13

ENVIRONMENTAL MINE WASTE MANAGEMENT: SUMMARY OF PREVENTION, CONTROL, AND TREATMENT STRATEGIES FOR PROBLEMATIC DRAINAGES

Advances in Mine Waste Management Project Progress Report to the Minerals Coordinating Committee June 30, 2000



#6782

Minnesota Department of Natural Resources Division of Lands and Minerals

ENVIRONMENTAL MINE WASTE MANAGEMENT: SUMMARY OF PREVENTION, CONTROL, AND TREATMENT STRATEGIES FOR PROBLEMATIC DRAINAGES

Advances in Mine Waste Management Project Progress Report to the Minerals Coordinating Committee June 30, 2000

> Emmelyn Leopold Kim Lapakko

Minnesota Department of Natural Resources Division of Lands and Minerals 500 Lafayette Road Saint Paul, MN 55155

TABLE OF CONTENTS

1.	INTRODUCTION	
	1.1. Mining in Minnesota	1
	1.2. Mine Waste Drainage Mitigation in Minnesota	1
	1.2.1. MN DNR laboratory and small-scale field studies	2
	1.2.2. MN DNR operational-scale field studies	3
	1.3. Objectives and Scope	3
	5	
2.	BACKGROUND: Mine Waste Dissolution and Acid Mine Drainage	8
3.	PHYSICAL CONTAINMENT OF REACTIVE MINE WASTES	
٠	3.1. Dry Cover Systems	0
	3.1.1. Objectives of a Dry Cover System	0
	3.1.2. Assessment of Dry Cover System Designs	2
	3.1.3. Cost Analysis of Dry Cover Systems	4
	3.1.4. Potential for the Use of Dry Cover Systems for Reactive	
	Mine Wastes in Minnesota	4
	3.2. Water Cover Systems	6
	3.2.1. Principles of Water Cover Systems	6
	3.2.2. Assessment of Water Cover Systems	7
	3.2.3 Cost Analysis of Water Cover Systems	8
	3.2.4. Potential for the Use of Water Cover Systems for	-
	Reactive Mine Wastes in Minnesota	8
		•
4.	ENHANCED SURFACE RECLAMATION USING BACTERICIDES	
	4.1. Inhibition of Biological Sulfide Oxidation	1
	4.2. Assessment of Bactericides as Sulfide Oxidation Inhibitors	.1
	4.2.1. Preliminary Evaluation of Bactericides	1
	4.2.2. Parameters Used to Assess Bactericide Effectiveness	2
	4.3. Cost Analysis of Bactericides	2
	4.4. Potential Use of Bactericides to Inhibit Sulfide Oxidation in	
	Minnesota Mine Wastes 2	2
5.	ALKALINE AMENDMENTS FOR REACTIVE MINE WASTES	
	5.1. The Goals of Alkaline Amendments	4
	5.2. Methods Used to Assess the Effectiveness of Alkaline Amendments 2	4
	5.3. Cost Analysis of Adding Alkaline Materials to Reactive Wastes 2	:5
	5.4. Alkaline Amendments and Reactive Mine Wastes in Minnesota 2	:5
6	TREATMENT SYSTEMS	
0.	6.1. Passive Treatment Systems	26
	6.1.1. Constructed Wetlands	26

TABLE OF CONTENTS continued

		6.1.2. Anoxic Limestone Drains
	6.2.	Active Treatment Systems
		6.2.1. Overview of Active Treatment Systems
		6.2.2. Methods Used to Assess the Effectiveness of Active Treatment . 29
		6.2.3. Cost Analysis of Active Treatment Systems
		6.2.4. Potential Uses of Active Treatment Systems in Minnesota 29
7.	ADD	ITIONAL MITIGATIVE STRATEGIES UNDER CONSIDERATION
	7.1.	Chemical Stabilization by Microencapsulation of Reactive Wastes 30
		7.1.1. Assessment of Microencapsulation Effectiveness
		7.1.2. Cost Analysis of Microencapsulation Techniques
		7.1.3. Potential Use of Microencapsulation Technology to Address
		Mine Waste Issues in Minnesota
	7.2.	Selective Handling of Reactive Mine Wastes
		7.2.1. Backfill and Paste Technology
		7.2.2. Removal of Metals Using In-situ Leaching
	7.3.	Innovative Treatment Systems
		7.3.1. Porous Reactive Walls
8.	FUTU	JRE WORK
9.	REFE	ERENCES

LIST OF FIGURES

Figure 1.	Schematic of a generalized dry cover system	43
Figure 2.	Dry cover designs generally fall into one of six categories	44

LIST OF TABLES

Table 1.	Summary of treatment methods for acidic drainage
Table 2.	Summary of methods to prevent or control the generation
	of acidic drainage
Table 3.	Summary of cover construction costs by type of cover

Table 4.	Oxygen solubility/concentration and diffusion coefficient
Table 5.	Factors that should be assessed prior to disposing of
	reactive waste rock beneath a water cover
Table 6.	Summary of results from laboratory studies designed to
	evaluate the effectiveness of bacterial inhibitors
Table 7.	Summary of results from field tests on the effect of bacterial inhibitors . 50
Table 8.	Summary of the advantages and disadvantages of some
	common neutralization chemicals
Table 9.	Summary of the sludge dewatering methods
Table 10.	Average operational costs of selected North American
	mine drainage treatment operations
Table 11.	Summary of information from sulfide microencapsulation
Table 12.	Summary of drainage quality results from microencapsulation 54

APPENDICES

- 1. Outlined List of Mitigation References, April 2000, In progress
- 2. Physical Containment of Reactive Mine Wastes-Dry Cover Systems
- 3. Physical Containment of Reactive Mine Wastes-Underwater Disposal
- 4. Enhanced Surface Reclamation of Reactive Mine Wastes-Bactericides
- 5. Alkaline Amendments
- 6. Treatment Systems for Problematic Drainage
- 7. Additional Mitigative Strategies under Consideration

ACKNOWLEDGMENTS

Funding for this project was provided by the Minerals Coordinating Committee from the Minerals Diversification Program of the Minnesota Legislature. Paul Eger is acknowledged for his contributions to the section on passive treatment of mine waste drainage.

EXECUTIVE SUMMARY

Considerable mineral potential for base and precious metals is associated with Minnesota's Precambrian rocks. If mineral development occurs, the potential for generation of acidic mine waste drainage is the primary water quality concern. Mine waste management strategies directed at mitigating problematic drainages include prevention, control, and treatment. Six current methods of environmental mine waste management that show potential for application in Minnesota were evaluated in terms of 1) a brief description of the strategy; 2) methods used to assess the strategy performance both before and after application; 3) generalized cost analysis; and 4) applicability of the strategy to environmental mine waste management issues in Minnesota.

This progress report does not represent an comprehensive literature review of environmental mine waste management, nor does it address the entire range of possible preventative technologies currently in use. Future work will focus on site-specific conditions and requirements for successful implementation of these mitigative strategies, identification and evaluation of additional mitigative strategies that may be suitable for environmental mine waste management in Minnesota, and communication with national and international organizations conducting similar research (e.g. ADTI, MEND, INAP etc.).

1. INTRODUCTION

1.1. 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 over 26,000 acres of state land (MN DNR, 1998).

Considerable mineral potential for base and precious metals is associated with Minnesota's Precambrian rocks, specifically its 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 such that the quality of waters of the state is not adversely impacted. Mine waste management strategies directed at this objective include prevention, control, and treatment of problematic drainage.

1.2. 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 and Lapakko,

1

1997; Berndt et al., 1998, 1999; Lapakko and Jakel, 1999, 2000). 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.

1.2.1. MN DNR laboratory and small-scale field studies

A two-phase, 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 (Eger et al., 1984; Lapakko and Eger 1981, 1983, 1988; Lapakko et al., 1983).

