn = 5 mg/L, Pb = 0.2 mg/L; 3 years: pH = 7.0, Zn = 0.2 mg/L, Pb = 0.009 mg/L (all qualified) in the lake: remained near background levels  
Costs: CND\$ 500,000 for relocation and submergence (about CDN \$5/t)  
References: Morin, K.A.; Hutt, N.M., ICARD 2000 pg 819-825.

### **Benson Lake, British Columbia**

Disposal Setting: vertical discharge from raft at 30 m depth  
Waste Material:  
Dates: in operation 1962-1973  
Parameters Monitored:  
Results:  
References: Dave (1992)

Location: Equity Silver, British Columbia  
Disposal Setting: man-made tailings pond  
Waste Material: tailings (Aspy, py, po), 2-10% S  
Dates: operation from 1980 to Jan. 1994  
Parameters Monitored: N, NO<sub>3</sub>, PO<sub>4</sub>, Cl, CN, Sb, As, Cd, Fe, Pb, Mn, Zn, C<sub>org</sub>, DO  
Results: Fe, Mn, Cu, Zn, Sb, As & CN in water cover are low, Cu decreasing, pH = 7-8, DO = 5 mg/L  
References: Yanful & Simms (1996); Pedersen et al. (1997) ICARD, V3, 989

### **Gunnar Pit, Lake Athabasca, Saskatchewan**

Disposal Setting: flooded open pit  
Waste Material: waste rock?  
Dates:  
Parameters Monitored:  
Results:  
References: MEND Report 2.36.1 (1995)

### **Cluff Lake “D” Pit, Saskatchewan**

Disposal Setting: flooded open pit  
Waste Material: waste rock  
Dates: monitored 1982-1994  
Parameters Monitored: pH, metals, anions,  
Results: 1989-1994: pH=7.36, SO<sub>4</sub>=8.1 mg/L, As=18.4 µg/L, Fe=0.892 mg/L, Ni=0.028 mg/L, U=66.75µg/L  
References: MEND Report 2.36.1 (1995); Quarshie & Rowson, Sud95, V1, pg 217

### **Rabbit Lake Mine, Wollaston Lake, Saskatchewan**

Disposal Setting: engineered pit disposal w/ bottom rock drain  
Waste Material: uranium tailings,  
Dates: operation 1975-1984, deposition 1985-?  
Parameters Monitored: pH, Al, As, Ca, Fe, K, Mg, Mn, Ni, Pb, SO<sub>4</sub>, Ra, U  
Results: 1994: pH = 8.4, SO<sub>4</sub> = 1886 mg/L  
Costs: estimated to be \$2.62/t tailings in 1995\$  
References: MEND Report 2.36.1 (1995)

### **Collins Bay B-Zone Pit, Wollaston Lake, Saskatchewan**

Disposal Setting: open pit  
Waste Material: waste w/ elevated As, Ni, S, or U  
Dates: decommissioned 1991,  
Parameters Monitored: T, DO, cond, pH, redox, trace metals, radionuclides, nutrients, major constituents  
Results: 1994: As=0.359 mg/L, Ni=0.303 mg/L, SO<sub>4</sub>=10.5 mg/L, Pb<sup>210</sup>=0.11 Bq/L, Ra<sup>226</sup>=0.06 Bq/L, U=0.014 mg/L  
References: MEND Report 2.36.1 (1995)

### **Anderson Lake, Manitoba**

Disposal Setting: discharge into lake at 3-6 m depth  
Waste Material: tailings (py, po, sphal, chalcoppy, gal, and arsenopy), 20-25% S  
Dates: disposal began 1979  
Parameters Monitored: T, O<sub>2</sub>, Cu, Pb, Zn, Ni, Co, Fe/Al, Mn, S<sub>T</sub>, C<sub>org</sub>, TKN, pH

Results: tailings dispersed up to 2 km from discharge, natural deposition of organic-containing sediment consumed O<sub>2</sub>, little post-deposition oxidation except in littoral zones, dissolved metal concentrations decreased abruptly in sediments (removal by flora/fauna?), pH decr from 8 to 6, metals and turbidity incr 5-8x, sulfate incr 19-20x,

Problems Identified: turbidity = suspension of fine particles and effects aquatic biota, burial (siltation) of bottom life has ecological effects, toxicity of soluble compounds on aquatic biota

References: Dave (1992); Pedersen et al. (1997) ICARD, V3, 989; Pedersen et al. (1999) Mining & Environment, v1, 165.

### **Mandy Lake, Manitoba**

Disposal Setting: disposal in shallow water along shore

Waste Material: tailings (py) 15-17% S

Dates: in operation 1917-1920 & 1943-1944

Parameters Monitored: pH, cond, SO<sub>4</sub>, Fe, Mn, Zn, Cu, Pb, Co, Ni, TKN, C<sub>org</sub>, O<sub>2</sub>, As,

Results: tailings dispersed up to 2 km from discharge, natural deposition of organic-containing sediment consumed O<sub>2</sub>, little post-deposition oxidation except in littoral zones, dissolved metal concentrations decreased abruptly in sediments (removal by flora/fauna?)

Problems Identified: turbidity = suspension of fine particles and effects aquatic biota, burial (siltation) of bottom life has ecological effects, toxicity of soluble compounds on aquatic biota

References: Dave (1992)

### **Quirke Lake Tailings Test Site, Elliot Lake, Ontario**

Disposal Setting: 192 ha tailings pond = series of terraced cells, water from a local lake, limestone addition

Waste Material: 45 million tonnes of uranium tailings & waste rock (py = 5.3%)

Dates: operated from 1956-1961 & 1968-1990; dike construction carried out from 1990 to 1995, flooded began in 1992

Parameters Monitored: pH, SO<sub>4</sub>, Fe, Zn

Results: pH variable from 4 to 9, sulfate decreasing slightly, Fe = 0.2 mg/L, Zn = 0.01 mg/L

References: Yanful & Simms (1996); Kam et al. (1997) ICARD, V2, 853; Gallinger (1999) MEND 2000 Workshop ME.02

### **Panel Wetlands, Elliot Lake, Ontario**

Disposal Setting: 123 ha wetland basin filled with tailings and minimum water level of 0.6m, lime addition

Waste Material: 14 million tonnes tailings

Dates: operated 1958-1961 & 1979-1990; tailings deposited in the late 1950s

Parameters Monitored: pH, SO<sub>4</sub>, Fe, bacteria,

Results: stable low levels of acidity, SO<sub>4</sub>, & Fe for 10 years

References: Yanful & Simms (1996)

### **Owl Creek, Timmins, Ontario**

Disposal Setting: flooded open pit  
Waste Material: waste rock  
Dates: operation 1981-1989, waste moved to pit in 1991  
Parameters Monitored: pH, T, DO, ICP metal scan, SO<sub>4</sub>, NO<sub>3</sub>  
Results: June 1994: pH neutral & water quality “continuing to recover”  
Costs: relocation and regrading of remaining waste approx. \$7,500,000  
References: MEND Report 2.36.1 (1995)

### **Falconbridge Limited, New Tailings Area, Falconbridge, Ontario**

Disposal Setting: tailings pond  
Waste Material: tailings  
Dates: These tailings were relocated underwater by truck and excavator in the fall of 1996, and by dredging in September and October of 1997. Tailings picked up by dredging in these areas were transported through a floating pipeline and re-deposited in areas immediately upstream of Dam 12. Prior to final phase of dredging in 1997, approximately 850 tonnes of lime were spread on the surface of the tailings.  
Parameters Monitored: pH, SO<sub>4</sub>, Ca, Cu, Fe, Ni, Zn  
Results: Peaks of metal concentrations have appeared each spring. The magnitude of these peaks, however, has declined from 6 mg/L Ni in 1996, to slightly more than 1 mg/L in 1999. The resulting increase in the Lower Terrace is correspondingly lower as well, and, since February 1999, nickel has been below provincial effluent quality limits.  
  
In conjunction with NTC, Lorax Environmental studied the sediment and pore water chemistry in 1998. The main conclusions from this work were: surficial sediments are organic-poor, and the organic matter is of aquatic source (plankton); maximum dissolved oxygen penetration into the sediments was only 3 mm; submerged tailings are anoxic at shallow depths, as indicated by the oxygen data, and supported by profiles of dissolved iron and manganese; and minor amounts of arsenic and molybdenum are diffusion from the tailings, although water column concentrations are quite low (1-2 mg/L)  
Cost: Total cost of construction is estimated at \$1.39 million. It is estimated that the cost of preliminary engineering, research and design totaled \$0.18 million. The estimated total cost of the project is \$2.61 million, or \$29,000 per hectare.  
References: MEND: <http://mend2000.nrcan.gc.ca/cases/falcon/falcon.htm>; DeVos et al. (2000) ICARD, V2, 933; Hall (1999) MEND 2000 Workshop ME.02

### **Strathcona sites, Sudbury, Ontario**

Disposal Setting: subaqueous deposition in a 90 ha arm of a larger lake basin

Waste Material: sand and silt sized tailings  
 Dates: deposition since 1968, > 20 m thick of sand and silt sized tailings  
 Parameters Monitored: grain size, permeability, SO<sub>4</sub>, water table depth  
 Results: Sulfate levels indicate that the bulk of the tailings pore water consists of mill process water unaffected by sulfide oxidation.  
 Cost:  
 References: Robertson et al. (1997) ICARD, V2, 621

### **Solbec Open Pit, Eastern Township, Quebec**

Disposal Setting: open pit  
 Waste Material: tailings in 1977, waste rock 1988  
 Dates: operation 1962-1970, mill operation til 1977, decommission 1977-1979,  
 Parameters Monitored: pH, Cu, Pb, Zn, Fe, Mn, Cd, T, cond, alk, DO, TDS  
 Results: 1992: pH=6.7, TDS=4.6 mg/L, Cu=0.07mg/L, Zn=2.6 mg/L, Fe=1.07 mg/L  
 Cost: 1988-1992 = \$1.069.100 or \$3.87/m<sup>3</sup>  
 References: MEND Report 2.36.1 (1995)

### **Solbec Open Pit, Eastern Township, Quebec**

Disposal Setting: flooded tailings pond  
 Waste Material: tailings in 1977, waste rock 1988  
 Dates: In 1992, a fourth phase was undertaken. To offset the leaching of the oxidation products from the oxidized tailings, limestone was added in the testing basins as an attempt to neutralize and stabilize the previously acidified water. In December 1993, flooding of the pond after addition of lime was recommended as the reclamation method.  
 Parameters Monitored: The 18 water cover samples are subjected to the following analysis without being filtered. In the field: pH, electrical conductivity and Fe+2. In the laboratory: Metals (Fe, As, Cu, Ni, Pb, Zn), sulfates, suspended matter, alkalinity and acidity if appropriate.  
 Results: (approximates) pH = 6-8.5, Results have demonstrated that the covering waters met all requirements outlined in Regulation 019 (Ministry of Environment and Wildlife) and even those of the drinkable water regulation.  
 Cost:  
 References: Amyot and Vezina (1997) ICARD, 2, 681; Amyot & Vezina (1999)  
 MEND: <http://mend2000.nrcan.gc.ca/cases/flooding-99/flooding-99.htm>

### **East Kemptville Tin Mine, Nova Scotia**

Disposal Setting: 250 ha tailings impoundment, submergence of potentially acid generating tailings  
 Waste Material: 18.8 million tonnes of pyritic tailings  
 Dates: operated from 1984 - 1992  
 Parameters Monitored: Zn, Al, F, Mn, Cu

Results:

Cost:

References:

Gallinger (1999) MEND 2000 Workshop ME.02

## SWEDEN

### **Stekenjokk, Sweden**

Disposal Setting: clarification pond  
Waste Material: tailings, 20% S  
Dates: operation 1976-1988, decommissioned 1991 w/ permanent flooding, geochemical study begun in 1995  
Parameters Monitored: Cu, Pb, Fe, Zn, SO<sub>4</sub>, DO  
Results (mg/L): Cu= 0.002, Cd=0.07, Zn = 0.1 mg/L, SO<sub>4</sub> = 14 mg/L, suspended solids < 5 mg/L  
References: Yanful & Simms (1996); Ljungberg et al. (1997) ICARD, V3, 1401

### **Udden Open Pit, Sweden**

Disposal Setting: flooded open pit  
Waste Material: weathered sulfidic waste rock  
Dates: relocated June 1994  
Parameters Monitored: pH, TDS, cond, SO<sub>4</sub>, Cu, Pb, Zn, Cd, flow  
Results: August 1994: pH=6.0, SO<sub>4</sub>=1200 mg/L, Cu=0.1-0.17 mg/L, Zn=81 mg/L, Cd=0.14 mg/L  
References: MEND Report 2.36.1 (1995)

