MEND MANUAL

VOLUME 4 – PREVENTION AND CONTROL

MEND 5.4.2d

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AUTHORS OF THE MEND MANUAL

This Manual was compiled on behalf, and under the direction of MEND and MEND 2000, by URS Norecol Dames & Moore, in association with SENES Consultants Limited, SRK Consulting, BC Research Inc., EVS Environment Consultants and O'Kane Consultants Inc. The different volumes and sections of the Manual were authored as follows:

Volume 1:	
	SENES Consultants Limited
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Section 2.1 to 2.3	SENES Consultants Limited
Section 2.4	URS Norecol Dames & Moore
	EVS Environment Consultants
Section 2.5	SENES Consultants Limited
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Volume 3:	
Section 3.1 to 3.3	URS Norecol Dames & Moore
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Section 4.1	SENES Consultants Limited
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Section 4.4	O'Kane Consultants Inc.
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Section 5.1	SENES Consultants Limited
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Section 5.3	URS Norecol Dames & Moore
Volume 6:	
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VOLUME 4 – PREVENTION AND CONTROL

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While considerable progress has been made in tackling the problems of acidic drainage, major challenges remain. Comments on this document and other aspects of acidic drainage should be sent to:

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DISCLAIMER

The primary purpose in producing this manual is to provide a succinct summary of the extensive work completed by MEND and MEND 2000 on the processes of acid generation from sulphur-bearing minerals and sulphide wastes in a manageable single reference document. A secondary objective is to provide additional recommendations on the application of currently available technologies. The result is a detailed reference on sampling and analyses, prediction, prevention, control, treatment and monitoring of acidic drainage. The information provided is based on the opinions of the authors of the particular sections and should not be construed as endorsement in whole or in part by the various reviewers or by the partners in MEND (the Government of Canada, Provincial Governments, the Mining Association of Canada, contributing mining companies and participating non-governmental organizations).

The user of this guide should assume full responsibility for the design of facilities and for any action taken as a result of the information contained in this guide. The authors and Natural Resources Canada (through the Mine Environment Neutral Drainage (MEND) and MEND 2000 programs) make no warranty of any kind with respect to the content and accept no liability, either incidental, consequential, financial or otherwise arising from the use of this publication.

PREFACE

Acidic drainage[1] has been identified as the largest environmental liability facing the Canadian mining industry, and to a lesser extent, the public through abandoned mines. This liability is estimated to be between $2 billion and $5 billion Canadian, depending on the sophistication of treatment and control technology used. There are numerous examples throughout the world where elevated concentrations of metals in mine drainage have adverse effects on aquatic resources and prevent the reclamation of mined land. Metal leaching problems can occur over an entire range of pH conditions, but are commonly associated with acidic drainage. In North America acidic drainage has resulted in significant ecological damage and multimillion-dollar cleanup costs for industry and governments.

The Canadian Mine Environment Neutral Drainage (MEND) Program was formed in 1989, to develop scientifically-based technologies to reduce or eliminate the liability associated with acidic drainage. This nine-year volunteer program established Canada as the recognized leader in research and development on acidic drainage for metal mines. Through MEND, Canadian mining companies and federal and provincial governments have reduced the liability due to acidic drainage by an estimated $340 million. It is also acknowledged that the reduction in liability is significantly higher than this quoted value, with a minimum of $1 billion commonly accepted. This is an impressive return on an investment of $17.5 million over nine years. A three-year program, called MEND 2000, was initiated in 1998 to further confirm MEND-developed reclamation technologies in the field. The key to MEND 2000 was technology transfer – providing state-of-the-art information and technology developments to users and to ensure that the information is clearly understood, particularly for newly-developed technologies.

THE MEND MANUAL

More than 200 technology-based reports were generated from the MEND and MEND 2000 programs. These reports represent a comprehensive source of information, however, it is not practical for users to have on hand or assimilate all the detailed information. For this reason, a single source of information on acidic drainage and on the results of MEND research is needed which is complimentary to many detailed technical reports. The MEND Manual describes the MEND-developed technologies and their applicability in terms of cost, site suitability and environmental implications - **a "toolbox" of techniques and options**.