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 ($0.053 < d \le 0.149$ 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 stockpile drainages from the Dunka mine site (Lapakko and Antonson, 1989b, 1990a, 1990b). Based on the results from the laboratory study, a field-scale limestone bed (1.4 m³ 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 barrier layers (composted yard waste, taconite tailings, compost and taconite tailings, and limestone), is under investigation in both the laboratory and field (MN DNR, unpublished data).

Although these laboratory and small-scale field test 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 were conducted on a single rock type. Consequently, conclusions on the potential application of these techniques to specific, large-scale environmental mine waste problems in Minnesota should be regarded cautiously.

1.2.2. 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., 1986) and 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 the 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, 1999). The stockpiles were covered with various types of capping 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, 1994; Eger and Wagner, 1995).

1.3. 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 drainage from sulfidic mine waste materials.

A literature search of the Reclamation Unit's literature database for the keyword "mitigation" yielded approximately 760 titles (MN DNR, Saint Paul, MN). An additional 366 titles were found in proceedings of mine waste management conferences that took place between 1994 and 2000 and other, miscellaneous sources that have not yet been entered into the database. Approximately 1126 titles were arranged in an outline format, grouped according to the mitigation strategy addressed (Appendix 1). It is important to note that the list of titles in Appendix 1 is largely comprised of references that were on hand in the MN DNR office in Saint Paul, MN. It does not represent an exhaustive literature review of mine waste mitigation strategies.

Due to the large number of titles 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) 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.

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 outlines 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 ADTI-Avoidance and Remediation Working Group (Skousen et al., 1998). However, this document was considered to be 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 Acid Drainage Technology Initiative (ADTI), a coalition of government agencies, industry, academia, and consultants, has been investigating environmental mine waste management issues since 1996. 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 (ADTI, 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 revegetation, including bactericide technology, addition of alkaline materials to mine wastes, special handling techniques for mine wastes and drainage, and remining.

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 1989. MEND has identified water covers as the "best prevention technology" against the generation of acidic drainage from unoxidized sulfidic wastes (Feasby et al., 1997). 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 sulfate reduction and the biosulfide process were mentioned as promising (Feasby et al., 1997).

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 identified as requiring additional investigation.

Several other Australian organizations exist that are concerned with environmental mine waste management. The Australian Nuclear Science and Technology Organisation's (ANSTO) Environmental Division conducts project-oriented, applied research in many areas of environmental impacts, 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. Their main area of focus is mineral processing. However, a few of these projects involve environmental mine waste management (AMIRA, 2000; Bennett, 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).

Sweden

A plan for a similar program in Sweden, the Mitigation of the Environmental Impact from Mining Waste (MiMi), was reported in December 1998 (MISTRA, 1998). The MiMi program goals include predicting which existing mine waste deposits can be reclaimed using simple, cost-efficient methods and developing strategies that will prevent future environmental problems associated with reactive mine wastes. To this end, five research projects were to be 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, porous reactive walls, and co-deposition of tailings and waste rock (MISTRA, 1998).

International

In October 1998, the International Network for Acid Prevention (INAP) was officially launched (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 (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.

Summary of Objectives and Scope

Thirty six different types of environmental mine waste management strategies were identified in aforementioned compilations (Tables 1 and 2). Six of these strategies were referred to in more than of the compilations. These strategies were:

- physical containment of reactive mine wastes using dry or water cover systems,
- enhanced surface reclamation using bactericides to inhibit biological sulfide oxidation in mine wastes,
- alkaline (e.g. limestone) amendments for reactive mine wastes,
- passive treatment using constructed wetlands and anoxic limestone drains, and
- active treatment (i.e. neutralization) of acidic mine drainage.

This document will review the much of the current state-of-technology of these strategies. Selected literature will be reviewed in terms of 1) a brief description of the mitigation strategy; 2) methods used to assess the strategy performance both before and after application; 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 annotated bibliographies (Appendices 2-7). It is important to note that this review does not represent an comprehensive literature review on the control, prevention, or treatment of acid mine drainage, nor does it address the entire range of possible preventative technologies currently in use.

2. BACKGROUND: 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 (after Stumm and Morgan, 1981; Nelson, 1978).

$$FeS_2 + (15/4)O_2 + (7/2)H_2O = Fe(OH)_3 + 2SO_4^2 + 4H^+$$
 (1)

$$FeS + (9/4)O_2 + (5/2)H_2O = Fe(OH)_3 + SO_4^{2-} + 2H^+$$
(2)

The rate of oxidation and attendant acid production is dependent on solid-phase compositional variables. Oxidation rates vary among sulfide minerals, reportedly decreasing in the order marcasite > pyrrhotite > pyrite. 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 of framboidal pyrite is reported to be much more rapid than that of euhedral pyrite.

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 (Nordstrom, 1982; Singer and Stumm, 1970; Kleinmann et al., 1981).

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 will exit as acid mine drainage.

Hydrated iron sulfate minerals precipitate during the evaporation of acidic, iron- and sulfaterich 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 and example, equations 3, 4, and 5 summarize the step-wise dissolution of melanterite.

$$FeSO_4 \cdot 7H_2O = Fe^{2+} + SO_4^{2-} + 7H_2O$$
 (3)

$$Fe^{2+} + (1/4)O_2 + H^+ = Fe^{3+} + (\frac{1}{2})H_2O$$
 (4)

$$Fe^{3+} + 3H_2O = Fe(OH)_3 + 3H^+$$
 (5)

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.

$$FeSO_4 \cdot 7H_2O + (1/4)O_2 = Fe(OH)_3 + SO_4^{2-} + (9/2)H_2O + 2H^+$$
(6)

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

$$KFe_3(SO_4)_2(OH)_6 = K^+ + 3FeOOH + 2SO_4^{2-} + 3H^+$$
 (7)

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 3 or 4 after contact with the jarosites (White et al., 1999). 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.

3. PHYSICAL CONTAINMENT OF REACTIVE MINE WASTES

3.1. Dry Cover Systems

3.1.1. 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 systems. 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 1). 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, active barrier layer, and any protection layers incorporated into the design.

The simplest and least expensive dry cover system to install is a **basic soil cover** (Figure 2). 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 2). 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 2). 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 2). 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 2). 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 liners. 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) liners 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 membranes can be installed year round and is very simple. 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 2).

Self-sealing/self-healing impermeable 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. 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.

- 3.1.2. Assessment of Dry Cover System Designs
 - 3.1.2.1. Variables Affecting the Type of Dry Cover System Implemented

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, constructability, and overall effectiveness.

3.1.2.2. Predictive Modeling of Proposed 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.

3.1.2.3. 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.

3.1.2.4. 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.

3.1.3. 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 3). 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.

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

Dry cover systems appear to be a promising method for preventing acidic drainage from reactive mine wastes in Minnesota. Simple soil and synthetic cover materials have been 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 Dunka 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 and continued for approximately three growing seasons. For the six year period of record, runoff coefficients for the control piles ranged from 0.44 to 0.58, as compared to 0.41 for natural watersheds in the area. 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 of 0.30, a value which was 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 -2.5 inches mixed with 5% bentonite, and a 20 mil PVC membrane. Laboratory tests indicated that the permeability of these materials ranged from 1.55×10^{-6} cm/s (49 cm/yr) for the glacial till down to 4.12×10^{-9} cm/s (0.13 cm/yr) for glacial till mixed with 5% bentonite, meeting the set standard of 2×10^{-6} cm/s (63 cm/yr). These results were similar to Hydrologic Evaluation of Landfill Performance (HELP) model simulations that predicted permeabilities ranging from 2.1×10^{-6} cm/s (66 cm/yr) for -2.5 inches of glacial till down to 5.2×10^{-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 -2.5 inches, till screened to -0.5 inches, till screened to -2.5 inches mixed with 5% bentonite, and the PVC membrane, respectively.

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 (Appendix 2).

There are several potential drawbacks to capillary barrier cover systems. First, most operational demonstrations on 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 (i.e. neutralizing) minerals. It is important to note however, that the availability of taconite tailings or any other alternative material will be highly site specific.