### **Kristineberg Mine, Sweden**

Disposal Setting: Tailings area consists of five individual ponds. In order to finally incorporate the tailings area into the surrounding environment the Vormbacken Creek, which had previously been diverted around the tailings ponds will be rediverted into the lower ponds turning them into artificial lakes naturally interacting with the creek.  
Waste Material: tailings  
Dates: mill closed in 1991, project commissioned in 1995, began in 1996, scheduled to be completed in 2000  
Parameters Monitored:  
Results:  
References: Lindvall et al. (1999) Mining & Environment, V3, pg 855

## NORWAY

### **Hjerkinn tailings pond, Norway**

Disposal Setting: tailings pond  
Waste Material: tailings, 5-18% S  
Dates: deposition 1968-1993  
Parameters Monitored: pH, SO<sub>4</sub>, Zn, Fe, suspended solids, turbidity  
Results: pH = 7.2, SO<sub>4</sub> = 600 mg/L, Zn incr to 200 µg/L and decr to 147 µg/L after deposition ceased, Fe incr to 600 µg/L and decr to 70 µg/L after deposition ceased, suspended solids incr from 1.5 to 2.5 mg/L, turbidity incr from 1.8 to 5.1 BTU  
References: Yanful & Simms (1996)

### **Lokken Site, Sør-Trøndelag County**

Disposal Setting: underwater disposal in a man-made pond  
Waste Material: tailings, 40% S  
Dates: deposition from 1974-1987  
Parameters Monitored: pH, SO<sub>4</sub>, Fe, Cu, Zn  
Results: during deposition pH decr from 10.1 to 5.3, SO<sub>4</sub> = 800 mg/L, Fe = 200-500 µg/L, Cu incr from 200 to 800 µg/L, Zn incr from 500 to 1200 µg/L; post-deposition pH = 4-6, SO<sub>4</sub> = 100 mg/L, Cu = 0-600 mg/L, Zn = 2500 µg/L  
References: Yanful & Simms (1996); Arnesen & Iversen (1997) ICARD, V3, 1093.

### **Lokken Site, Wallenberg Mine, Sør-Trøndelag County**

Disposal Setting: flooded mine workings  
Waste Material: mine workings, pyrite  
Dates: began flooding by natural inflow in 1983  
Parameters Monitored: pH, SO<sub>4</sub>, Fe, Cu, Zn  
Results: pH increased from 2.0 to 4, Cu decreased from 530 to 1 mg/L, Zn increased from 1500 to 4000 mg/L  
References: Arnesen & Iversen (1997) ICARD, V3, 1093.

## APPENDIX 4

### Chemical Stabilization of Reactive Mine Waste Materials

A4.1.	Overview of Alkaline Amendment Studies .....	A4.3
A4.1.1.	Waste Rock .....	A4.3
A4.1.2.	Tailings and Fine-Grained Waste Rock .....	A4.4
Table A4.2.	Bibliographic annotations of research related to the use alkaline amendments .....	A4.6
Table A4.3.	Bibliographic annotations of research related to chemical stabilization of reactive mine wastes using microencapsulation .....	A4.7



#### A4.1. Overview of Alkaline Amendment Studies

Numerous studies have been conducted on the application of alkaline materials to neutralize acid released from reactive mine wastes. The purpose of this section is not to provide an extensive literature review of alkaline addition case studies. The following provides a brief overview of recent alkaline addition methodology and applications, including results from a few laboratory and field studies.

##### A4.1.1. Waste Rock

###### **Laboratory studies**

Stewart et al. (1994) reported on column studies using mixtures of fly ash, limestone, or rock phosphate with acid-producing coal refuse. The loadings of mitigative solid were varied and the tests were conducted for periods of two to three years. Coal refuse (2.3% S) was mixed with Westvaco ash mix (20% ash with an NP of 5 kg/t  $\text{CaCO}_3$ ) in ratios of 4:1 and 2:1. Both ratios produced neutral drainage (pH 8) for three years, as compared to a control pH of 1.9. In a second phase, column experiments were conducted on various mixtures of coal refuse (4% S) with the Westvaco ash (5, 10, 20, and 33% loadings) and Clinch River fly ash (20 and 33% loadings). With the Westvaco ash, only the highest loading remained neutral after two years. The Clinch River ash (which was more alkaline than the Westvaco ash) maintained neutral drainage for two years, as did a 13% limestone mixture.

Limestone requirements to prevent acid generation and metal release from gold deposit mine wastes were evaluated in a five-year laboratory column study (Day, 1994). Each column tested a different blend of waste rock and limestone with NPRs ranging from 0 to approximately 1:1. The results indicated that a ratio of at least 1:1 is required to delay the onset of drainage acidification. The 1:1.11 limestone blend successfully prevented drainage acidification throughout the study, however, did not prevent zinc release. However, increasing sulfate concentrations, slightly decreasing pH, and acidic residues in the upper portions of the column suggested that the drainage will acidify in the future. Day (1994) concluded that the actual limestone requirement to delay acidification indefinitely would be at least 2:1.

###### **Field studies**

Waste materials have been blended with acid-producing mine wastes. Evans and Rose (1995) added lime kiln flue dust (an alkaline waste product) to 360 tons pyritic shale (1.9% S) in amounts providing neutralization potential 0 to 1.71 times the acid production potential. All conditions produced acidic drainage throughout the 10-month field test, although drainage pH increased from 1.9 to 2.6 as the flue dust loading increased. Incomplete mixing of the flue dust and shale, as well as preferential flow, were cited as limitations to the effectiveness of the treatment.

With metal mine waste rock, the blending and layering of limestone is reported to be of minimal mitigative success, due to problems such as inadequate homogeneity of mixtures and preferential flow through acid-generating layers (Mehling et al. 1997). The large particle size of the waste rock, the dimensions of which can reach several feet, most likely contributes to these problems. The reactivity of limestone is also limited when particle size is large.

Whereas ratios of NP:AP have been used to determine the amount of alkaline material required for maintaining neutral drainage from waste rock piles, analysis by Kempton et al. (1997) and Morin and Hutt (2000) indicate that preferential flow has a dominant influence on the effectiveness of waste rock blending. The latter publication indicated that waste rock drainage acidity is dependent on the flow path length within acid neutralizing rock separating zones of acid generating rock. The authors' analysis indicated that waste rock with a bulk NP:AP ratio of 300:1 could release acidic drainage if appropriate neutralizing rock flow path length was not attained.

#### A4.1.2. Tailings and Fine-Grained Waste Rock

With finer mine wastes (e.g. tailings or fine mine waste) and alkaline solids a homogeneous mixture can be more readily attained and, due to more uniform particle sizing, flow tends to be more uniform than in the wide range of particle sizes in waste rock piles. Blending of alkaline solids with tailings is further attractive due to 1) greater practicality in determining tailings composition within standard operating procedures; 2) their potential for a higher degree of compositional homogeneity over extended periods; and 3) the potential to achieve more intimate blends by moderate modification of standard tailings handling procedures. Furthermore, it may be necessary to treat only those tailings above the water table, since the rate of sulfide Mineral oxidation below this level will be limited by oxygen diffusion through water.

For tailings, the operational phase is less problematic since drainage acidification tends to be limited by 1) the subaqueous environment in which the majority of tailings are disposed and 2) the typical addition of alkaline processing reagents to tailings impoundments. Furthermore, any problematic drainage that is generated can be collected and returned to the impoundment. However, after operations cease and before final closure measures are implemented, mitigation measures may be required, particularly in areas in which sulfidic tailings are not submerged. The addition of alkaline solids to tailings as the tailings are generated may be an effective, low-cost method of neutralizing acid produced during this period. It is also possible that such addition would accelerate the formation of precipitate coatings on the iron sulfide minerals responsible for acid production and, thereby, render these minerals much less reactive.

## **Laboratory studies**

Some success has been attained in mixing limestone with finer acid-producing materials in the laboratory. Lapakko et al. (1997) reported that -10-mesh limestone homogeneously mixed with finely-crushed acid producing rock neutralized acid produced by iron sulfide oxidation. Furthermore, sulfide oxidation rates decreased to the extent that host rock Mineral dissolution neutralized acid produced by iron sulfide oxidation after the limestone was depleted.

Lapakko et al. (2000) conducted laboratory tests under conditions similar to those used with the limestone blends, examining the effects of mixing rotary kiln fines (RK fines) with finely-crushed acid producing rock. Five loadings of RK fines were mixed with finely-crushed acid producing rock to produce neutralization potential (NP) added: acid potential (AP) quotients of 0, 0.11, 0.22, 0.66, and 1.1. The mixtures were subjected to wet-dry cycling for 585 weeks. The RK fines elevated pH and inhibited sulfate release for time periods which increased with the mass of RK fines present. The pH of drainage from the 1.1 quotient typically remained above 8.0 throughout the period of record.

## **Field studies**

A lime amendment was incorporated at a historical abandoned tailings deposit (Davis et al. 1999). Site examination four to five years after the amendment revealed that the pH in pore waters of amended tailings was higher than values associated with unamended tailings. The authors also noted that trace metals were removed from pore waters in the amended tailings by sorption to ferrihydrite and/or coprecipitation. The authors concluded that lime amendment “represents a viable long-term alternative to mitigate a historical problem.” Whereas this conclusion is generally consistent with the results presented, it may be premature based on the fairly short-term success. In addition, the paste pH of one tailings sample at an amended site was 4.7, despite a net carbonate value of 13 percent.

Table A4.2. Bibliographic annotations of research related to the use alkaline amendments to control generation of acidic drainages. Italicized numbers refer to reference numbers for the Reclamation Section's literature database (Papyrus).

Reference	Study Scale <sup>1</sup>	Waste Material		Alkaline Material				
		Hard Rock	Coal	Limestone/Lime	Waste Rock	Coal Combustion Waste	Municipal Waste	Paper Mill Waste
MEND (1998)	R	U	U					
Scheetz et al. (1998)	R		U			U		
Ziemkiewicz & Black (2000)	R		U			U		
Morin & Hutt (2000)	M							
Lapakko & Antonson (1991) #1355	L	U		U				
MEND (1994) #3273	L	U		U				
Day (1994) #3578	L, F	U		U				
Lapakko et al. (1997)	L	U		U				
Taboada et al. (1997)	L	U		U				
Iversen & Arnesen (1988)	O	U		U				
Davis et al. (1999) #3440	O	U		U				
Denholm & Hallam	F, O	U			U			
Morin & Hutt (1997)	L	U			U			
Mehling et al. (1997)	L, F, O	U			U			
Condon (1999)	O	U			U			
Lapakko. et al. (2000)	L	U				U		
Chtaini et al. (1997)	L, F	U						U
Coleman et al. (1997)	F		U				U	

<sup>1</sup> R= review, M=modeling, L= laboratory, F= field, O= operation

Table A4.3. Bibliographic annotations of research related to chemical stabilization of reactive mine wastes using microencapsulation to control generation of problematic drainages. Italicized numbers refer to reference numbers for the Reclamation Section's literature database (Papyrus).

Reference	Study Scale <sup>1</sup>	Microencapsulation Technique			Objective	
		Phosphate	Silicate	Other	Coating Feasibility	Drainage Quality
Evangelou (1994)	L	U			U	
Evangelou (1995) #3709	L	U			U	
Georgopoulou, et al. (1995)	L	U			U	
Evangelou (1996) #3738	L	U			U	
Roy & Worrall (1999)	L	U			U	
Conca et al. (1999)	F	U				U
Jensen et al. (1999)	O	U				U
US EPA (1999) #3688	F	U				U
Littlepage et al. (2000) #3745	F	U				U
Fytas & Evangelou (1998) #3636	L	U			U	U
Zhang & Evangelou (1998) #3637	L		U		U	
Trezek (1992)#1741	F		U			U
Chatham & Svec (1996) #3675	F		U			U
Mitchell & Anderson (2000) #3747	R, L, F		U			U
Fytas et al. (1999)	L		U		U	U
Fytas et al. (2000)	L		U		U	U
KEECO #3647	R		U		U	U
Moskalyk (1995)	L			U	U	U
Chen et al. (1999)	L			U	U	
Maki et al. (1995)	L			U		U

Reference	Study Scale <sup>1</sup>	Microencapsulation Technique			Objective	
		Phosphate	Silicate	Other	Coating Feasibility	Drainage Quality
Vandiviere & Evangelou (1998) #3113	F	U	U			U
Adams et al. (1994)	L	U		U	U	
Williams et al. (1999)	F	U	U	U		U

<sup>1</sup> R= review,M=modeling, L= laboratory, F= field, O= operation

## APPENDIX 5

### Biological Stabilization of Reactive Mine Wastes Bactericides

A5.1.	Overview of Bactericide Studies .....	A5.3
A5.1.1.	Surfactants .....	A5.3
A5.1.2.	Heterocyclic Mercaptans .....	A5.4
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### A5.1. Overview of Bactericide Studies

Since Mineral potential in Minnesota is largely associated with metallic minerals, this presentation focuses on the application of bacterial inhibitors to sulfidic mine wastes rather than coal mine wastes. Very few studies have been conducted on metal mine wastes using bacterial inhibitors. The majority of these studies evaluated the ability of surfactants such as sodium lauryl sulfate (SLS) to inhibit bacterial growth, thereby reducing drainage acidity and dissolved metal concentrations (Patterson, 1987; Sobek, 1987; Watzlaf, 1988; Parisi et al., 1994). Two studies evaluated the effectiveness of heterocyclic mercaptans (Stichbury et al., 1995; Lortie et al., 1999). The purpose of this section is to review the results of these studies in terms of parameters used to indicate bacteria inhibition (i.e. pH, acidity, chemical release, and bacteria counts).