The objective of the manual was to summarize work completed by MEND in a format that would provide practitioners in Canadian industry and government with a manageable single reference document. The document is not a “How to” manual. It is a set of comprehensive working

[1] The terms “acidic drainage”, “Acid Mine Drainage” (AMD) and “Acid Rock Drainage” (ARD) are used interchangeably throughout the manual to describe effluent generated from the oxidation of sulphide minerals

references for the sampling and analyses, prediction, prevention, control, treatment and monitoring of acidic drainage. The document provides information on chemistry, engineering, economics, case studies and scientific data for mine and mill operators, engineering design and environmental staff, consulting engineers, universities and governments.

The MEND Manual consists of six volumes. The information found in each volume is as follows:

Volume 1: Condensed "<u>stand-alone</u>" summary of the Manual.

Volume 2: <u>Sampling and Analyses</u>

- Water and solids sampling
- Biological sampling
- Geophysics and remote sensing
- Chemical analyses for water and solids

Volume 3: <u>Prediction</u>

- Laboratory static and kinetic geochemical tests methods
- Field methods
- Modelling

Volume 4: <u>Prevention and Control</u>

- Water covers
    - Non-oxidized and oxidized waste materials
- Dry covers
    - Soil and organic covers
- Disposal technologies
- Saturation (elevated water table)
- Blending and layering
- Separation and segregation
- Backfilling (in-pit) and co-disposal
- Permafrost

Volume 5: <u>Treatment</u>

- Active treatment methods
    - Chemical treatment
    - Metal recovery/recycling
    - Treatment byproducts
- Passive treatment
    - Anoxic limestone drains

- Aerobic wetland treatment systems
- Passive anaerobic treatment systems
- Biosorption treatment methods
- Passive *in situ* treatment methods
- Hybrid active/passive treatment systems

Volume 6: Monitoring
- Monitoring objectives
- Monitoring program design and data management
- Recent developments affecting acidic drainage monitoring

BACKGROUND ACIDIC DRAINAGE FROM SULPHIDE MINERALS

Base metal, precious metal, uranium, diamond and coal mines often contain sulphide minerals, in the mined ore and the surrounding rock. When these sulphide minerals, particularly pyrite and pyrrhotite, are exposed to oxygen and water, they oxidize, and the drainage may become acidic unless sufficient acid-neutralizing minerals such as calcite are present.

The acidic water may contain elevated concentrations of metals and salts. These can include typical major rock constituents (Ca, Mg, K, Na, Al, Fe, Mn) as well as trace heavy elements such as Zn, Cu, Cd, Pb, Co, Ni, As, Sb and Se. Rainfall and snow-melt flush leachate from the waste sites. If acidic drainage is left uncollected and untreated, the drainage can contaminate local water courses and groundwater, affecting plants, wildlife, and fish.

Naturally occurring alkalinity, such as carbonate minerals and carbonate ions in solution may partially or completely neutralize acidity *in situ*. The resulting leachate is non-acidic with very low iron concentrations[1] but can contain elevated concentrations of sulphate, calcium and magnesium.

Neutralization by reactions with acid consuming minerals (carbonate minerals in particular) may result in low concentrations of dissolved metals due to the low solubility of metal carbonates, basic carbonates, hydroxides and oxyhydroxides at pH 6 to 7.

DURATION

The lag time for acid drainage to appear (if at all) is controlled by the concentration and reactivity of the iron sulphides, and the availability of carbonate minerals. Acid may be generated and released by high sulphur wastes having small amounts of carbonate minerals a few days after exposure. Low sulphur (< 2%) wastes with some carbonate may not release acid for years or decades.

[1] Under anoxic conditions Fe will remain in solution in its reduced state

Once acidic oxidation of iron sulphide minerals is initiated the rate tends to increase until a peak is reached. The general trend is for a long-term decrease in acidity release. As the readily available mineral-grains are consumed, the reactive surface shrinks and oxidation product coatings limit reactivity. The rate of decrease is determined by numerous factors but mainly the reactivity of the sulphide minerals, the size of particles, and the availability of reactants (i.e. oxygen and other oxidants). The decrease in oxidation rates may not be apparent in mine waste drainage because oxidation products are stored and released over a long period during flushing events at a rate controlled by the solubility of the oxidation products.