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 (Appendix 2). 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 (Appendix 2). Among these are forestry wastes, peat, paper residue, and various municipal wastes. Paper recycling residues, and possibly municipal wastes, appear to be the most promising organic cover materials in northern Minnesota. This determination was made based on demonstrated results reported in the literature and local availability. Furthermore, the use of one waste material to contain another is attractive both aesthetically and economically.

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.

3.2. Water Cover Systems

3.2.1. 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 25° C, whereas oxygen concentrations in air are 285 mg/L (21%) at 25° C (Table 4). Furthermore, the oxygen diffusion coefficient is four orders of magnitude slower in water than in air (Table 4). 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).

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 be slowed 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.

There are three basic types of water cover systems: underwater disposal, elevated water tables and perched water tables. Perched water tables are also referred to as a dry cover system "capillary barriers." Since this topic was discussed in section 3.1.1., it will not be revisited here.

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.

- 3.2.2. Assessment of Water Cover Systems
 - 3.2.2.1. 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 5). Each of these variables is site-

specific. Consequently, a water cover system designed for one location will not be suitable for any other.

3.2.2.2. 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.

3.2.2.3. 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.

3.2.3. Cost Analysis of Water Cover Systems

Information for this section is, at this time, incomplete.

3.2.4. Potential for the Use of Water Cover Systems for Reactive Mine Wastes in Minnesota

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. Unmodified subaqueous disposal tests were initially conducted in flasks using Virginia Formation hornfels rock containing 14 % pyrrhotite and in columns using sulfidic Duluth Complex rock. 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 possible produce acidic waters.

Modified subaqueous disposal experiments with barrier layers were conducted in flasks, beakers, and columns (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 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 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 = 12 in.), fitted with a 30 mil PVC liner. The tanks were filled to a depth of 48 inches with -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.

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*, *Potamegeton 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). 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.

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 widespread 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.

4. ENHANCED SURFACE RECLAMATION USING BACTERICIDES

4.1. Inhibition of Biological Sulfide Oxidation

Bacteria, specifically *Thiobacillus ferrooxidans*, have been shown to catalyze the oxidation of ferrous iron in reactive mine waste effluents below a pH of approximately 4.5. Acid is produced as a result of bacterial oxidation and results in highly acidic drainage. **Surfactants** have been shown to interrupt the activity of *T. ferrooxidans*, reducing the rate of oxidation, and therefore, drainage acidification.

Surfactants can be obtained as a powder or in a control 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 reapplication. Reapplication rates are estimated at three to six months.

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 reapplication.

4.2. Assessment of Bactericides as Sulfide Oxidation Inhibitors

4.2.1. 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).

4.2.2. 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 and 7). Unfortunately, these data did not indicate whether or not the drainage met the environmental water quality standard (pH >6). Furthermore, the fact that acidity was present indicated 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 6 and 7).

4.3. 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.

4.4. 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.

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 longenough for vegetation to become established on a reclaimed surface. In these cases, revegetation 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 revegetated surface, can be placed over the mine waste.

There are two major drawbacks to the use of bactericides. First, as described in section 4.2.2, surfactants cannot be relied upon to maintain drainage pH above 6.0. This may be sufficient to maintain a healthy vegetative cover in some instances. However, it 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 recommend conducting biological toxicity tests on drainages containing these compounds prior to extensive use (Clark, 2000).

ALKALINE AMENDMENTS FOR REACTIVE MINE WASTES

5.1. The Goals of Alkaline Amendments

5.

Acid release from reactive mine waste may be decreased by the mixing of alkaline solids with waste rock. The alkaline solids neutralize acid produced by the oxidation of sulfide minerals. This neutralization has three secondary effects. First, the elevated pH yields and 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 concentrations is due to increased trace metal 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.

5.2. Methods Used to Assess the Effectiveness of Alkaline Amendments

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).

5.3. Cost Analysis of Adding Alkaline Materials to Reactive Wastes

Information for this section is, at this time, incomplete.

5.4. Alkaline Amendments and Reactive Mine Wastes in Minnesota

While limited success has 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. Morin and Hutt's (2000) 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.

6. TREATMENT SYSTEMS

6.1. Passive Treatment Systems

6.1.1. Constructed Wetlands

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). A more detailed summary of these topics will be included in subsequent reports.

Although the design of wetland treatment systems can vary substantially, there are two basic types; overland or surface flow wetlands, and vertical flow wetlands (Eger, 2000). Surface flow wetlands are effective when the mine drainage is net alkaline. Removal reactions occur primarily in the surface water or upper portion of the substrate. Iron removal occurs primarily through precipitation. Trace metals are removed by adsorption, chelation, or ion exchange processes associated with the organic component of the substrate. Vertical flow wetlands are effective when the mine drainage is acidic. The wetland is constructed with an organic substrate that is usually supplemented with limestone. Acid is neutralized both through sulfate reduction processes and limestone dissolution, while metals are removed through precipitation as hydroxides, carbonates, and sulfides, in addition to the processes operating in surface flow systems.

.6.1.1.1. Methods Used to Assess the Effectiveness of Constructed Wetland Treatment Systems

Wetland treatment systems are evaluated primarily by the water quality at the outflow of the wetland. The goal of these systems is to produce an effluent which meets water quality standards. In order to accomplish this goal, both surface and vertical flow wetlands must be designed properly. Important factors in the design are the pH and the total metal content of the drainage. These factors will determine the type of system needed (surface flow versus vertical flow) and the size (Eger, 2000).

6.1.1.2. Cost Analysis of Constructed Wetland Treatment Systems

Costs for these systems can vary depending on the system design and complexity (e.g. from shallow surface flow to more complicated lined vertical flow). At the Dunka Mine site, costs varied from \$16 - \$26 per square yard of wetland (Eger, 2000).

6.1.1.3. Constructed Wetlands Treatment of Problematic Drainages in Minnesota

Currently there are five wetland treatment systems in Minnesota, all at the Dunka Mine site (Eger, 2000). These are basically overland flow wetlands, although some subsurface flow occurs through limestone drains which were built as part of the berms within the wetlands.

A vertical flow sulfate reduction system has been proposed to treat the drainage from Soudan State Park (Eger, 2000). This system will probably be built in the spring of 2001. In the future, these systems could be used in Minnesota to treat residual seepage from reclaimed areas.

6.1.2. Anoxic Limestone Drains

Although limestone is an effective neutralizing material, the high concentrations of iron in most acidic mine drainage prevents the use of limestone in aerobic situations, due to the coating of the limestone with iron oxyhydroxides (Eger, 2000). In the absence of oxygen, iron remains as the soluble ferrous species. Drains are typically constructed in an area with seeps. A trench is constructed to intercept the contaminated ground water and filled with two to three inch limestone. A settling pond is constructed at the outlet of the drain to remove the iron as it precipitates. The goal of the drain is to provide sufficient alkalinity to the drainage so that when the water surfaces, and the iron is converted from ferrous to ferric iron and precipitates, there is sufficient alkalinity to maintain a neutral pH. This approach has been widely used in the coal fields of the eastern United States.

6.1.2.1. Methods Used to Assess the Effectiveness of Anoxic Limestone Drains

In order to be effective, limestone drains must produce net alkaline water. Laboratory tests can be done to estimate the effectiveness of the proposed drain (Eger, 2000). Additional details on these tests will be included in future progress reports.

6.1.2.2. Cost Analysis of Anoxic Limestone Drains

Information for this section is, at this time, incomplete.

6.1.2.3. Potential for the Use of Anoxic Limestone Drains in Minnesota

In order for anoxic limestone drains to effectively treat problematic drainages from mine wastes in Minnesota, the water must be kept or made anoxic. New stockpiles could be constructed with a drain at the base to provide mitigation of acidic drainage. This approach

could reduce treatment costs during operation and could serve as part of the final treatment system for a reclaimed stockpile.