#### A5.1.1. Surfactants

The type of surfactants used and application methods varied widely in the studies reviewed here. In three instances, the surfactant used was not identified (Patterson, 1987; Sobek, 1987). SLS was the most commonly tested surfactant, however, Watzlaf (1988) also tested potassium benzoate and potassium sorbate.

Surfactant concentrations were not always reported, although laboratory test claimed to be used to determine appropriate dosages. Based on the two studies that did report surfactant dosages, the field test (Parisi et al., 1994) used a dosage that exceeded the dosage used in unrelated laboratory tests (Watzlaf, 1988) by one to three orders of magnitude.

Test duration varied from a set of eight week laboratory experiments (Sobek, 1987) to a fifteen month field trial (Parisi et al., 1994). As a result of these differences, it was difficult to make valid comparisons between studies.

In general, surfactant effectiveness was reported in terms of the percent reduction in acidity. Application of surfactants during laboratory studies reduced acidity by 50% - 70% when adequate surfactant dosages were used (Sobek, 1987; Watzlaf, 1988). Of the two field studies reviewed, only one was considered successful with acidity reduced from 2500 mg/L to 200 mg/L, or approximately 92% (Parisi et al., 1994). Direct measurements of pH were not reported in any of the laboratory studies and rarely reported for field test sites. Parisi et al. (1994) reported that drainage from waste rock at a silver mine had an average pH of 5.6 during the fifteen month study. These reported results suggest that while drainage acidity was greatly reduced, drainage pH was probably not maintained above the environmental standard of 6.0.

In one study, surfactant effectiveness was determined by chemical release in the drainage. Patterson (1987) reported on a laboratory study in which a 50% reduction (500 to 250 mg/L) in copper release to solution was used as an indicator of SLS effectiveness (Patterson, 1987). Similarly, a field test at a silver mine reported decreases in sulfate concentrations from 1000 mg/L to 300 mg/L (Patterson, 1987).

A third method used to determine surfactant effectiveness is to estimate bacteria population size. Watzlaf (1988) used a most probable numbers method to determine that SLS and potassium benzoate inhibited bacterial growth in batch leach tests for 182 and 231 days, respectively.

#### A5.1.2. Heterocyclic Mercaptans

Two papers dealt with the application of heterocyclic mercaptans to sulfidic wastes (Stichbury et al., 1995; Lortie et al., 1999). Both papers report on the same basic study, using 2,5-dimercapto-1,3,4-thiadiazole (DMT) and 5-amino-1,3,4-thiadiazole-2-thiol (ATT) as bacterial inhibitors. Stichbury et al. (1995) focused on laboratory experiments conducted over a six week period. In the laboratory experiments, 20 g of fresh tailings and 0.5 g oxidized tailings were treated with 85 ml of inhibitor (at 100, 200, or 500 mg/L) and 5 ml of culture medium containing *T. thioparus*.

Field test plots consisted of plastic barrels (d = 53 cm, h = 82 cm) filled with fresh tailings and backfilled into an old tailings area (Lortie et al., 1999). Test plots were treated with 500 mg inhibitor per kilogram of tailings and monitored for approximately one year.

In both cases, pH values were reported as indicators of inhibitor effectiveness rather than acidity. In laboratory experiments, only one dosage of DMT (500 mg/L) resulted in a final pH above 6.0 (Stichbury et al., 1995). In all other cases, pH decreased below 5.0. After one year, cores were taken from the field test plots and cut into two sections, representing 0-25 cm and 25-50 cm depths (Lortie et al., 1999). Pore water pH in these sections decreased to 3.0 and 3.2, respectively, in the treated test plots compared to 3.5 and 4.2, respectively, in the control test plots.

The results of the laboratory experiments in terms of sulfate concentration were similar to the pH results in that sulfate production was inhibited only in the 500 mg/L DMT incubation. However, sulfate production in the field test plots was reduced from approximately 30,000 mg/L in the controls to 14,000 - 18,000 mg/L in the treated barrels. Thus, sufficient quantities of DMT appear to reduce sulfate production in these tailings by approximately 50% compared to untreated tailings.

Tailings pore waters from the field test plots were also analyzed for Fe, Al, Ca, Co, Cu, Mg, Mn, Na, Ni, and Zn (Lortie et al., 1999). Concentrations of five of these elements (Al, Co, Mg, Mn, and Na) were actually higher in the treated samples than in the controls. No significant differences were seen in concentrations of Ca, Cu, Ni, or Zn. Only iron concentrations were reduced below the levels found in the control samples (i.e. approximately 14,500 mg/L to 4000 mg/L).

The number of *T. thioparus* bacteria associated with each core from the field test plots were also estimated using a most probable numbers technique (Lortie et al., 1999). After one year, bacteria counts in the control samples had reached  $10^5$  to  $10^6$ , whereas bacteria counts in the treated samples did not exceed 380.



Table A5.2. Bibliographic annotations of research related to the use bactericides to control generation of acidic drainages. Italicized numbers refer to reference numbers for the Reclamation Section's literature database (Papyrus).

Reference	Study Scale <sup>1</sup>	Targeted Bacteria		Waste Material	
		Acidophiles	Neutrophiles	Coal	Hard Rock
Rastogi & Sobek (1986) #772	R	U		U	
Sobek (1987) #1526	R	U		U	
Benedetti et al. (1990) #1256	R	U		U	
Rastogi et al. (1990) #1255	R	U		U	
Rastogi (1995) #3515	R	U		U	
Ziemkiewicz (1995)	R	U		U	
Delaney et al. (1997)	R	U		U	U
Watzlaf (1988) #1003	L	U		U	
Kleinmann (1980) #3514	L, F	U		U	
Fox & Rastogi (1983) #3496	L, F	U		U	
Kleinmann & Erickson (1988) #1254	L, F	U		U	
Patterson (1987) #1529	L, F	U			U
Stichbury et al. (1995) #3733	L		U		U
Lortie et al. (1999) #3732	F		U		U
Sobek et al. (1990) #1257	O	U		U	
Parisi et al. (1994)	O	U		U	U

<sup>1</sup> R= review, L= laboratory, F= field, O= operation

## APPENDIX 6

### Treatment Systems for Problematic Drainage

Table A6.1.1. Bibliography of problematic drainage passive treatment studies conducted by the MN DNR .....	A6.3
Table A6.1.2. Summary of operational-scale passive treatment systems .....	A6.5
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AUSTRALIA .....	A6.14
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Table A6.1.3. Bibliographic annotations of research related to passive constructed wetland systems, including anoxic limestone drains and permeable reactive barriers .....	A6.15
Table A6.2.1. Active Treatment Systems Treating Hard Rock Mine Drainage .....	A6.18
UNITED STATES .....	A6.18
CANADA .....	A6.22
Table A6.2.2. Bibliographic annotations of research related to active treatment systems	A6.23



Table A6.1.1. Bibliography of problematic drainage passive treatment studies conducted by the MN DNR.

Strategy	Study Scale	Reference
Wetlands	Natural Wetlands	Eger, P., Lapakko, K., Otterson, P. 1980. Trace metal uptake by peat: Interaction of a white cedar bog and Mining stockpile leachate. In Proc. of the 6th International Peat Congress. Duluth, MN. Aug. 17-23, 1980. p. 542-547.
		Eger, P., Lapakko, K. 1988. Nickel and copper removal from mine drainage by a natural wetland. In Proc. 1988 Mine Drainage and Surface Mine Reclamation Conference, April 19-21, 1988, Pittsburgh, PA., V1, 301-309.
		Eger, P., Lapakko, K. 1989. The use of wetlands to remove nickel and copper from mine drainage. In Constructed Wetlands for Wastewater Treatment (June 13-17, 1988, Chattanooga, Tennessee). Hammer, D. A. (Ed), Lewis Publishers, Chelsea, MI. p. 780-787.
	Laboratory	Lapakko, K., Eger, P. 1981. Trace metal removal from Mining stockpile runoff using peat, wood chips, tailings, till, and zeolite. In Proc. 1981 Symposium on Surface Mining Hydrology, Sedimentology and Reclamation. Lexington, KY. p. 105-116.
		Lapakko, K., Eger, P. 1983. Passive treatment of sulfide stockpile runoff. In Proc. 1983 National Conference on Environmental Engineering, A. Medine and M. Anderson eds. ASCE, New York. p. 643-651.
		Lapakko, K., Strudell, J., Eger, A. 1986. Trace metal sequestration by peat, other organics, tailings, and soils: A literature review. (Final Report BuMines Contract J0205047). U.S.D.I. Bureau of Mines NTIS # PB 87-186144. 45 p.
		Lapakko, K., Eger, P. 1988. Trace metal removal from stockpile drainage by peat. In Proc. 1988 Mine Drainage and Surface Mine Reclamation Conference, April 19-21, 1988, Pittsburgh, PA. V. 1. Mine Water and Mine Waste. U.S.D.I. Bureau of Mines IC9183. p. 291-300.
	Small-scale Field	Eger, P. 1994. Wetland treatment for trace metal removal from mine drainage: the importance of aerobic and anaerobic processes. In Wat. Sci. Tech., Vol. 29, No. 4. p. 249-256.
		Eger, P., Wagner, J., Kassa, Z., Melchert, G. 1994. Metal removal in wetland treatment systems. Proc. International Land Reclamation and Mine Drainage Conference/Third International Conference on the Abatement of Acidic Drainage. Pittsburgh, PA, April 25-29, 1994.
	Operation	Eger, P., Wagner, J., Melchert, G., 1996. The use of overland flow wetland treatment systems to remove nickel from neutral mine drainage. In Successes and Failures: Applying Research Results to Insure Reclamation Success. Proc. 13th National Meeting, Knoxville, TN, May 18-23, 1996. p.580-89

Strategy	Study Scale	Reference
		Eger, P., Wagner, J., Melchert, G., 1997. The use of a peat/limestone system to treat acid rock drainage. Proc. Fourth International Conference on Acid Rock Drainage. Vancouver, B.C., Canada, May 31-June 6, 1997. p. 1195-1209
		Eger, P., Melchert, G., Wagner, J. 1999. Using passive systems for mine closure—a good approach or a risky alternative. In Mining in a new era. CD-Rom. Society of Mining Engineers Annual Meeting, Denver, CO, March 1-3, 1999. Preprint 99-38.
		Eger, P., Wagner, J., Melchert, G., Antonson, D., Johnson, A. 2000. Long term wetland treatment of mine drainage at LTV Steel Mining Company's Dunka Mine. MN DNR, Saint Paul, MN. 54p. plus appendices.
Sulfate Reduction	Barrel tests	Eger, P. 1992. The use of sulfate reduction to remove metals from acid mine drainage. In Proc. 9th National Meeting ASSMR, Duluth, MN, June 14-18, 1992. p. 563-575.
		Eger, P. 1994. The use of sulfate reduction to remove metals from acid mine drainage. Proc. International Land Reclamation and Mine Drainage Conference / Third International Conference on the Abatement of Acidic Drainage. Pittsburgh, PA, April 25-29, 1994. p. 412.
		Eger, P., Wagner, J. 1995. Sulfate reduction for the treatment of acid mine drainage: Long term solution or short term fix? In Proceedings of Sudbury '95 - Mining and the Environment. May 28-June 1, 1995, Sudbury, Ontario, Canada. p. 515-524
		Eger, P., Wagner, J. 2001. Sulfate reduction - Designing systems for long term treatment. SME Annual Meeting, Denver, CO, Feb 26-28, Preprint 01-115.

Table A6.1.2. Summary of operational-scale passive treatment systems.