SEASONAL EFFECTS

Under all climatic regimes, release of acidity is controlled, to varying degrees by seasonal precipitation patterns (e.g. transport medium). Under uniform precipitation conditions, the acid load and concentrations leached from a reactive waste are constant. As precipitation patterns vary, the following is observed:

- During dry spells, base flow conditions develop. A small proportion of the reactive surfaces are leached which allows oxidation products to build up in unleached sections;
- As infiltration increases (either due to snow pack melting or increased rainfall), a greater degree of leaching may occur due to rinsing of greater reactive surface areas. The contaminant load and usually the concentration increases;
- As wet conditions persist, the load leached decreases due to removal of acid products and flows are diluted resulting in lower concentrations; and
- When dry conditions are re-established, loads may be similar or lower than wet conditions but concentrations may increase.

SOURCES

Acidic drainage may originate from a variety of natural and man-made sources. Potential natural sources can include:

- Talus;
- Runoff from rock faces; and
- Groundwater seeps.

Man-made sources can include:

- Mines and associated facilities;
- Road cuts and fill;

- Quarries; and
- Other construction fill.

Mines are the major source of acidic drainage primarily because sulphide minerals are concentrated in geological environments containing ore deposits. In addition, rock removal and processing occurs on a large scale, and the methods involved (from blasting to processing) result in particle size reduction thereby increasing the surface area available for reactions. Some significant natural and non-mining sources of acidic drainage have also been documented. For example, at the Halifax International Airport in Nova Scotia, remedial measures are necessary to treat acidic drainage from excavated slates.

At active mine sites (and many inactive mine sites), systems are operated to collect and treat effluents and seepage, and prevent downstream environmental impacts. In some instances, acid generation may persist for hundreds of years following mine closure. The operation of treatment plants for very long periods of time is clearly not desirable. In addition, conventional water treatment technologies produce sludges with low solids content. In some extreme cases, the volume of sludge produced from the acidic drainage effluent can exceed the volume of tailings and/or waste rock. Storage capacity could become an issue for decommissioned mine sites.

LIABILITY ASSOCIATED WITH ACIDIC DRAINAGE

CANADA

Estimates for Canada in 1986 showed that acid generating waste sites totaled over 12,000 hectares of tailings and 350 million tonnes of waste mine rock. These wastes were observed to have mainly accumulated in the previous fifty years of mining. This survey did not represent the entire Canadian inventory, since it did not include abandoned mine sites for which responsibility had reverted to the responsible government authority.

The Canadian mine waste inventory was updated by CANMET in 1994 by surveying mining companies and provincial databases (MEND 5.8e). The results of this survey are summarized in Table 1. A complete national database on mine wastes has never been completed, although several provinces and territories have made considerable progress in defining their own mine waste inventories.

Using a wide variety of nationwide sources, estimates were made of the amount of acid-producing mine wastes (Table 2). Estimates of acid-producing and potentially acid-producing wastes are less accurate than the mine wastes for the following reasons:

- Only a portion of tailings and waste rock piles may be potentially acid producing;
- Some, or all, of the wastes may be stored in a way to eliminate acid potential; and
- Acid production may appear decades after the waste was produced.

Table 1
Estimates of Mine Wastes in Canada

	Tailings (tonnes * 10^6)	**Waste Rock (tonnes * 10^6)**
Newfoundland and Labrador	600	600
Nova Scotia	20	50
New Brunswick	80	30
Québec	1,900	2,700
Ontario	1,700	130
Manitoba	200	100
Saskatchewan	400	50
British Columbia	1,700	2,600
Territories	200	30
Canada	6,800	6,290

Table 2
Canadian Acid-Generating Wastes

	Tailings (tonnes * 10^6)	**Waste Rock (tonnes * 10^6)**
Newfoundland and Labrador	30	-
Nova Scotia	10	40
New Brunswick	80	30
Québec	250	70
Ontario	1,000	80
Manitoba	200	70
Saskatchewan	70	20
British Columbia	200	420
Territories	60	20
Canada	1,900	750

A summary of the estimated existing liability associated with acid-producing mine wastes is shown in Table 3. The assumptions made to calculate the reclamation and maintenance costs for the various options are presented in MEND 5.8e.