6.2. Active Treatment Systems

6.2.1. Overview of Active 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. Although other forms of active treatment (e.g. ion exchange, reverse osmosis, etc.) have been studied, this presentation will focus on neutralization methods. 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 8) for use in active treatment systems. Generally speaking, 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 through the addition of oxidants such as ozone, hydrogen peroxide, and potassium permanganate.

Neutralization treatment methods produce large volumes of sludge that must be handled and disposed if 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. High density sludges (up to 50 % solids by weight) can be produced by recycling the sludge into the neutralization reactor. Sludge handling and dewatering tend to be the most costly aspects of neutralization treatment systems. Settling units vary from large earthen basins to mechanical clarifiers to filtration units (Table 9). Engineered basins offer a low cost alternative. However, mechanical clarifiers and filtration units offer more control over the final sludge characteristics.

6.2.2. Methods Used to Assess the Effectiveness of Active Treatment

The goal of an active treatment system is to increase drainage pH and decrease metal concentrations to meet local and national regulatory discharge standards. 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.

6.2.3. Cost Analysis of Active Treatment Systems

Operational costs of active treatment systems vary depending on the choice of neutralization chemicals, sludge handling and disposal, and site specific considerations such as transportation costs. Based on information collected from several North American treatment plants (Appendix 6), operational costs for mine drainage treatment are on the order of several dollars per thousand gallons treated (Table 10). 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. Based information provided for a single operation, the Argo Tunnel, CO, these costs are estimated at \$5.8 million dollars.

6.2.4. Potential Uses of Active Treatment Systems in Minnesota

Information for this section is, at this time, incomplete.

7. ADDITIONAL MITIGATIVE STRATEGIES UNDER CONSIDERATION

7.1. Chemical Stabilization by Microencapsulation of Reactive Wastes

Reactive mine wastes can be isolated from oxidizing agents (i.e. O_2 , Fe^{3+}) 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:

$$FeS_2 + 15/2H_2O_2 \rightarrow Fe^{3+} + 2SO_4^{2-} + 7H_2O + H^+$$
 (8)

$$FeS_2 + 15/4Ca(OCl)_2 + 1/2H_2O \rightarrow Fe^{3+} + 2SO_4^{2-} + 15/4Ca^{2+} + 15/2Cl^- + H^+$$
 (9)

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:

$$Fe^{3+} + KH_2PO_4 \rightarrow FePO_4 + K^+ + 2H^+$$

$$\tag{10}$$

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, however, some examples include epoxy resins, dust suppressants, and latex coatings.

7.1.1. Assessment of Microencapsulation Effectiveness

Roughly half of the studies cited here dealt with the feasibility of chemically producing coatings on reactive mine wastes and tailings (Table 11). 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.

Ten laboratory and field studies addressed issues of drainage quality from treated samples of waste rock and tailings (Table 12). Six 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 (Fytas and Evangelou, 1998; Vandiviere and Evangelou, 1998; Fytas et al., 1999) and field (Chatham and Svee, 1996; Conca et al., 1999; Jensen et al., 1999; Rybock et al., 1999; US EPA, 1999; Williams et al., 1999; KEECO) 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).

7.1.2. Cost Analysis of Microencapsulation Techniques

Information for this section is, at this time, incomplete.

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

Information for this section is, at this time, incomplete.

- 7.2. Selective Handling of Reactive Mine Wastes
 - 7.2.1. Backfill and Paste Technology

Backfilling underground mines with tailings has gained favor in the last several years. One benefit of backfilling is the increased stability of underground mine workings. Also, backfilling provides a means of disposing of acid generating mine wastes in an environment isolated from the atmosphere. Flooding of backfilled mine wastes further decreases exposure to oxygen and reduces the possibility of acid production.

Backfill materials can be introduced into a mine void using two general methods. 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).

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 um) 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).

Pastes can be transported through pipelines in a plug flow. The fine material is forced to the outside, acting as a lubricating layer and carrying the coarser particles in the center. These characteristics are essentially identical to those of cement prior to hydration (Cincilla et al., 1997).

7.2.2. Removal of Metals Using In-situ Leaching

Information for this section is, at this time, incomplete.

7.3. Innovative Treatment Systems

7.3.1. Porous Reactive Walls

Information for this section is, at this time, incomplete.

8. FUTURE WORK

The goal of this document was to evaluate current methods and identify additional strategies for environmental mine waste management that show potential for application in Minnesota. As noted in several locations throughout the document, this process is not yet complete. Additional information will be gathered and compiled during the upcoming year in order to accomplish this goal. Specific objectives for future work include:

- Continue to compile information on the application of the mitigative strategies addressed in this document paying special attention to site-specific conditions and requirements;
- Expand the present analysis to include additional mitigative strategies including: sulfate reduction (e.g. "bioreactors"), passive treatment using porous reactive walls, active treatment methods such as reverse osmosis, sulfide precipitation, and ion exchange, removal of metals using in-situ leach mining, and backfilling mine wastes using paste technology;
- Conduct additional literature searches in order to expand the MN DNR Reclamation Unit's literature database on mitigative strategies;
- Monitor the progress of national and international research-oriented efforts (e.g. ADTI, MEND, INAP, etc.) to address environmental waste management issues;
- Review and summarize capital, construction, and operational costs of these strategies; and
- Evaluate the potential application of each mitigative strategy in terms of environmentally sound mine waste management in Minnesota.

9. **REFERENCES**

ACMER, 1998. Australian Centre for Mining Environmental Research. URL: http://acmer.com.au.

Adams, R.L.; Ninesteel, J.J.; Rauch, H.W., 1994. Laboratory testing of coatings for prevention of acid drainage in underground coal mines. In: Internat. Land Reclamation and Mine Drainage Conf. and 3rd Internat. Conf. on the Abatement of Acidic Drainage. April 24-29, Pittsburgh, PA. pg 218-225.

AMIRA, 2000. Australian Mineral Industries Research Association LTD. URL: http://www.amira.com.au.

ANSTO, 1999. Australian Nuclear Science and Technology Organisation. URL: http://www.ansto.gov.au/ansto/environment1/abt.html.

Aziz, M.L.; Ferguson, K.D., 1997. Equity silver mine- integrated case study. In: 4th Internat. Conf. on Acid Rock Drainage. May 31-June 6, Vancouver, B.C., Canada. Vol. 1, pg 181.

Benedetti, B.A.; Rastogi, V.; Sobek, A.A. 1990. Minimizing water treatment costs at active operations. Nat. Symp. on Mining, Knoxville, TN.

Bennett, J.W. 2000. Principal Research Scientist. ANSTO, Environmental Division, Managing Mine Wastes Project. personal communication.

Berndt, M.E.; Lapakko, K.A., 1997. Chemistry of water associated with taconite processing and tailings reactions in the Mesabi Iron Range district, NE Minnesota. MN Dept. Nat. Resour., Division of Minerals, Saint Paul, MN. 69 p. plus appendices.

Berndt, M.E.; Lapakko, K.A.; Jakel, M.E., 1998. In-pit disposal of taconite tailings: geochemistry. Progress report. MN Dept. Nat. Resour., Division of Minerals, Saint Paul, MN. 70 p.

Berndt, M.E.; Lapakko, K.A.; Jakel, M.E., 1999. In-pit disposal of taconite tailings: geochemistry. Final report. MN Dept. Nat. Resour., Division of Lands and Minerals, Saint Paul, MN. 77 p. plus appendices.

Brown, R. 1989. Water management at Brenda Mines. In: Proc. of the 13th Ann. British Columbia Mine Reclamation Symp. June 7-9,1989, Vernon, British Columbia. pg 8-17.

Burnett, J.M.; Burnett, M.; Ziemkiewicz, P.; Black, D.C., 1995. Pneumatic backfilling of coal combustion residues in underground mines. In: Proc. of a Conf. on Mining and the Environment, May 28-June 1, Sudbury, Ontario. pg 1211-1220.