UNITED STATES

**Dunka Mine Site, Babbitt, Minnesota:**

References: Eger et al. (1998) ASSMR; Eger et al. (1997) ICARD, V3, pg 1195-1209.  
Eger et al. (2000)

Mine Type: Taconite, sulfidic overburden stockpiles

Results (mg/L):

Wetland Treatment System	Parameter of Concern	Influent (mg/L)	Effluent (mg/L)
W2D-3D <sup>1</sup>	pH	7.1	7.0
	Cu	0.06	<0.005
	Ni	1.94	0.06
	Co	0.02	<0.005
	Zn	0.06	<0.005
W1D <sup>2,4</sup>	pH	7.2	7.4
	Cu	0.03	<0.005
	Ni	0.90	0.13
	Co	0.01	0.02
	Zn	0.02	<0.005
Seep 1 <sup>2,4</sup>	pH	6.9	7.2
	Cu	0.27	0.06
	Ni	7.24	3.34
	Co	0.19	0.06
	Zn	0.94	0.41
Seep X <sup>2</sup>	pH	7.1	7.2
	Cu	0.41	0.14
	Ni	1.82	0.99
	Co	0.09	0.03
	Zn	0.57	0.35
EM-8 <sup>3</sup>	pH	7.2	7.1
	Cu	0.03	0.01
	Ni	1.62	1.04
	Co	0.01	<0.005
	Zn	0.05	0.04

<sup>1</sup>average influent concentrations monitored from 1992-1994, average effluent concentrations for 1992-1998

<sup>2</sup>average concentrations from July 1995 (wetland brought online) through 1999

<sup>3</sup>average concentrations from 1998 (wetland brought online) through 1999

<sup>4</sup>"Influent" monitoring stations represent the effluent from limestone pretreatment cells intended to raise drainage pH prior to entering the W1D and Seep 1 wetlands.

Capacity: flow is generally at 3-840 L/min, but 6000 L/min possible, design capacity was for 75 L/min

Cost estimates: construction=\$1.2 million, operation=\$40,000

Other notes: Wetland treatment systems at Dunka are part of an integrated system involving stockpile capping, passive, and active treatment.

**Dunka Mine Site, Babbitt, Minnesota, continued:**

References: Lapakko et al., 1998; Lapakko and Antonson, 1990  
Mine Type: Taconite, sulfidic overburden stockpiles  
Treatment Process: pilot-scale limestone bed  
Results (mg/L): The bed was 100% efficient in neutralizing acidity in 1990, raising the mean pH from 5.0 to 6.7 and the mean net alkalinity from -85 to +22 mgCaCO<sub>3</sub>/L. Cu reduced by almost 68%, Zn reduced 42%, Ni & Co reduced 13%. After switching to larger sized limestone (to prevent clogging), treatment became ineffective.  
Capacity: The bed received 3600 m<sup>3</sup> of flow at rates of 0.276 to 1.356 L/s, averaging 0.73 L/s or about 45 bv/d.  
Cost estimates:  
Other notes:

**Ferris-Haggarty Mine, Carbon County, Wyoming:**

References: Reisinger et al. (2000) ICARD pg 1071  
Mine Type: inactive underground copper mine  
Inflow quality (mg/L): Osceola Tunnel: pH 6.5-7.0, 3-5 mg/L Cu but up to 15-20 mg/L during spring runoff  
Chute/shaft water: pH 3.5-4.0, Cu=12-23 mg/L  
Effluent quality: (approx values est. from a graph) pH >6.5, Cu <3  
Treatment process: Pilot-scale, anaerobic cell (d=15', 4' deep) filled with cattle manure (containing SULFATE REDUCING BACTERIA\*), sawdust, limestone, hay, and alfalfa.  
Capacity: gravity feed flow less than 5 gpm  
Cost estimates: none given  
Other notes:

**Lilly/Orphan Boy Mine, Elliston, Montana:**

References: Canty, M. (2000) ICARD pg 1139; #3518-Canty (1999) Mining Engineering  
Mine Type: metal mine  
Inflow quality: 2-8 gpm, pH=3, As=1.07mg/L, Al=9.69mg/L, Cd=0.33mg/L, Cu=0.32mg/L, Fe=27.7mg/L, Mn=6.21mg/L, Zn=26.1mg/L, SO<sub>4</sub>=277mg/L  
Effluent quality: tunnel: pH=7, 70-100% removal of Al, Cd, Cu, Zn; lower for Fe and As; Mn removal efficiency = 30-80%  
portal: pH=6,  
Treatment process: "in situ biological reactor" in an underground mine. Platforms containing animal by-products were suspended 30' below the static water level and in the main tunnel. Sulfate reduction increases pH (approx. 7) and decreases Eh.  
Capacity: up to 8 gpm  
Cost estimates: none  
Other notes: This was an EPA Mine Waste Technology Program (MWTP), paper covers 4 years of monitoring

## **Butte, Montana:**

### References:

McCarthy et al. (1999) 5<sup>th</sup> Internat. Conf. on Tailings and Mine Waste, 725-733.

Gammons et al. (2000) ICARD, 2, 1159-1168.

### Mine Type:

historic metal Mining district

### Inflow quality (µg/L):

Metro Storm Drain: As=6, Cd=42, Cu=100-500, Pb=3, Zn=12176

Butte Reduction Works Wetlands: As=4.2, Cd=39.4, Cu=328.8, Pb=3.2, Zn=11273, pH=7.6

Colorado Tailings Wetlands: As=8.4, Cd=34.4, Cu=286.1, Pb=2.0, Zn=9474, pH=7.2

### Effluent quality:

Metro Storm Drain: As=25-32, Cd#1, Cu=15-36, Pb=3, Zn=242-1516

Butte Reduction Works Wetlands: As=5.1, Cd=25.0, Cu=84.7, Pb=2.7, Zn=6995.2, pH=7.9

Colorado Tailings Wetlands: As=10.3, Cd=1.5, Cu=22.0, Pb=2.1, Zn=68, pH=8.3

### Treatment process:

Metro Storm Drain: four parallel cells - Cell 1=horizontal flow, anaerobic, gravel-nonorganic substrate; Cell 2=horizontal flow, anaerobic, gravel-20% compost substrate; Cell 3=vertical flow, anaerobic, gravel-50% compost substrate; Cell 4=same as cell 1 but half the size/RT - effluents run through aerobic ponds & back to the drains

Butte Reduction Works Wetlands: three open water cells separated by two treatment walls (one of d<sub>50</sub>=6" river rock, other of 90% rock/10% compost) which serve as subsurface flow zones where physical, chemical, and biological processes take place.

Colorado Tailings Wetlands: same as BRW, but with lime addition

### Capacity:

MSD: 10 gpm per cell

BRW: receives approx 50-150 gpm from Missoula Gulch baseflow (urban storm sewers)

CTW: captures & treats contaminated alluvial groundwater at approx 125 gpm

### Cost estimates:

### Other notes:

## **Calliope (abandoned) Mine, Butte, Montana:**

### References:

Zaluski et al. (2000) ICARD pg 1169

Zaluski et al. (1999) In: Phytoremediation and Innovative Strategies for Specialized Remedial Applications. 5<sup>th</sup> Internat. In Situ and On-Site Bioremediation Symp. pg 205-210.

### Mine Type:

metal mine

### Inflow quality:

pH=4, Al=5.7mg/L, As=26µg/L, Cd=38µg/L, Cu=1.7mg/L, Fe=8.7mg/L, Mn=3.0mg/L, Zn=7.9mg/L, SO<sub>4</sub>=165mg/L, alk=<10mg/L CaCO<sub>3</sub>

### Effluent quality:

pH approx 8; Zn, Cd, Cu and Mn decreased significantly but weren't quantified well

### Treatment process:

Organic carbon pretreatment lowers Eh, organic carbon substrate supplies food for SULFATE REDUCING BACTERIA\*, crushed limestone pretreatment provides alkalinity and increases pH, cobbles are used for the stable substrate for SULFATE REDUCING BACTERIA\* growth. All cells were 12' wide

Cell II - below grade, 5' organic pretreatment, 5' crushed limestone pretreatment, 5' organic substrate, 50' cobbles  
 Cell III - below grade, 5' organic substrate, 50' cobbles  
 Cell IV - above grade, 5' organic pretreatment, 5' crushed limestone pretreatment, 5' organic substrate, 50' cobbles

Capacity: 1 gpm, 5.5 day residence time

Cost estimates:

Other notes: in general, pretreatment and below grade gave the best results

### **Mike Horse Mine, Lewis & Clark County, Montana:**

References: Anderson & Hansen (1999) 5<sup>th</sup> Internat. Conf. on Tailings and Mine Waste, 715-724.

Inflow quality (mg/L): pH = 5.5-6.0, Fe=60-180, Zn=20->150, alkalinity < 100, Cd=0.25, Cu=2.3-3.0, Pb=0.35-0.37, Mn=76-86, SO<sub>4</sub>=2000-3000

Effluent quality: pH = 6.7, Fe=0.31, Zn=14, alkalinity < 100, Cd<0.001, Cu=0.008, Pb<0.002, SO<sub>4</sub>=2200

Treatment process: went online in 1996; ALD within the mine (120 m<sup>3</sup> limestone, 15-30 cm) 6 jet pump aeration system 6 lined oxidation/precipitation pond (2300 m<sup>3</sup>) 6 sand filter 6 anaerobic gravel wetland (0.8 ha) 6 aerobic wetland

Capacity: 1.95-4.40 L/s

Cost estimates:

Other notes:

### **Big Five Wetland, Idaho Springs, Colorado:**

References: Wildeman et al. (1993) Wetland Design for Mining Operations

Mine Type: abandoned metal mine

Inflow quality (mg/L): pH=2.8-3.2, Mn=29-35, Fe=33-60, Zn=9, Cu=0.5-0.7

Effluent quality: (approx. values, 1 yr of monitoring) pH=6, Mn=20-35, Fe<20, Zn<1, Cu<0.05

Treatment process: Concrete structure (W=10', L=60', D=2') divided into three 20', 30 mil Hypalon-lined, sections separated by 2x6 treated wood. Substrates for cells A, B, and C were mushroom compost (50/50 animal manure/barley mash waste); equal parts peat, aged steer manure, decomposed wood shavings and sawdust; and the same as cell B underlain by 4-6" of 2-3" limestone rock, respectively. Cattail, sedge, and rush species were transplanted into the cells. Cell A was modified to increase the flow path. Cell B was modified into upflow and downflow cells, and cell D (aerobic polishing) and cell E (downflow, subsurface wetland) were constructed using cell B substrate.

Capacity: #2.6 L/min

Cost estimates:

Other notes: Winter operation possible if delivery systems are insulated, cells receive winter sun, top insulated with hay and plastic

### **West Fork Unit, Reynolds Co, Missouri:**

References:	Gusek et al. (2000) ICARD, 2, 1133-1137 Gusek et al. (1999) In: Phytoremediation and Innovative Strategies for Specialized Remedial Applications. 5 <sup>th</sup> Internat. In Situ and On-Site Bioremediation Symp.
Mine type:	Pb-Zn
Inflow quality (mg/L):	Pb=0.4, Zn=0.36, Cd=0.003, Cu=0.037, pH=7.94
Effluent quality:	Pb=0.03-0.05, Zn=0.06-0.09, Cd<0.002, Cu<0.008, pH=6.6-7.8, H <sub>2</sub> S=0.01-0.03, BOD<1-3
Treatment process:	two stage biotreatment; 1) anaerobic, sulfate reduction for Pb removal and 2) aerobic rock filter/wetland to reoxygenate & polish accomplished in five major parts - settling pond (3030 m <sup>2</sup> surface, 1930 m <sup>2</sup> base, 3' deep), two anaerobic cells (1930 m <sup>2</sup> surface, 1390 m <sup>2</sup> base, cow manure sawdust, limestone & alfalfa substrate), rock filter (5900 m <sup>2</sup> by 1' w/ limestone cobbles), and an aeration pond (8000 m <sup>2</sup> ).
Capacity:	1200 gpm
Cost estimates:	construction approx. \$500,000
Other notes:	

### **Tennessee Valley Authority:**

References:	Wildeman et al. (1993) Wetland Design for Mining Operations; Skousen et al. (1998)
Mine Type:	Coal power utility
Inflow quality (mg/L):	Generally: (approx. values from all 13 systems) pH = 3.5-6.2, Fe = 0.7-150, Mn = 4-17, acidity = 350, alkalinity=0 Nine systems: pH=4-6.5, Fe<70, Mn<17, Al<30, net alkalinity=35-300 Four systems: Fe>170, no net alkalinity An additional system: Fe<0.7, Mn=5.3
Effluent quality:	Generally: pH = 3-7, Fe = 0.4-83, Mn = 0.6-13, acidity=40, alkalinity=100 Nine systems: meet discharge standards Four systems: two use an ALD, two require NaOH treatment An additional system: ineffective at Mn removal
Treatment process:	Aerobic wetland In general: ALD (alkalinity) 6 deep pond (Fe/Mn removal) 6 deep marsh (Fe removal) 6 shallow marsh (Mn removal) 6 rock filter (Mn removal) 6 alkaline bed (pH increase) 6 polishing cell ALD: trench (W=10-15', L=260', D=5') installed upstream from the impoundment in a seepage area, filled with 3/4 - 1-1/2" limestone, covered with two layers of 10 mil plastic, geofabric, and 2' local soil, seeded, mulched, and fertilized
Capacity:	maximum flow ranged from 250 - 7700 L/min, Fe load # 21 gdm, Mn load # 2gdm
Cost estimates:	TOTAL \$50,000 = design \$6000 + permitting \$1000 + land acquisition \$6000 + construction \$22,000 + operation & maintenance \$9000 + misc. \$6000 ALD alone cost \$19,000
Other notes:	additional references = Brodie et al. 1991, 1992, 1993

### **Various sites in West Virginia:**

References: Faulkner & Skousen (1994), Pitt, 2, 250-257.

Mine type: abandoned coal mines

Results:

Treatment System	Parameter of Concern	Influent (mg/L)	Effluent (mg/L)	% Reduction
Wetlands (N=5)	pH	2.5-4.0	3.1-5.4	---
	Acidity	106-2388	56-1366	3-76
	Alkalinity	---	---	---
	Fe	10-376	2-86	62-80
	Mn	8-51	7-50	0-11
	Al	9-206	7-163	25-63
	SO <sub>4</sub>	217-2821	227-2660	---
ALDs (N=19)	pH	2.6-5.7	3.4-6.8	---
	Acidity	170-2210	0-1000	11-100
	Alkalinity	0-23	0-340	---
	Fe	22-570	1-360	---
	Mn	3-200	2-160	---
	Al	0-287	0-100	---
	SO <sub>4</sub>	101-2900	45-2000	---

Treatment process: Wetlands: In general, 0.6 to 1 m of organic material composed of peat and hay overlie about 15-30 cm of limestone. Hay bales were usually used as barriers to slow and direct flow. Two wetlands (S. Kelly and Z&F) were designed to encourage subsurface flow.

ALDs: From the bottom, constructed with 10- to 20- mil liners, filter fabric, gravel-sized limestone, and hay bales separated by the filter fabric. Liners were wrapped around the limestone and hay.

Capacity: Wetlands: 17-98 L/min, 408-1417 m<sup>2</sup>

ALDs: 4-87 L/min

Cost estimates: Wetlands: max. installation cost for the smallest wetland = \$225,000, \$110,000 reported for another.

Other notes:

### Various sites in Appalachia:

References: Hedin & Watzlaf (1994), Pitt, 1, 185-194.

Mine type: abandoned coal mines

Results:

Treatment System	Parameter of Concern	Influent (mg/L)	Effluent (mg/L)
ALDs (N=21)	pH	2.3-5.9	3.6-6.7
	Alkalinity	0-33	0-469
	Fe	3-1416	0-625
	Mn	4-136	2-132
	Al	0-486	0-152
	SO <sub>4</sub>	430-6719	55-5432

Design Criteria: Limestone mass = 35-945 mt, Mass per flow = 1.0-324 mt/L/min

Capacity: Flow = 0.5-218 L/min, detention time = 5-1588 hrs

Cost estimates:

Other notes: mass of limestone required,  $M = Q\rho_b t_d / V_v + QCT/x$   
where, Q=flow,  $\rho_b$ =bulk density of limestone,  $t_d$ =detention time,  $V_v$ =bulk void volume (decimal), C=expected alkalinity (mg/L), T=design life,  $x$ =CaCO<sub>3</sub> content of limestone (decimal)

## CANADA

### Greater Vancouver Area:

References:	McGregor et al. (2000) ICARD, pg 1227 McGregor et al. (1999) SUD, pg 645
Inflow quality (µg/L):	tidal flow, pH=6.36, Cd=15.9, Cu=4510, Ni=118, Pb=3.8, Zn=2396
Effluent quality:	pH=6.57, Cd<0.1, Cu=7.7, Ni=6.5, Pb=0.7, Zn=27.5
Barrier description:	reactive material = 85% pea gravel, 15% compost (by vol.) and a trace amount of limestone; dimensions = 6.7m deep, 10m long, 2.5m thick
Capacity:	maximum flow estimated at 66 cm/d
Cost estimates:	
Other notes:	residence time estimated at 3.8 days; measurements taken after 21 months

### Bell Copper Mine, near Smithers, British Columbia:

References:	Sobolewski et al. (1995) Mining and the Environment, V2, 683-692.
Mine Type:	Cu
Inflow quality (mg/L):	source #1: pH=6-8, Cu=0.3-1.0; source #2: pH=3.5, Cu=35-50; SO <sub>4</sub> >2000 for both
Effluent quality:	for about 6 weeks after receiving high Cu flow...pH=7.1-7.5, Cu generally less than 1, spikes up to 10
Treatment process:	(experimental systems) large (300 m <sup>2</sup> ) and small (75 m <sup>2</sup> ) ponds fertilized with manure and planted with floated peat mats
Capacity:	large pond: 8 L/min, retention time = 12 days small pond: 2 L/min, retention time = 22.5 days
Cost estimates:	
Other notes:	

### Makela Test Cell System-INCO Copper Cliff's tailings near Sudbury, Ontario:

References:	Fyson et al. (1995) Mining and the Environment, V2, 459-466.
Mine Type:	
Inflow quality (mg/L):	pH=5.65-6.04, Al<1, Cu<1, Fe=220-420, Ni=17.1-74.4, S=800-869
Effluent quality:	pH=6.1-6.3, acidity- 28-339 Al- 1, Cu- 0.01-0.22, Fe- 1-7.6, Ni- 0.8, S- 298-450
Treatment process:	ARUM = 'acid reduction using microbiology'; four cells constructed in series (100m x 20m) with two holding ponds between cells 2&3 and cells 3&4; Cell 1= oxidation/precipitation, Cell 2= oxidation/precipitation, Cell 3= ARUM w/ floating cattail rafts, Cell 4=ARUM w/ floating cattail rafts
Capacity:	1 L/min, retention time = 168 days
Cost estimates:	
Other notes:	

### **Nickel Rim, Sudbury, Ontario:**

References:	Benner et al. (2000) ICARD, pg 1221 Mayer et al. (1999) Sud, pg 145 (modeling) Benner et al. (1997) GW Monitor. and Remed. Benner et al. (1997) ES&T Blowes et al. (1995) Sud, pg 979 (test cell)
Inflow quality:	flow=49ft/yr, pH=4-6, SO <sub>4</sub> =2400-4600mg/L, Fe=250-1300mg/L, Ni<500µmol/L, alkalinity=0-50mg/L (as CaCO <sub>3</sub> )
Effluent quality:	pH=7.0, SO <sub>4</sub> =200-3600mg/L, Fe1-40mg/L, Ni<4 µmol/L, alkalinity=600-2000mg/L
Barrier description:	dimensions = 15m long, 3.6m deep, 4m thick in the direction of flow; substrate = 40% municipal compost, 40% leaf compost, 20% wood chips; pea gravel mixed in to maintain hydraulic conductivity (50/50 organic/gravel)
Capacity:	16 m/yr
Cost estimates:	materials and installation = US\$ 30000, approx. half for each
Other notes:	controlling factors = ground water temp., residence time, organic carbon reactivity; removal efficiencies (linked to organic carbon reactivity) have been declining over the 38 months of operation

### **Halifax International Airport:**

References:	Bechard et al. (1995) SUD, pg 545 Bechard et al. (1991) ICARD, pg 171
Inflow quality (mg/L):	see note below
Effluent quality:	see note below
Barrier description:	three clay lined cells built in sequence; dimensions of each cell = 16.8m long, 1m wide, 1m deep; substrate = 1 part planer shavings to 3 parts partially decomposed straw, 54kg sucrose, 6kg urea fertilizer, 1kg phosphate rock
Capacity:	23 L/min
Cost estimates:	
Other notes:	The cells were not very effective. Best results were observed between days 252 and 336 (less than three months) out of >500 days of monitoring.

## AUSTRALIA

### Hilton Mine, northern Australia:

References: Jones et al. (1995) Mining and the Environment, v2, pg 755-763.  
Mine Type: Ag, Pb, Zn  
Inflow quality (mg/L): Zn>6.5, Mn - 3, pH - 7  
Effluent quality: Zn - 0.3  $\mu$ M, Mn<0.06, pH - 8  
Treatment process: Aerobic metals removal: mine water is pumped to surface settling ponds, flows freely down a hillside to an earthen channel leading to an evaporation pond, and flows down another channel to the Hilton Pump Weir, which pumps water to the ore processing plant  
Capacity: up to 5000 m<sup>3</sup>/d  
Cost estimates:  
Other notes:

## EUROPE

### Spain:

References: Ordóñez et al. (2000) 5<sup>th</sup> ICARD, V2, pg 1121-1129.  
Inflow quality (mg/L): pH=3.4, DO=4.3, NH<sub>4</sub><sup>+</sup>=19.1, SO<sub>4</sub><sup>2-</sup>=7779, Al=135, FeII=4.6, FeIII=2.6, Mn=140  
Effluent quality: pH=7.2, DO=0.2, NH<sub>4</sub><sup>+</sup>=92, SO<sub>4</sub><sup>2-</sup>=7193, Al=1.0, FeII=0.2, FeIII=1.8, Mn=55  
Treatment process: SAPS (21 m<sup>3</sup> tanks filled w/ 1m limestone, 0.5m of 70% cow manure, 30% straw) 6 ALD (20m x 1m x 1m, isolated w/ liner) 6 oxidation cascade (3m long, 7-10cm high steps) 6 sedimentation pond (6m x 4m x 1m) 6 substrate wetland (20m x 8m x 1.2m, lined, 60% cow manure, 20% straw, 20% sand) 6 limestone filter  
Capacity: regulated flow at 200 L/hr  
Cost estimates:  
Other notes:

Table A6.1.3. Bibliographic annotations of research related to passive constructed wetland systems, including anoxic limestone drains and permeable reactive barriers. Italicized numbers refer to reference numbers for the Reclamation Section's literature database (Papyrus).

Reference	Study Scale <sup>1</sup>	Passive Treatment Components					
		ALD	Anaerobic Wetland	Aerobic Wetland	Settling Pond	Reactive Barrier	Bio-reactor
Brodie et al. (1992) #702	O	U					
Hedin & Nairn (1992) #701	R	U					
Kleinmann & Hedin (1993) #1747	R	U					
Watzlaf & Hedin (1993) #703	L	U					
Hedin et al. (1994) #2911	R, M	U					
Brant et al. (1995)	L, F	U					
Wildeman et al. (1997)	R, L, F	U					
Cravotta & Trahan (1999)	O	U (oxic)					
Eger et al. (1997)	O	U	U				
Brodie et al. (1991) #592	O	U		U			
Brodie (1991) #597	O	U		U			
Skousen (1991) #598	R, O	U		U			
Brodie et al. (1992) #704	O	U		U			
Skousen & Faulkner (1992) #595	O	U		U			
Ordenez et al. (2000)	L, F	U	U		U		
Kuyucak & St-Germain (1994)	L	U	U				U
Anderson & Hansen (1999)	O	U	U	U	U		
Kalin & Smith (1991) #1390	L, F		U				
Eger (1992) #2571	F		U				
Borek et al. (1995)	F		U				

Reference	Study Scale <sup>1</sup>	Passive Treatment Components					
		ALD	Anaerobic Wetland	Aerobic Wetland	Settling Pond	Reactive Barrier	Bio-reactor
Eger & Wagner (1995) #2879	F		U				
Fyson et al. (1995)	F		U				
Sobolewski et al. (1995)	F		U				
Willow et al. (1998)	L		U				
Kalin et al. (1999)	L		U				
McCarthy et al. (1999)	F		U				
Tisch et al. (1999)	L		U				
Marchand & Silverstein (2000)	L		U				
Reisinger et al. (2000)	F		U				
Schmiermund (2000)	F		U				
Gammons et al. (2000)	O		U		U		
Eger et al. (2000)	F			U			
Jones et al. (1995)	O				U		
Blowes & Ptacek (1992)	L, M					U	
Bechard et al. (1995)	F					U	
Blowes et al. (1995)	F					U	
Waybrant et al. (1995)	L					U	
Benner et al. (1997)	O					U	
Waybrant et al. (1997)	M, L					U	
Blowes et al. (1998) #3764	R					U	
Benner et al. (1999) #3437	M					U	
Conca et al. (1999)	L					U	
Martin & Kempton (1999)	M, L					U	
Mayer et al. (1999)	M					U	
McRae et al. (1999)	L					U	
Astrup et al. (2000) #3852	L					U	