Cincilla, W.A.; Landriault, D.A. Verburg, R., 1997. Application of paste technology to surface disposal of mineral wastes. In: Tailings and Mine Waste '99. A.A. Balkema, Rotterdam, Netherlands. 343-356.

Clark, R. 2000. Operations/Planning/Major Facilities, North District, Minnesota Pollution Control Agency. personal communication.

Conca, J.; Wright, J.; Mockler, T.J., 1999. Phosphate-induced metal stabilization for the remediation of acid mine drainage and heavy-metal contamination. In: Proc. of a Conf. on Mining and the Environment II, September 13-17, Sudbury, Ontario. Vol. 3, pg 1143.

CSIRO, 2000. Commonwealth Scientific and Industrial Research Organisation. URL: http://www.csiro.au.

Dave, N.K., 1992. Water cover on acid generating mine/mill wastes: A technical review. CANMET, Project # 30.14.99, 118 p.

Davis, A.; Eary, L.E.; Helgen S., 1999. Assessing the efficacy of lime amendment to geochemically stabilize mine tailings. Environ. Sci. and Tech., 33, 2626-2632.

Day, 1994. Evaluation of acid generating rock and acid consuming rock mixing to prevent acid rock drainage. In: Internat. Land Reclamation and Mine Drainage Conf. and 3rd Internat. Conf. on the Abatement of Acidic Drainage. April 24-29, Pittsburgh, PA. pg. 77.

Delaney, T.; Hockley, D.; Sollner, D., 1997. Applications of methods for delaying the onset of acidic drainage. In: 4th Internat. Conf. on Acid Rock Drainage. May 31-June 6, Vancouver, B.C., Canada. Vol. 2, pg 795.

Dobrokhotov, G. R., Maiorova, E. V. 1962. Kinetics of autoclave leaching of chalcopyrite. Zhurnal Prikladnoi Khemii, 35 (8): 702-1709.

Eger, P., 1992. The use of sulfate reduction to remove metals from acid mine drainage. In: Proc. 9th Nat. Meeting ASSMR, June 14-18, Duluth, MN. 563-575.

Eger, P., 1994. The use of sulfate reduction to remove metals from acid mine drainage. In: Internat. Land Reclamation and Mine Drainage Conf. and 3rd Internat. Conf. on the Abatement of Acidic Drainage. April 24-29, Pittsburgh, PA. pg. 412. Eger, P., 2000. Principal Engineer. Minnesota Department of Natural Resources, Division of Lands and Minerals. personal communication.

Eger, P; Antonson, D.; Folman, J.; Wagner, J. Melchert, G., 2000a. Creating wetlands on acid-generating tailings; an alternative to encapsulation. MN Dept. Nat. Resour., Div. of Lands and Minerals, St. Paul, MN. 33 p. plus appendices.

Eger, P; Johnson, A.; Melchert, G.; Antonson, D.; Dewar, S., 2000b. The use of dredge material as an organic substrate to create wetlands in taconite tailings basins. MN Dept. Nat. Resour., Div. of Lands and Minerals, St. Paul, MN. 54 p. plus appendices.

Eger, A.P.; Lapakko, K.A. 1985. Heavy metals study progress report on the field leaching and reclamation program: 1977-1983. MN Dept. Nat. Resour., Div. of Minerals, St. Paul, MN. 53 p. plus appendices.

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 Conf., April 19-21, Pittsburgh, PA, Vol. 1, pg 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 12-17, Chattanooga, TN. pg 780-787.

Eger, P.; Lapakko, K.; Chamberlain, P., 1984. Mixed disposal of waste rock and tailings to reduce trace metal release from waste rock. In: Proc. 1984 Symp. on Surface Mining Hydrology, Sedimentology and Reclamation, Lexington, KY. pg 49-56.

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 Internat. Peat Congress, Aug. 17-23, Duluth, MN, 542-547.

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. Soc. of Mining Engineers Ann. Meeting, March 1-3, Denver, CO. Preprint 99-38.

Eger, P.; Wagner, J., 1995. Sulfate reduction for the treatment of acid mine drainage: Long term solution or short term fix? In: Proc. of a Conf. on Mining and the Environment, May 28 - June 1, Sudbury, Ontario. pg 515-524.

Eger, P.; Wagner, J.; Kassa, Z.; Melchert, G., 1994. Metal removal in wetland treatment systems. In: Internat. Land Reclamation and Mine Drainage Conf. and 3rd Internat. Conf. on the Abatement of Acidic Drainage. April 24-29, Pittsburgh, PA.

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 Nat. Meeting, May 18-23, Knoxville, TN. 580-589.

Eger, P.; Wagner, J.; Melchert, G., 1997. The use of a peat/limestone system to treat acid rock drainage. In: 4th Internat. Conf. on Acid Rock Drainage. May 31-June 6, Vancouver, B.C., Canada. pg 1195-1209.

Evangelou, V.P., 1994. Potential microencapsulation of pyrite by artificial inducement of FePO₄ coatings. In: Internat. Land Reclamation and Mine Drainage Conf. and 3rd Internat. Conf. on the Abatement of Acidic Drainage. April 24-29, Pittsburgh, PA. Vol. 2, pg. 96-103.

Evangelou, V.P. 1996. Oxidation proof silicate surface coating on iron sulfides. U.S. Patent, Patent number 5494703.

Evans, D.R.; Rose, A.W., 1995. Experiments on alkaline addition to coal mine spoil. In: Proc. of a Conf. on Mining and the Environment, May 28-June 1, Sudbury, Ontario. pg 49-58.

Feasby, D.G.; Tremblay, G.A.; Weatherell, C.J., 1997. A decade of technology improvement to the challenge of acid drainage- A Canadian perspective. In: 4th Internat. Conf. on Acid Rock Drainage. May 31-June 6, Vancouver, B.C., Canada. Vol. 1, pg i.

Fytas, K.; Bousquet, P.; Evangelou, B., 1999. Application of silicate coatings on pyrite to prevent acid mine drainage. In: Proc. of a Conf. on Mining and the Environment II, September 13-17, Sudbury, Ontario. pg 1199-1207.

Fytas, K.; Evangelou, B., 1998. Phosphate coating on pyrite to prevent acid mine drainage. Internat. J. Surface Mining, Reclamation and Environment, 12, 101-104.

Garrels, R. M., Christ, C. L. 1965. Solutions, minerals, and equilibria. Freeman, Cooper and Company, San Francisco, CA.

Georgopoulou, Z.J.; Fytas, K.; Soto, H.; Evangelou, B., 1995. Pyrrhotite coating to prevent oxidation. In: Proc. of a Conf. on Mining and the Environment, May 28-June 1, Sudbury, Ontario. Vol. 1, pg 7.

Hewett, M. J. 1980. Hydrology of stockpiles of sulfide bearing gabbro in northeastern Minnesota. MN Dept. Nat. Resour., Division of Minerals, St. Paul, MN. 184 p.

INAP. 2000. International Network for Acid Prevention. URL: http://www.inap.com.au.

Jensen, R.; Sogue, A.; Schurman, S. 1999. Treating heavy metals in mine soil and waste using EnvirobondTM binder. In: Tailings and Mine Waste '99. A.A. Balkema, Rotterdam, Netherlands. pg 743-474.

KEECO (Klean Earth Environmental Company), Lynnwood, WA.

Kempton, J.H.; Swanson, D.; Bennett, M.; MacDonald, R.; Locke, W.; Gillespie, C.; Lechner, M.; Maley, P.,1997. Application of probabilistic acid/base accounting to minimize waste-rock handling in semi-arid climates. In: 4th Internat. Conf. on Acid Rock Drainage. May 31-June 6, Vancouver, B.C., Canada. Vol. 2, pg 871-888.

Kleinmann, R.L.P.; Erickson, P.M., 1988. Control of acid drainage from coal refuse using anionic surfactants. US BOM Report of Investigations 8847. 16p.

Lapakko, K.A.; Antonson, D.A., 1989a. Alkaline addition reactor experiments: April 1989 status report. MN Dept. Nat. Resour., Division of Minerals. Saint Paul, MN. 28 p.