Reference	Study Scale <sup>1</sup>	Passive Treatment Components					
		ALD	Anaerobic Wetland	Aerobic Wetland	Settling Pond	Reactive Barrier	Bio-reactor
Chilakapati et al. (2000) Environ. Sci. &Tech.	M					U	
Greenwald et al. (1999)	F					U	
McGregor et al. (1999)	O					U	
Benner et al. (2000)	O					U	
McGregor et al. (2000)	O					U	
Phillips et al. (2000) #3853	F					U	
Su & Puls (2001)	L					U	
Yabusaki et al. (2001)	F					U	
Kleinmann et al. (1991) #1284	F						U
Kuyucak & St-Germain (1994) #2874	L						U
Bechard et al. (1995)	F						U
Beaulieu et al. (1999)	L						U
Canty (1999a)	F						U
Canty (1999b)#3518	F						U
Gusek et al. (1999)	O				U		U
Kolmert et al. (1999)	L						U
Zaluski et al. (1999)	F						U
Canty (2000)	F						U
Gusek et al. (2000)	O				U		U
Johnson et al. (2000)	L						U
Zaluski et al. (2000)	F						U

<sup>1</sup> R= review/survey, M=modeling, L= laboratory, F= field, O= operation

Table A6.2.1. Active Treatment Systems Treating Hard Rock Mine Drainage

UNITED STATES

**LTV Steel Dunka Mine, MN:**

References:	<ol style="list-style-type: none"> <li>1. LTV Steel Mining Company Dunka Mine, Updated Final Closure Plan, Rev. 4, Mar. 15, 1996.</li> <li>2. Dunka Data Summary: 1976-1993, Draft, MN DNR, Div. of Minerals, March 15, 1995.</li> </ol>
Inflow quality:	pH generally neutral, occasional acidic drainage from Seeps 1 & 3, Ni = 1-20 mg/L, occasionally as high as 30 mg/L, Cu = 0-1 mg/L
Treatment process:	<p>Lime (CaO) is added to the water to bring the pH up to at least 10, creating base metal carbonate and/or sulfate precipitates. The precipitate is treated with both cationic and anionic flocculents to enhance settling. The water then passes into a plate type thickener. The flocculated solids settle to the bottom of the thickener and form a sludge which is pumped to a holding tank. From there, the sludge is pumped to a pressure filter where the solids are retained on a filter media forming a filter cake for disposal.</p> <p>The filtrate is pumped back to the system for reprocessing. The clear water overflow from the plate thickener passes through a sand filter which serves as a final polishing step. The water then passes to a pH adjustment tank where a small amount of sulfuric acid is added to bring the pH back to the compliance level range of 6.5 to 8.5. In actuality, the pH is controlled to 7.0.</p>
Capacity:	Annual flow from EM8 (roughly 2/3 of flow) and Seeps 1, 3, & X = approximately $3 \times 10^6$ L/yr, which flows to a collection basin and into the treatment plant. Seep 1 flow first passes through a limestone/peat pretreatment system.
Cost estimates:	Operation = roughly \$200,000/yr, Construction (equalization pond, pump stations, treatment plant) = \$1,494,000
Other notes:	LTV contact person - Jason Augenes, 218-225-4364

**Hibbing Research Facility, MN:**

References:	MN DNR, unpublished data
Inflow quality:	pH 4, Cu=50-150 ppm, Ni=100-300 ppm, Co=10-30 ppm, Zn=10-30 ppm
Effluent quality:	pH 7.25-9.0, Cu=0.001-0.050 ppm, Ni=0.137-2.1 ppm, Co=0.001-0.1 ppm, Zn=0.001-1.34 ppm, SO <sub>4</sub> =30.4-421 ppm
Required limits:	pH=6.5-8.5, Cu=0.5 ppm, Ni=3.0 ppm, Co=1.0 ppm, Zn=4.0 ppm
Treatment process:	The treatment system is fully automated using a computerized control system to operate sensors, timers, and other electronic equipment to treat and discharge water generated at the research site. The water is neutralized in a 500 gal. mixing tank using magnesium hydroxide. The sludge produced is allowed to settle and the treated water overflows (i.e. gravity flow) to a settling basin and subsequently is pumped to the Hibbing sewer system. The sludge is pumped to a 500 gal. holding tank and then dewatered with a filter press. The sludge is landfilled after being analyzed and meeting TCLP limits.
Capacity:	1.5 gpm
Cost estimates:	Operation =\$12,000/yr, Building construction = \$65,000; Treatment system = \$30,000

Other notes:

**Eagle Mine:**

Contact: Dan Scheppers, State of Colorado, Dept of Health, 303-692-3398  
Inflow quality:  
Treatment process: Lime precipitation of heavy metals with recirculation of sludge  
Capacity: 160 gpm treated on average  
Cost estimates: Treatment plant run by the responsible party, therefore no figures available  
Other notes: additional contact = Gene Taylor, Project Manager, EPA, 303-312-6536

**Summitville:**

Contact: Angus Campbell, State of Colorado, Dept of Health, 303-692-3385  
Inflow quality: pH = 3-3.2, Cu up to 40 mg/L (these are the monitoring parameters), also contains Fe, Mn, Zn, and Al  
Treatment process: Hydrated lime precipitation process involving sludge recycle. Water enters a mixing tank that also receives lime (midpoint) and recycled sludge (bottom?). The mixture flows via gravity feed to a reactor and onto a 60' wide x 20' high thickener. A polymer is added to the flow just before the thickener to aid flocculation. Thickener underflow, 7-9% solids, is recycled to the mixing tank. The remaining sludge goes through a filter press and is disposed of on site in cells constructed in open pits. The sludge meets TCLP limits and is not considered hazardous. The small amount of NP provided by the sludge is considered beneficial to the entire system.  
Capacity: The plant runs 24 hours a day during the summer months (approx. 9 month operating season). It consists of 2 "trains", each receiving 550 gal/min inflow and 100 gal/min sludge = 650 gal/min per train or 1300 gal/min total flow. From April 27 through November 12, 1999, they treated 262,996,843 gal of water.  
Cost estimates: Operating costs approximately 2.5 million \$/year. No good information on capital costs because they renovated an existing processing plant from the 1960s. Renovation consisted mainly of modifying existing tanks and plumbing for mixing and aeration capabilities etc.  
Other notes: Most of this information is contained in an Annual Report. Also, they will be testing several passive systems at Summitville during summer 2000 (a saturated organic limestone cell, zeolite system -actually done by a contractor-, and Aquafix).  
Additional Contacts: Victor Ketallapper, EPA, 303-312-6578

**Argo Tunnel:**

Contact: Mary Scott, State of Colorado, Dept of Health, 303-692-3413  
Inflow quality: pH = 2.8, Fe = 180 mg/L, Mn = 100 mg/L, Zn = 70 mg/L, Al = 25 mg/L, Cu = 7 mg/L, total load = 1200 lb metals/day  
Treatment process: Chemical precipitation using 50% NaOH, more expensive but easier to handle except for the fact that the entire system must be heated because the NaOH solution freezes at 54 EF. Originally intended for the plant to be fully automated but that hasn't worked out so far (since 1998). Crews have worked shifts twenty four hours a day since operation began.

Inflow enters a rapid mixing tank, where it is mixed with a 50% NaOH solution to elevate the pH to 9.9. Water then flows into an Infilco Degremont Process Tank, which is essentially two tanks stuck together. The first unit is the particle building tank where water is circulated to enhance particle interaction. A polymer and about 5% of the sludge (depending on flow, e.g. 200 gpm flow requires approx. 10 gpm sludge) from the clarifier are added here to improve flocculation. This product flows into the second unit, a 12' diameter clarification tank. Sludge from the bottom of this unit is 3-4% solids. Sludge is pumped to liquid sludge storage and onto a filter press (mechanical dewatering). The filter cake is roughly 16% solids and disposed of in a municipal landfill. They are looking at building an on-site sludge repository but that may be a couple years away yet.

Clarified water (top of the clarification unit) has low turbidity (approx. 0.2 ntu) but still contains about 1 mg/L Mn. Therefore it is run through a mixed media gravity filter, which decreases Mn to 0.2 mg/L. The pH of the water is then adjusted to 8 using CO<sub>2</sub>.

Capacity: Average inflow is approximately 320 gpm, ranging from roughly 250 gpm during the winter months to 600 gpm during the spring thaw. They can handle up to 700 gpm on two drains (350 gpm each). However they only have about twelve hours worth of storage capacity in the event of a total shut down. On average they use 1000 gal/day of the 50% NaOH solution, producing 17 yd<sup>3</sup>/day.

Cost estimates: Design = \$1 million in engineering costs, capital = \$1 million in equipment, construction = \$3.8 million, operation = \$1.2 million per year.

Other notes:

### **California Gulch Superfund Site, Leadville drainage:**

Contact: Mike Holm, EPA, 303-312-6607

Inflow quality: pH ranges from 2.5 to 7, typical is 4-4.5

Treatment process: NaOH precipitation followed by pH adjustment using H<sub>2</sub>SO<sub>4</sub>. Sludge tends to fail TCLP and must be disposed of as hazardous waste. They are doing some testing of fixation compounds to bind metals so that the sludge can be disposed of as an industrial solid rather than hazardous waste.

Capacity: Can treat up to 2000 gpm, typically treats 1500-1700 gpm. However, subsidence in the area is changing the hydrology and the flow from this tunnel is increasing.

Cost estimates: Operation = \$900,000 per year, Sludge disposal = \$100,000 per year

Other notes: Automated operation run by the Bureau of Reclamation. Treats drainage from one of the tunnels in California Gulch. They are also looking at options to treat contaminated surface water around Leadville. Additional research in the area of phytoremediation is being conducted on a field scale and looks promising (removes approximately 70% of metals if pH >5.5). see EPA Tech Trends paper by Brad Littlepage and Duane Johnson.

Additional Contacts: Brad Littlepage, plant operator at 719-486-2035.

**California Gulch Superfund Site, Yak Tunnel (Leadville):**

Contact: Mike Holm, EPA, 303-312-6607  
Inflow quality: pH ranges from 2.5 to 7, typical is 4-4.5  
Treatment process: NaOH precipitation followed by pH adjustment using H<sub>2</sub>SO<sub>4</sub>.  
Capacity: 2000 gpm maximum. However, flow from this tunnel is decreasing. Currently the plant operates 3-4 days per week, treating approximately 400 gpm.  
Cost estimates:  
Other notes: Manual operation run through a joint venture between Asarco and Resurrection Mining Companies. Treats drainage from the other California Gulch tunnel.  
Additional Contacts: Gary Slifica, Asarco, 719-486-1056,

## CANADA

### Equity Silver:

Reference: Billings 2000 Conference-CD ROM  
Inflow quality: Cu = 116 mg/L, Fe = 1340 mg/L, Zn = 154 mg/L  
Treatment process: Lime  
Capacity: 60 - 150 L/s, approximately 5000 tonnes lime  
Cost estimates: \$1.2 million/year  
Other notes:

Reference: Aziz & Ferguson, Van97, V1, pg 181  
Inflow quality: average pH = 2.62, acidity = 8180 mg/L, Cu = 116 mg/L, Fe = 1340 mg/L, Zn = 154 mg/L

Treatment process: 'ARD collection ditches flow into two collection ponds. The upper pond was designed for extra storage during high flow periods. The lower pond has a pumphouse that can pump a maximum of 4850 USG/min. The ARD is normally pumped from this pond to a storage pond located beside the treatment plant, but a portion of the flow can also be pumped directly to the plant for processing.

Neutralization is achieved by adding quick lime slurry to a pH between 8.0 to 8.5. The ARD and lime slurry are combined in the first of three reaction tanks in series with a combined retention time of 20 minutes. Constant agitation is required to ensure maximum lime efficiency. The third reaction tank contains pH probes that control the addition of the lime slurry. Maximum plant throughput is 2150 gal/min.

From the treatment plant the neutralized slurry is discharged to settling ponds where the metals drop out as a metal hydroxide sludge and the treated supernatant decants to a holding pond. During the freshet period when regional creeks are at their maximum flows the treated water is released to the environment using dilution ratios that are specified in the Equity water permit. The ARD sludge is pumped to the Main Zone pit for long term storage.'

Capacity: 'Over the last nine years (pre-1997) the average volume of ARD collected and treated at Equity annually has been approximately 880,000 m<sup>3</sup>. The lime required to treat this volume of ARD depends on the acidity, but has averaged 5,060 tonnes/yr.'

Cost estimates: Altogether the annual cost of running the ARD collection and treatment system averages around \$1.1 million of which the lime represents 70% of the cost.'

Other notes: additional contact = Bill Price, 250-847-7389

Table A6.2.2. Bibliographic annotations of research related to active treatment systems.  
 Italicized numbers refer to reference numbers for the Reclamation Section's  
 literature database (Papyrus).