Lapakko, K.A.; Antonson, D.A., 1989b. Limestone column treatment of stockpile drainage: April 1989 status report. MN Dept. Nat. Resour., Division of Minerals. Saint Paul. MN, 16 p.

Lapakko, K.A.; Antonson, D.A., 1989c. Seep 1 limestone bed: 1988 Progress report. MN Dept. Nat. Resour., Division of Minerals. Saint Paul, MN. 17 p.

Lapakko, K.A.; Antonson, D.A., 1990a. Laboratory experiments on the mixture of alkaline solids with sulfide bearing rock. MN Dept. Nat. Resour., Division of Minerals. Saint Paul, MN. 37 p. plus appendices.

Lapakko, K.A.; Antonson, D.A., 1990b. Laboratory treatment of three acidic stockpile drainages by limestone columns. MN Dept. Nat. Resour., Division of Minerals. Saint Paul, MN. 22 p. plus appendices.

Lapakko, K.A.; Antonson, D.A., 1990c. Pilot scale limestone bed treatment of the Seep 1 waste rock drainage. MN Dept. Nat. Resour., Division of Minerals. Saint Paul, MN. 24 p. plus appendices.

Lapakko, K.A.; Antonson, D.A., 1991. Mixing of limestone with acid producing rock. In: Proc. of the 2nd Internat. Conf. on the Abatement of Acidic Drainage, Sept. 16-18, 19991. Montreal, Quebec. p. 343-358.

Lapakko, K.; Antonson, D.; Jakel, E., 1998a. Mine waste characterization and drainage mitigation: Research summary 1998. MN Dept. Nat. Resour., Division of Minerals. Saint Paul, MN. 50 p.

Lapakko, K.A.; Antonson, D.A.; Wagner, J.R., 1997. Mixing of alkaline solids with finelycrushed acid-producing rock. In: 4th Internat. Conf. on Acid Rock Drainage. May 31-June 6, Vancouver, B.C., Canada. pg 1345-1360.

Lapakko, K.A.; Antonson, D.A.; Wagner, J.R., 2000. Mixing of rotary kiln fines with finegrained acid-producing rock. In: Proc. of the 5th Internat. Conf. on Acid Rock Drainage, May 21-24, Denver, CO.

Lapakko, K.; Eger, P., 1981. Trace metal removal from mining stockpile runoff using peat, wood chips, tailings, till, and zeolite. In: Proc. 1981 Symp. on Surface Mining Hydrology, Sedimentology and Reclamation. Lexington, KY. pg 105-116.

Lapakko, K.; Eger, P., 1983. Passive treatment of sulfide stockpile runoff. In: Proc. 1983 Nat. Conf. on Environmental Engineering, A. Medine and M. Anderson eds. ASCE, New York. 643-651.

Lapakko, K.; Eger, P., 1988. Trace metal removal from stockpile drainage by peat. In: Proc. 1988 Mine Drainage and Surface Mine Reclamation Conf., April 19-21, Pittsburgh, PA. Vol. 1, pg 291-300.

Lapakko, K.A.; Eger, A.P., Strudell, J.D., 1986a. Low-cost removal of trace metals from copper-nickel mine drainage. Final Report BuMines Contract J0205047. U.S.D.I. Bureau of Mines NTIS #PB 87-186136. 103p.

Lapakko, K.; Jakel, M.E., 1999. Summary of potential impacts of taconite tailings disposal in the Minorca Pit on water quality in the Missabe Mountain Pit. MN Dept. Nat. Resour., Division of Lands and Minerals. Saint Paul, MN. 29 p. plus appendices.

Lapakko, K.; Jakel, E., 2000. Detailed analysis of potential impacts of taconite tailings disposal in the Minorca Pit on water quality in the Missabe Mountain Pit. Final report. MN Dept. Nat. Resour., Division of Lands and Minerals. Saint Paul, MN. 83 p. plus appendices.

Lapakko, K.; Strudell, J.; Eger, P., 1983. Sulfide stockpile leachate flow through a tailing bed: Trace metal removal and flow modeling for a low maintenance treatment system. Proc. 1983 Symp. on Surface Mining Hydrology, Sedimentology and Reclamation. Lexington, KY. pg 321-327.

Lapakko, K.; Strudell, J.; Eger, P., 1986b. 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. 45p.

Lortie, L.; Gould, W.D.; Stichbury, M.; Blowes, D.W.; Thurel, A., 1999. Inhibitors for the prevention of acid mine drainage. In: Proc. of a Conf. on Mining and the Environment II, September 13-17, Sudbury, Ontario. Vol. 3, pg 1191.

Majima, H., Peters, E. 1966. Oxidation rates of sulfide minerals by aqueous oxygen at elevated temperatures. Trans. of Metall. Soc of A.I.M.E. 236, p. 1409-1413.

McKay, D. R., Halpern, J, 1958. A kinetic study of the oxidation of pyrite in aqueous suspension. Trans. of Metall. Soc. of A.I.M.E., June.

Mehling, P.E.; Day, S.J.; Sexsmith, K.S., 1997. Blending and layering waste rock to delay, mitigate or prevent acid generation: A case study review. In: 4th Internat. Conf. on Acid Rock Drainage. May 31-June 6, Vancouver, B.C., Canada. pg 951-969.

Miller, B., Hornaday, C. 1998. Geosynthetic clay liners for the mining industry. Tailings and Mine Waste '98. A.A. Balkema, Rotterdam, Netherlands. 968 p.

MN DNR, 1998. October 1997 (Q2 F1998) Report on state mineral leasing activities. MN Dept. Nat. Resour., Division of Minerals, Minerals Leasing and Mineral Rights Management Section, St. Paul, MN.

MN DNR, 2000. DNR Waters, Climate of Minnesota. URL: http://www.dnr.state.mn.us/waters/climate/index.html.

Minnesota Department of Revenue, 1997. Personal communication with Tom Schmucker, Administrative engineer, Eveleth, MN.

Minnesota Environmental Quality Board. 1979. Regional copper-nickel study, Executive summary, volume 1. MN State Planning Agency, St. Paul, MN 118 p.

MISTRA, 1998. Programme plan for the period 1999-2000. Mitigation of the environmental impact from mining waste. A MISTRA research programme. 50 p.

Morin, K.A.; Hutt, N.M., 2000. Discrete-zone blending of Net-acid-neutralizing and netacid-generating rock: Avoiding the argument over appropriate ratios. In: Proceedings of the 5th Intern. Conf. on Acid Rock Drainage, May 21-24, Denver, CO.

Moskalyk, R.R., 1995. Development of a protective cover for mine waste rock. In: Proc. of a Conf. on Mining and the Environment, May 28-June 1, Sudbury, Ontario. Vol. 1, pg 99.

Nelson, M. 1978. Kinetics and mechanisms of the oxidation of ferrous sulfide. Ph. D. Thesis, Stanford University, Palo Alto, CA.

PA DEP, 1998. Coal mine drainage prediction and pollution prevention in Pennsylvania. The Pennsylvania Dept. of Environmental Protection. Harrisburg, PA. 375p.

Parker, G.; Robertson, A., 1999. Acid drainage. Occasional Paper No. 11. Australian Minerals and Energy Environment Foundation. Melbourne, Victoria. 227 p.

Patterson, R.J., 1987. Environmental and reclamation measures at Equity Silver Mines Ltd. Proc. Acid Mine Drainage Seminar/Workshop, March 23-26, Halifax, Nova Scotia. pg 457.

Rastogi, V.; Sobek, A.A.; Benedetti, D.A., 1990. Prevention of water pollution problems in mining: The bactericide technology. Society for Mining, Metallurgy, and Exploration, Inc.

Roy, S.; Worral, F., 1999. Pyrite oxidation in coal-bearing strata - the use of P-blocking techniques. In: Proc. of a Conf. on Mining and the Environment II, September 13-17, Sudbury, Ontario, Canada. Vol. 1, pg 177.