Reference	Active Treatment System				
	Neutralization	Chemical Precipitation	Ion Exchange	Membrane Filtration	Biologica l
US EPA, (1983)	U				
Eger et al., (1993)	U				
Kuyucak et al., (1995)	U				
Murdock et al., (1995)	U				
Orava et al., (1995)	U				
Aube & Payant, (1997)	U				
Aziz & Ferguson, (1997)	U				
Miedecke et al., (1997)	U				
Poirier & Roy, (1997)	U				
Aube, (1999)	U				
Aube & Zinck, (1999)	U				
Ericksen et al., (1999)	U				
Willow & tenBraak, (1999)	U				
Zinck & Aube, (2000)	U				
Zinck & Griffith, (2000)	U				
Robertson & Rohrs, (1995)	U	U			
Berg & Arthur, (1999)	U	U		U	
Smit, (2000) #3918	U	U			
Lapakko, (1993)		U			
Mitchell & Atkinson, (1995) Mitchell & Wheaton, (1999) Mitchell et al., (2000)		U			
Rybock et al., (1999) #3672		U			
McKinnon et al., (2000)		U			
Aube & Stroiazzo, (2000)		U			

Reference	Active Treatment System				
	Neutralization	Chemical Precipitation	Ion Exchange	Membrane Filtration	Biological
Choung et al., (2000)		U			
Sato & Robbins, (2000)		U			
Leppert et al., (1990) #802			U		
Gussmann et al., (1991) #1357			U		
Vos & O'Hearn, (1993) #1415			U		
Schultze et al., (1994)			U		
Riveros, (1995)			U		
Gilbert et al., (1999)			U		
Metre-General Inc.			U (chelation)		
Sastri & Ashbrook, (1976) #3574				U	
Blackshaw et al., (1983) #1185				U	
Awadalla & Hazlett, (1992) #2177				U	
Stewart et al., (1997)				U	
Nakamura, (1988) #1488					U
Dvorak et al., (1991) #1387					U
Borek et al., (1995)					U
de Vegt et al., (1997; 1998) #2781; #3129					U
Diaz et al., (1997)					U
Rowley et al., (1997)					U
Dijkman et al., (1999)					U
Kolmert et al., (1999)					U
Federal Remediation Technology Roundtable		U	U		U
Bowell, (2000) #3748		U	U	U	U

## APPENDIX 7

### Integrated Case Studies

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A7.4.	Additional sources of information regarding integrated environmental mine waste managment system case studies	A7.14



A7.1. Dunka Mine, Minnesota.

Table A7.1.1. Dunka rock stockpile summary (MN DNR, 1996).

Stockpile	Stockpile Size		Chemical Composition		
	Area (ft <sup>2</sup> x 10 <sup>6</sup> )	Mass (LT x 10 <sup>6</sup> )	% NiO <sub>2</sub>	% CuO	% S
8011	3.73	20.50	0.03	0.06	0.23
8013	3.58	17.75	0.03	0.09	0.24
8015	NA	4.80	0.02	0.11	1.00
8031	3.17	9.50	0.04	0.10	0.35
8027	NA	0.14	<0.01	0.01	0.12
8012	0.35	1.20	0.07	0.27	NA
8014	2.40	7.00	0.09	0.28	0.73
8018	0.76	2.20	0.09	0.28	0.98

## A7.2. Flambeau Mine, Wisconsin.

Table A7.2.1. Calculations showed that the flux of backfill pore water to the Flambeau River would be  $6.5 \times 10^5$  times lower than the ten year, seven day, low flow for the river (Flambeau Mining Company, 2001).

Parameter	Magnitude
Permeability (K)	0.028 ft/d
Hydraulic gradient (I) between west end of pit & river	0.03 ft/ft
Cross-sectional area of pit (A)	65000 ft <sup>2</sup>
pit width	650 ft
aquifer depth	100 ft
Pore water flux (Q = KIA)	54.5 cfd or 0.00063 cfs
Flambeau River average flow	1749 cfs
Flambeau River 10-yr, 7-d low flow	412 cfs

Table A7.2.2. Summary of surface water monitoring results between July 1991 and November 2000 (Flambeau Mining Company, 2001).

Parameter	SW-1 (upstream)	SW-2 (downstream)
pH	6.5 - 8.7	5.5 - 8.3
Alkalinity (mg/L)	nd	nd
Cu (µg/L)	0 - 0.012	0 - 0.012
SO <sub>4</sub> (mg/L)	5.2 - 8.6	5.2 - 8.4

Table A7.2.3. Ground water monitoring well specifications in and around the Flambeau Pit (Flambeau Mining Company., 2001).

Monitoring Well	Depth (feet)		Location	Material Contacted in Pit
	Original	Adjusted		
MW-1000 PR	57.37	57.8	Down gradient from backfill	--
MW-1004 S	38.02	27.0	Northwest edge of pit	--
MW-1004 P	89.34	75.9	Northwest edge of pit	--
MW-1002	17.71	--	North of pit	--
MW -1002 G	53.88	--	North of pit	--
MW-1005	20.16	--	East side of operation	--
MW-1005 S	52.43	--	East side of operation	--
MW-1005 P	93.46	--	East side of operation	--
MW-1010 P	115.4	--	Down gradient from backfill	--
MW-1013	24.2	--	Backfill pore water	Till
MW-1013 A	46.4	--	Backfill pore water	Low S rock
MW-1013 B	86.3	--	Backfill pore water	High S rock
MW-1013 C	201.5	--	Backfill pore water	High S rock
MW-1014	33.7	--	Backfill pore water	Sandstone
MW-1014 A	63.9	--	Backfill pore water	Low S rock
MW-1014 B	104.9	--	Backfill pore water	High S rock
MW-1014 C	156.6	--	Backfill pore water	High S rock

Table A7.2.4. Most ground water quality parameters are monitored on a quarterly basis. However, a more extensive analysis is conducted annually (Flambeau Mining Company, 2001).

Parameter	Monitoring Frequency
Field pH	Quarterly
Lab pH	Quarterly
TDS	Quarterly
Hardness	Quarterly
Alkalinity	Quarterly
As	Annually
Ba	Annually
Cd	Annually
Cr	Annually
Cu	Quarterly
Fe	Quarterly
Pb	Annually
Mn	Quarterly
Hg	Annually
Se	Annually
Ag	Annually
Zn	Annually
SO <sub>4</sub>	Quarterly
Color	Quarterly
Field Conductivity	Quarterly
Odor	Quarterly
Turbidity	Quarterly

Table 7.2.2.5. Partial summary statistics for ground water monitoring wells in the Flambeau Pit backfill (Flambeau Mining Company, 2001). Monitoring results from February 1999 to April 2000.

Copper (µg/L)

Monitoring Well	Minimum	Maximum	N	Material Contacted in Pit
MW-1013	ND	ND	ND	Till
MW-1013 A	ND	ND	ND	Low S rock
MW-1013 B	<4.7	36	6	High S rock
MW-1013 C	<4.7	100	6	High S rock
MW-1014	ND	ND	ND	Sandstone
MW-1014 A	<6.0	<6.0	1	Low S rock
MW-1014 B	420	810	6	High S rock
MW-1014 C	<0.47	16	6	High S rock

ND = No data, well not recovered sufficiently to collect ground water sample.

Alkalinity (mg/L)

Monitoring Well	Minimum	Maximum	N	Material Contacted in Pit
MW-1013	ND	ND	ND	Till
MW-1013 A	ND	ND	ND	Low S rock
MW-1013 B	520	630	6	High S rock
MW-1013 C	400	510	6	High S rock
MW-1014	ND	ND	ND	Sandstone
MW-1014 A	390	390	1	Low S rock
MW-1014 B	460	570	6	High S rock
MW-1014 C	300	380	6	High S rock

ND = No data, well not recovered sufficiently to collect ground water sample.

Sulfate (mg/L)

Monitoring Well	Minimum	Maximum	N	Material Contacted in Pit
MW-1013	ND	ND	ND	Till
MW-1013 A	ND	ND	ND	Low S rock
MW-1013 B	770	1900	6	High S rock
MW-1013 C	870	2000	6	High S rock
MW-1014	ND	ND	ND	Sandstone
MW-1014 A	970	970	1	Low S rock
MW-1014 B	580	1600	6	High S rock
MW-1014 C	370	700	6	High S rock

ND = No data, well not recovered sufficiently to collect ground water sample.

A7.3. Richmond Hill Mine, South Dakota.

A7.3.1.Excerpts from the Summary of the Mining Industry in South Dakota 1999. Prepared by the Minerals and Mining Program, South Dakota Department of Environment and Natural Resources, April 2000.

<http://www.state.sd.us/denr/DES/Mining/1999minesum.html>

**Acid Mine Drainage Mitigation Update at Richmond Hill Mine, pg 5:**

The pit impoundment, backfilled with acid generating rock and covered with a low permeability capping system, continued to exceed expectations in 1999. Four seasons of monitoring data show that only minimum amounts of oxygen and water are being detected in the impoundment. This indicates the cap is effective in limiting oxygen and water infiltration and is preventing acid generation. No signs of settling or slumping were found during several inspections of the pit impoundment by the department and LAC contractors. Only minor erosion was noted on a few portions of the impoundment. A dense, self-sustaining vegetative cover is becoming established on the pit impoundment and waste dump area.

The capped leach pads are also performing well. No signs of settling or slumping were found on the leach pads. Only minor erosion was noted in several areas. A good vegetative cover is becoming established. Monitoring data shows that the capping systems are effective in reducing water infiltration into the spent ore. Because of the low metal concentrations in the pad effluent, LAC believes that passive treatment may be feasible for long-term water treatment. A pilot plant has been constructed to test passive treatment, and test results are so far encouraging.

Ground and surface water quality around the mine site continues to improve. Biological assessments of Squaw Creek below the mine show that the stream is healthy. At the end of 1999, LAC discontinued water treatment for the winter since the volume of water requiring treatment had been reduced to almost zero. Only seasonal water treatment will be conducted, starting in the summer of 2000. Because of decreased water treatment and reclamation requirements at the site, LAC reduced its workforce to three employees at the end of the year.

**LAC Minerals (USA), Inc./Richmond Hill Mine, pg 20:**

LAC Minerals may make minor changes to the reclamation plan, including changes to roads and final pond configurations. Proposals on final water treatment plans, including passive treatment, may be submitted to the department. They plan to continue monitoring and seasonal water treatment in 2000.

Table A7.3.1. Typical drainage chemistry from the Spruce Gulch waste rock depository between 1992 and 1995 was acidic and contained elevated heavy metals (Durkin, 1995; Deux, 2000). Concentrations are in mg/L unless noted otherwise.

Parameter	Typical Concentration Range
pH	2.6 - 3.6 s.u.
TDS	800 - 5700
SO <sub>4</sub>	700 - 3400
Fe	50
Al	100
Cu	8
Zn	7
Flow	100 - 200 gpm

Table A7.3.2. Monitoring results of the Richmond Hill Engineered Pit Backfill Facility indicate that the reclamation plan has been successful to date (Duex, 2000).

Parameter Monitored	Result
Inflow	
Basin Lysimeter Readings	0.43 gpd in 1998
Barrel Lysimeter Readings	four of six have remained dry; cover permeability < $1 \times 10^{-7}$ cm/s
Pore Gases - Oxygen	0 - 4.8%
Temperature	< 20EC
Settlement	total = 0.12 ft or 0.1%
Ground Water	metals generally meet ground water standards
Surface Water (Squaw Creek)	no measurable affects
Revegetation	a good vegetative cover has been established

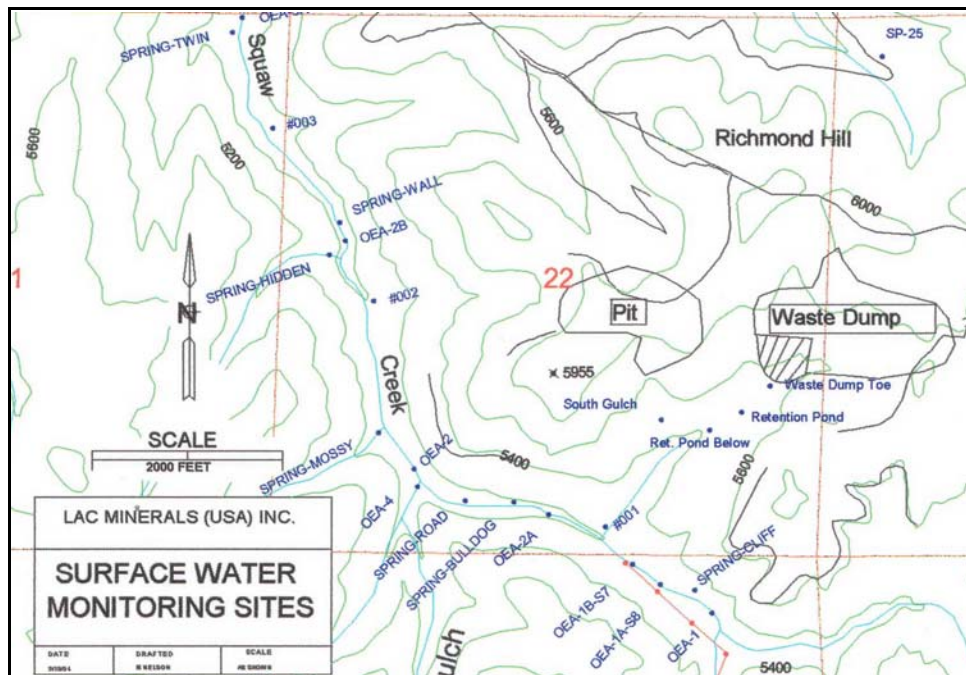


Figure A7.3.1. Partial map of surface water monitoring sites, Richmond Hill Mine, South Dakota (Nelson, 2001).

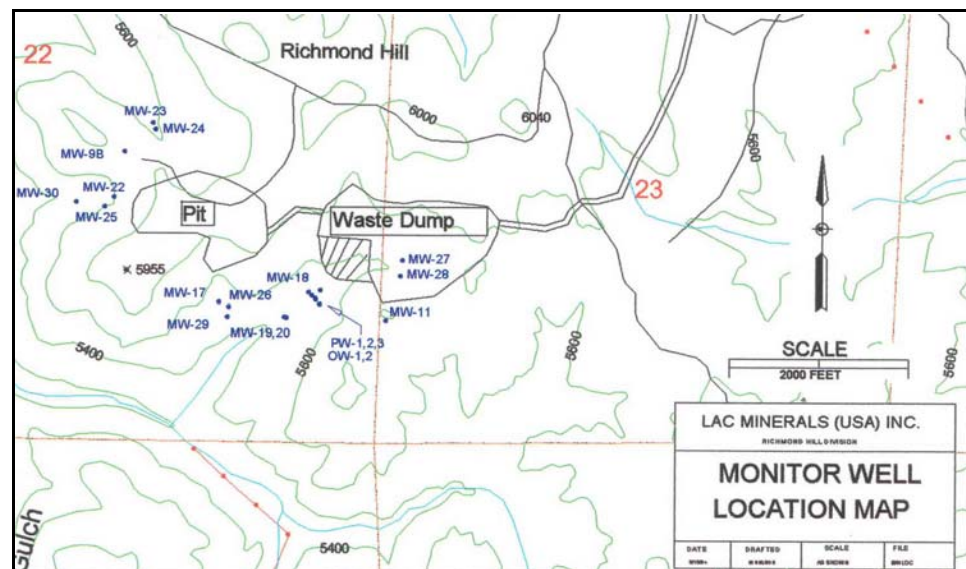


Figure A7.3.2. Partial monitoring well location map at the Richmond Hill Mine, South Dakota (Nelson, 2001).

# Richmond Hill Mine, South Dakota

Water Quality from the Waste Dump Toe.

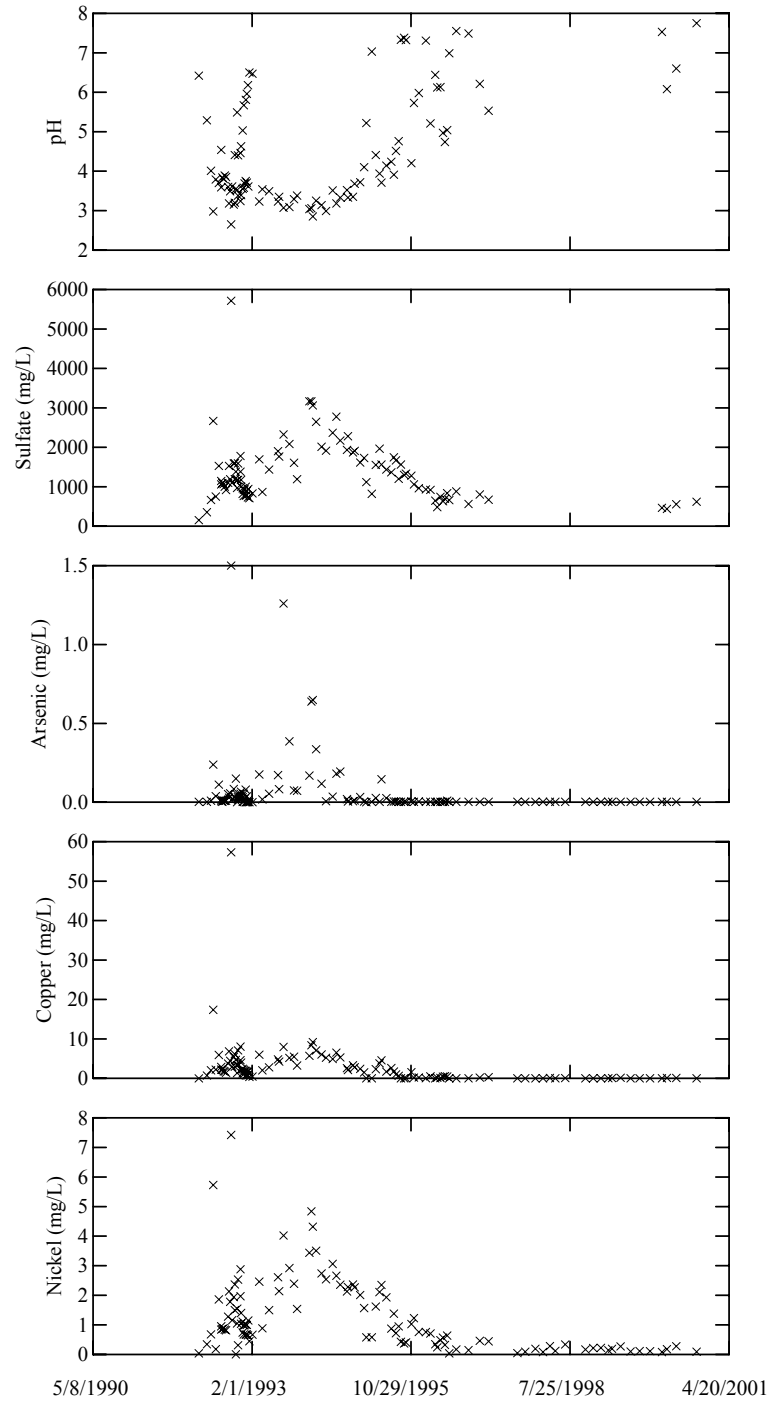


Figure A7.3.3.

Surface water quality monitoring results from the toe of the waste dump, Richmond Hill Mine, South Dakota.

# Richmond Hill Mine, South Dakota

Ground Water Quality Near the Backfilled Pit and Former Waste

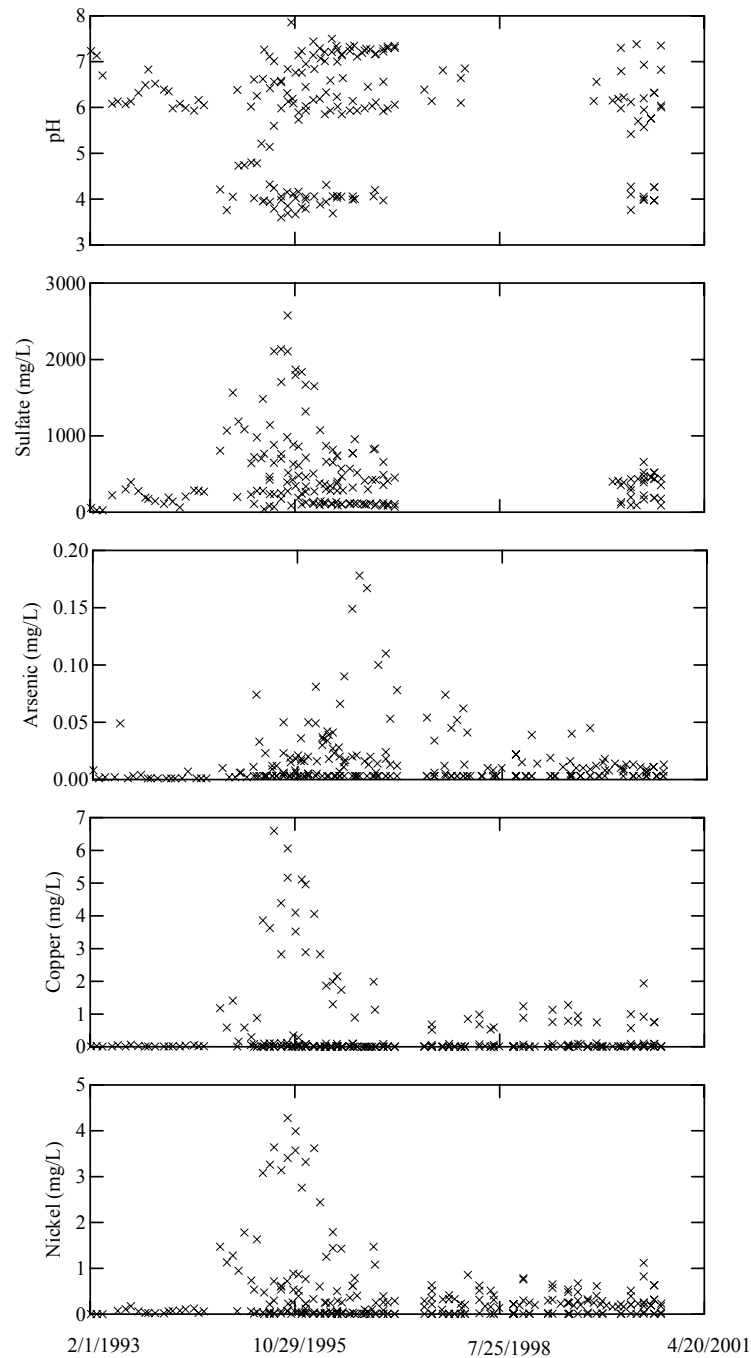


Figure A7.3.4 Ground water quality found in monitoring wells located around the backfilled pit and former waste dump at the Richmond Hill Mine, South Dakota.

A7.4. Additional sources of information regarding integrated environmental mine waste management system case studies.

Web Sites that Present Case Studies:

[www.infomine.com](http://www.infomine.com)  
<http://mend2000.nrcan.gc.ca>  
<http://www.inap.com.au/inap/homepage.nsf>  
[www.epa.gov/ORD/NRMRL/](http://www.epa.gov/ORD/NRMRL/)

**Summitville Mine, CO**

Strategies:

- Removal of waste rock piles,
- Backfilling and capping the open pit mine,
- Closure of the heap leach pad,
- Plugging two drainage adits,
- Active treatment of drainages (acid neutralization-precipitation)

References:

- Bigelow, R.C.; Plumlee, G.S.; Edelman, P. (1995) On-line update of US Geologic Survey OFR 95-23. [www.infomine.com](http://www.infomine.com).
- Campbell, A.; Gobla, M.J. (2000) Proc. 5<sup>th</sup> Internat. Conf. on Acid Rock Drainage (ICARD), V2, 1243-1250.
- Ketallapper, V.L.; Christiansen, J.W. (1999) Conf. Proc. Mining and the Environment II, Sudbury, Ontario, September 13-17, V1, 199-21

**Equity Silver Mine, British Columbia, Canada**

Strategies:

- Water cover over tailings pond,
- Covered waste rock dumps: compacted clay & till cover,
- Southern Tail Pit = backfill with reactive rock beneath ground water table, covered with inert waste rock and till and various clay covers,
- ARD collection and acid neutralization treatment

References:

- Aziz, M.L.; Ferguson, K.D. Equity Silver Mine - Case Study. <http://mend2000.nrcan.gc.ca>. Contact: Keith D. Ferguson, Manager, Environmental Affairs, Placer Dome North America Ltd. 604-661-1916 or Keith [Ferguson@placerdome.com](mailto:Ferguson@placerdome.com).
- Aziz, M.L.; Ferguson, K.D. (1997) Proc. 4<sup>th</sup> Internat. Conf. on Acid Rock Drainage (ICARD), Vancouver, B.C., May 31-June 6, V1, 181-195.
- Pedersen, T.F. et al. (1997) Proc. 4<sup>th</sup> Internat. Conf. on Acid Rock Drainage (ICARD), Vancouver, B.C., May 31-June 6, V3, 989-1005.
- Wilson, G.W. et al. (1997) Proc. 4<sup>th</sup> Internat. Conf. on Acid Rock Drainage (ICARD), Vancouver, B.C., May 31-June 6, V1, 197-210.