Sato, M., 1960a. Oxidation of sulfide ore bodies, I., Geochemical environments in terms of Eh & pH: Econ. Geology 55: 928-961.

Sato, M., 1960b. Oxidation of sulfide ore bodies, 11. Oxidation mechanisms of sulfide minerals at 25'C. Econ. Geol. 55: 1202-1231.

Sato, M., Mooney, H. M. 1960. The electrochemical mechanism of sulfide self-potentials. Geophysics 25 (1): 226-249.

Skousen, J.G.; Ziemkiewicz, P.F., 1995. Acid mine drainage: Control & treatment. West Virginia University and the National Mine land Reclamation Center, Morgantown, WV. 255p.

Skousen, J.; Rose, A..; Geidel, G.; Forman, F., Evans, R.; Hellier, W. 1998. A Handbook of Technologies for Avoidance and Remediation of Acid Mine Drainage, Acid Drainage Technology Initiative (ADTI), Morgantown: National Mine Land Reclamation Center at West Virginia University.

Stewart, B.R.; Jackson, M.L.; Daniels, W.L., 1994. The effects of fly ash blending rates and ash properties on leachate quality from fly ash/coal refuse mixtures. In: Internat. Land Reclamation and Mine Drainage Conf. and 3rd Internat. Conf. on the Abatement of Acidic Drainage. April 24-29, Pittsburgh, PA. Vol. 3, pg. 17.

Stewart, D., vonMaubeuge, K. 1997. Cost effective and efficient solutions with geosynthetic clay liners for sealing applications in the mining industry. In: Tailings and Mine Waste '97. A.A. Balkema, Rotterdam, Netherlands. 788 p.

Stichbury, M.; Bechard, G.; Lortie, L.; Gould, W.D., 1995. Use of inhibitors to prevent acid Mine drainage. In: Proc. of a Conf. on Mining and the Environment, May 28-June 1, Sudbury, Ontario. Vol. 2, pg 613.

STS Consultants Ltd., 1993. Final closure plan for Dunka Mine. Report to LTV Steel Mining Company. Hoyt Lakes, MN. 218 p. plus appendices.

STS Consultants Ltd., 1994a. Stockpile 8018 capping, Dunka Mine NPDES Permit No. MN0042579. Report to LTV Steel Mining Company. Hoyt Lakes, MN. 6 p. plus appendices.

STS Consultants Ltd., 1994b. 051 treatment system, Dunka Mine NPDES Permit No. MN0042579. Report to LTV Steel Mining Company. Hoyt Lakes, MN. 14 p. plus appendices.

Stumm, W., Morgan, J. J. 1981. Aquatic Chemistry-An introduction emphasizing chemical equilibria in natural waters. John Wiley and Sons, Inc. New York. 780p.

US EPA. 1983. Design manual: Neutralization of acid mine drainage. U.S. Environmental Protection Agency, Industrial Environmental Research Laboratory. EPA-600/2-83-001. 231p.

US EPA. 1999. United States Environmental Protection Agency, Mine Waste Technology Program. URL: http://www.epa.gov/ ORD/NRMRL/std/mtb/activities.htm.

Vandiviere, M.M.; Evangelou, V.P. 1998. Comparative testing between conventional and microencapsulation approaches in controlling pyrite oxidation. J. Geochem. Exploration, 64, pg. 161-176.

Walker, J.S., 1993. State-of-the-art Techniques for Backfilling Abandoned Mine Voids. (Information Circular 9359) United States Department of the Interior, Bureau of Mines, Pittsburgh, PA. 17 p.

White, W.W., Lapakko, K.A.; Cox, R.L. Static-test methods most commonly used to predict acid-mine drainage: Practical guidelines for use and interpretation. In: Reviews in Economic Geology, Vol. 6A, The Environmental Geochemistry of Mineral Deposits: Processes, Techniques, and Health Issues, 325-338.

Zhang, Y.L.; Evangelou, V.P., 1998. Formation of ferric hydroxide-silica coatings on pyrite and its oxidation behavior. Soil Sci., 163, 1, 53-62.





Figure 1. Schematic of a generalized dry cover system design for acid producing mine waste material.

DRY COVER SYSTEM DESIGNS



Figure 2. Dry cover designs generally fall into one of six categories, each of which is designed to minimize water infiltration and/or oxygen diffusion into the mine waste.

	TREATMENT STRATEGY	I				ADTI			DENN	
APPROACH		MEND	AMEEF	EPA	MiMi	Metal Mines	Coal	NMLRC	DEP	Total
Active:	Aeration/Oxidation			Х		Х	Х	Х		4
	Biosulfide Process	x								1
	Electrodyalysis					х	Х			2
	Flocculants/Coagulants			Х		Х	Х	Х		4
	Ion Exchange (biofix beads, clay deriv, resins, zeolites)	x	x			х	х			4
	Neutralization	x	Х	х		х	Х	Х		6
	Phosphate Addition		Х							1
	Reverse Osmosis					Х	Х			2
	Sludge Recycle	x	Х			х	Х			4
	Sulfate Reduction Reactors									0
Semi-Passive:	Liquid Organic Amendments					х				1
	Aquafix					х		Х		2
	Neutromil					х				1
	Combination Systems					х				1
Passive:	Bioremediation (SO ₄ reduction, CN degradation, etc.)			x	X	х	х			4
	Constructed Wetlands	x	х	х .	Х	ż	х	х		7
	Natural Wetlands		х		Х	Х	Х	Х		5
	Porous Reactive Walls	х	х	Х	Х					4

· ·

•

. -

Table 1.Summary of treatment methods for acidic drainage.

.

			AMEEF	EPA		ADTI		W/W/11/	DCMM	
APPROACH	TREATMENT STRATEGY	MEND			MiMi	Metal Mines	Coal	NMLRC	DEP	Total
	Anoxic Limestone Drains	X	Х			Х	Х	Х		5
	Diversion Wells					Х	Х			2
	Limestone Ponds					х	Х	Х		3
	Limestone Sand Treatment					х	Х			2
	Open Limestone Channels					х	Х	Х		3
	Vertical Flow Systems					х	X	X		3

:

:

:

:

46

:

.

						ADTI		DENINI	Τ	
APPROACH	MITIGATION STRATEGY	MEND	AMEEF	EPA	MiMi	Metal Mines	Coal	NMLRC	DEP	Total
Amendments:	Alkaline Materials	X ¹	Х	Х		Х	Х	Х	Х	7
	Organics					х				1
	Phosphate	x		Х		х		Х		4
Covers:	Bacterial Biofilms		Х							1
	Dry	х	х	Х	Х	Х	Х	Х		7
	Water	х	х		Х	Х	Х	Х	х	.7
Mineral Surface Treatments	Biooxidation/Biological Passivation?					Х				1
	Desulfurization	х		Х						2
	Electrochemical Control		х							1
	Microencapsulation/Chemical Passivation?		х	Х		X	Х			4
Reclamation	Bactericides	х	х			Х	Х	Х	Х	6
Selective Handling	Backfilling (alkaline, paste, etc.)	х						х	х	3
	Co-disposal				Х	Х				2
	Physical Separation					Х				1
	Re-mining/processing			Х		Х		Х	x	4
Source Water Control	Freezing/Permafrost	х				х				2
	Grouting					х		X		2
	Mine Sealing					X				1
	Porous Surroundings	Х				•		Х	X	3
	Surface Water Diversion					х			х	2

Table 2.Summary of methods to prevent or control the generation of acidic drainage.

¹Particulary when wastes will be covered and/or placed back into a pit

. .

Table 3.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 ¹)
Single layer of clay or clayey till	\$9 - 62 k ²
GCL	\$17 k
Single layer of soil	\$ 15 - 25 k ²
PVC	\$36 - 60 k ³
Capillary barrier (non-reactive mine wastes)	\$59-70 k
Capillary barrier (general)	\$50 - 300 k

¹No attempt was made to adjust for inflation or fluctuating exchange rates.

²The higher end of this range includes compaction costs.

³Range varies depending on the required thickness.

Table 4.Oxygen solubility/concentration and diffusion coefficient in air and water at
25 °C (adapted from Dave, 1992).

Parameter	Air	Water	Air:Water
Concentration (mg/L)	285	8.6	33
Molecular Diffusion Coefficient, $D_0 (m^2/s)$	1.78 x 10 ⁻⁵	2 x 10 ⁻⁹	8900
Effective Diffusion Coefficient, D _e (m ² /s)	1.43 x 10 ⁻⁶	1.6 x 10 ⁻¹⁰	8900
Flux at the Interface (g m ² /s)	3.5 x 10 ⁻⁴	1.1 x 10 ⁻⁷	3100

	Factor	Tailings Basin	Open Pit	Underground Mine	Natural Lake
	Aquatic Resources	Х	Х	X	Х
Biological	Biota Inventory				Х
Economic	Cost Estimate	х	Х	Х	Х
	Dam Safety	X			
Engineering	Extreme Climate Events (drought, earthquake, flood)	х	Х	Х	х
	Structural Integrity	Х		Х	
	Ground Water Chemistry	Х	Х	Х.	X
	Pore Water Chemistry	Х	Х	Х	Х
	Surface Water Chemistry	Х	Х		Х
	Waste Chemistry	Х	х	Х	Х
l	Waste Mineralogy	Х	х	Х	Х
	Waste Particle Density	х		Х	Х
	Waste Particle Size	Х		Х	Х
	Waste Type (fresh or oxidized)	Х	х	х	х
	Climate	Х	Х		X
Hydrologic	Local Topography	Х	Х		х
	Water Balances	Х	X	Х	X
	Bathymetry		х		Х
	Discharge Rate	Х	Х	X	Х
	Minimum Water Depth	Х	Х		Х
Physical	Orientation	Х	х		х
	Storage Volume	Х	Х	Х	Х
	Surface Dimensions	Х	Х		Х
	Wave Action	X			X

Table 5.Factors that should be assessed prior to disposing of reactive waste rock
beneath a water cover.

.

•

Table 6.	Summary of	results from	laboratory	studies	designed	to eval	luate the
	effectiveness	of bacterial	inhibitors a	t reducin	g the ef	ffects of	sulfide
	oxidation.						

Deferre	Inhibitor Used	Maintained pH >6	Param	Inhibited		
Kelerence			Acidity	Sulfate	Metals	Growth
Sobek, 1987	surfactant	nd	70	nd	nd	nd
Patterson, 1987	surfactant	nd	nd	nd	Cu: 50	nd
Watzlaf, 1988	three surfactants	nd nd nd	50 50 50	nd nd nd	nd nd nd	yes yes no
Stichbury et al., 1995	two heterocyclic mercaptans	no yes	nd nd	neg. ~50	nd nd	nd nd

nd = not determined

.

neg. = negligible reduction

Table 7.Summary of results from field tests on the effect of bacterial inhibitors on
reactive mine waste drainage quality.

.

Pafaranaa	Inhibitor	Maintained pH >6	Param	on (%)	Inhibited Bacteria	
	Used		Acidity	Sulfate	Metals	Growth
Patterson, 1987 ¹ & Delaney et al., 1997	surfactant	no	none	none	none	no
Parisi et al., 1994	surfactant	no	92	70	nd	nd
Lortie et al., 1999	heterocyclic mercaptan	no	nd	52	Fe: 76 Co: -39 Cu: 56 Mn:-230 Ni: 8 Zn: 14	yes
Lortie et al., 1999	heterocyclic mercaptan	no	nd	42	Fe: 66 Co:-4000 Cu:-133 Mn: -38 Ni:-1550 Zn: -18	yes

nd = not determined

neg. = negligible reduction

Table 8.	Summary	of	the	advantages	and	disadvantages	of	some common
	neutralizati	ion (chemi	cals used fo	r activ	e treatment of a	cidic	drainages.

Neutralizatio n Chemical	Advantages	Disadvantages
Lime	 available in bulk low cost/gram acidity neutralized efficient neutralization simple handling and storage 	 quicklime must be slaked which increases capital costs and requires daily monitoring may absorb moisture during storage may contain "grit" sludges settle slowly
Limestone	 generally inexpensive higher density sludge than lime 	 slow reactivity adequate reactivity requires high . quality limestone (high Ca, -325 mesh, high specific surface area) difficult to achieve an adequate pH for iron and manganese removal
Caustic Soda	 highly reactive liquid, can be pumped or fed by gravity mixes easily with drainage produces effluent with low suspended solids, turbidity, and Fe content sludges have "acceptable" settling properties 	 expensive dangerous to handling properties 50% solution freezes at 54 °F
Soda Ash	 briquettes easy to handle dissolution rate ensures continual treatment treatment cells can be arranged in series or parallel to suit needs sludge settles and compacts comparable to lime 	 expensive limited availability only effective for low flow drainages briquettes may become coated with iron

.

•

Table 9.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

Table 10.	Average	operational	costs	of	selected	North	American	mine drainage
	treatment	operations.						

Operation	Process	Average Flow (gpm)	Estimated Annual Operation Costs (\$ 10 ⁶ /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 ¹	1.0	1.61
California Gulch Superfund Site, Yak Tunnel, CO	Caustic Soda	400 ²	nr	nr
Eagle Mine, CO	Lime	160	nr	nr
Summitville, CO	Hydrated Lime	1300 ¹	2.5	9.51
Equity Silver, British Columbia	Quicklime	nr	1.1	4.73

¹Assumes operational period of approximately nine months, excluding winter ²Plant operates three to four days a week

nr = not reported

•

Reference	Coating Type		Test Objective		
	Phosphate	Silicate	Other	Coating Feasibility	Drainage Quality
Roy and Worral, 1999	X			Х	
Conca et al., 1999	x				Х
Jensen et al., 1999	x				Х
« US EPA, 1999	x				Х
Fytas and Evangelou, 1998	x			X ·	X
Georgopoulou et al., 1995	x			х	
Evangelou, 1994	x			Х	
Williams et al., 1999	x	Х	Х		Х
Vandiviere & Evangelou, 1998	x	X			Х
Fytas et al., 1999		Х		X	Х
Rybock et al., 1999		Х			Х
Zhang and Evangelou, 1998		Х		x	
Chatham and Svee, 1996		х			Х
KEECO		х		X	Х
Chen et al., 1999			X	x	
Moskalyk, 1995			Х	x	
~ Maki et al., 1995			Х		Х
Adams et al., 1994		λ.	X	x	

Table 11.Summary of information from sulfide microencapsulation studies reported in
this document.

Defense	Coating Trees	Drainage Quality Results			
Kelefence		pH > 6	Reduced Metal Conc.		
Laboratory Studies:					
Fytas et al., 1999	Silicate	yes	Fe, SO ₄		
Fytas and Evangelou, 1998	Phosphate	yes	Fe (98%), SO ₄ (75%)		
Vandiviere & Evangelou, 1998	Phosphate & Silicate	yes ¹	nd		
Field Studies:					
Conca et al., 1999	Apatite II (P)	nd	Pb, Zn, Cd,		
Jensen et al., 1999	Envirobond [™] (P)	nd	As, Pb, Zn		
US EPA, 1999	Envirobond [™] (P)	nd	Pb		
Williams et al., 1999	Phosphate & Silicate	yes	Cd, Pb, Zn		
Rybock et al., 1999	KB-1 treated drainage (Si)	yes	As, Cd, Cu, Fe, Pb, Zn		
Chatham and Svee, 1996	KB-SEA (Si)	yes	Al, As, Be, Bi, Co, Cr, Cu, Fe, Mn, Ni, Pb, Sr, Zn		
KEECO	KB-SEA (Si)	nd	Cd, Cu, Pb, Zn, As		

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

nd = not determined

¹mixed with CaCO₃

.