Table 3
Liability for Acidic Drainage from Mine Wastes

Waste	Options	$Billions
Tailings	Collect, Treat	1.5
	Water Cover	1.5
	Dry Cover	3.2
Waste Rock	Collect, Treat	0.4
	Dry Cover	0.7
	Relocate to Pit	2.1

The liability was estimated to be between $1.9 billion and $5.3 billion, depending on the sophistication of treatment and control technology selected. The most economical strategy to meet environmental objectives may be to collect water and treat it for a very long time, but such practice raises concerns about treatment product disposal and sustainability of the process.

WORLDWIDE

In the United States approximately 20,000 kilometres of streams and rivers have been impacted by acidic drainage, 85-90% of which receive acidic drainage from abandoned mines (Skousen 1995). Although there are no published estimates of total U.S. liability related to acidic drainage, several global examples indicate the scope of the issue:

- Leadville, a Superfund site in Colorado, has an estimated liability of US$290 million due to the effects of acidic drainage over the 100-year life of the mine;
- The Summitville Mine, also in Colorado, has been declared a Superfund site by the U.S. Environmental Protection Agency (USEPA). The USEPA estimated total rehabilitation costs at approximately US$175 million;
- More than US$253 million dollars have been spent on Abandoned Mine Lands reclamation projects in Wyoming (Richmond 1995);
- At an operating mine in Utah, U.S. regulators estimate the liability to be US$500-US$1,200 million (Murray *et al*. 1995);
- The Mineral Policy Center in the US has estimated that there are 557,000 abandoned mines in 32 states, and that it will cost between US$32 - $72 billion to remediate them (Bryan 1998); and
- Liability estimates for Australia in 1997 and Sweden in 1994 were $900 million and $300 million respectively (Harries 1997; Gustafsson 1997).

Based on these data, as well as the number of new mining projects under development, and mine sites in regions not mentioned above (Europe, South America, Africa), the total worldwide liability is estimated to be around US$100 billion.

MINE ENVIRONMENT NEUTRAL DRAINAGE (MEND) PROGRAM

In the 1970s and early 1980s, the Canadian mining industry and the government of Canada conducted research into methods of establishing sustainable vegetative growth on tailings and waste rock. At that time, closure of mine sites involved recontouring and revegetation for stability and erosion control. It was believed, at the time, that this technology would also address acidic drainage and allow the sites to be abandoned without future liability. Very successful re-vegetation methods were developed, and many sites were revegetated. However, after several years, the quality of water drainage from vegetated sites had not significantly improved, and mine site operators were faced with the prospect of operating water treatment plants indefinitely.

In response, the Canadian mining industry initiated a task force in 1986 to research new methods to remediate acid generating mines sites. The task force consisted of a steering committee and a technical working group, with representation from the mining industry, Energy, Mines and Resources, Environment Canada, British Columbia, Manitoba, Ontario, Québec and New Brunswick. It was referred to as the RATS (Reactive Acid Tailings Stabilization) task force. Its recommendations were published in July 1988 (MEND 5.5.1), and were implemented by the Mine Environment Neutral Drainage (MEND) program. Provincial groups worked with MEND to coordinate research. Provincial initiatives included:

- British Columbia - British Columbia Acid Mine Drainage (BC AMD) Task Force,
- Ontario - MEND Ontario (MENDO); and
- Québec - Programme de Neutralisation des eaux de drainage dans l'environnement minier (NEDEM Québec).

The initial MEND research plan was based on a five-year budget of $12.5 million (MEND 5.5.1).

Three years into the program, the original "RATS" plan was revised and a "Revised Research Plan" was produced. This plan expanded MEND to a 9-year program and the partners agreed to an expanded budget of $18 million (MEND 5.7.1). Planned funding for MEND was divided equally between the three major partners; the mining industry, the federal government and five provincial governments. When MEND ended in December 1997, the two levels of government together with the Canadian mining industry had spent over $17 million within the MEND program to find ways to reduce the estimated liability (Table 4).

Table 4
Funding Contribution by MEND Partners

Partners	Spent ($M)	Funding (%)
Federal government	6.3	37
Mining industry	6.7	39
Provinces	4.1	24

ORGANIZATION OF THE MEND PROGRAM

Two important objectives for MEND Program were established:

- To provide a comprehensive, scientific, technical and economic basis for the mining industry and government agencies to predict with confidence the long-term management requirements for reactive tailings and waste rock; and
- To establish techniques that will enable the operation and closure of acid generating tailings and waste rock disposal areas in a predictable, affordable, timely and environmentally acceptable manner.

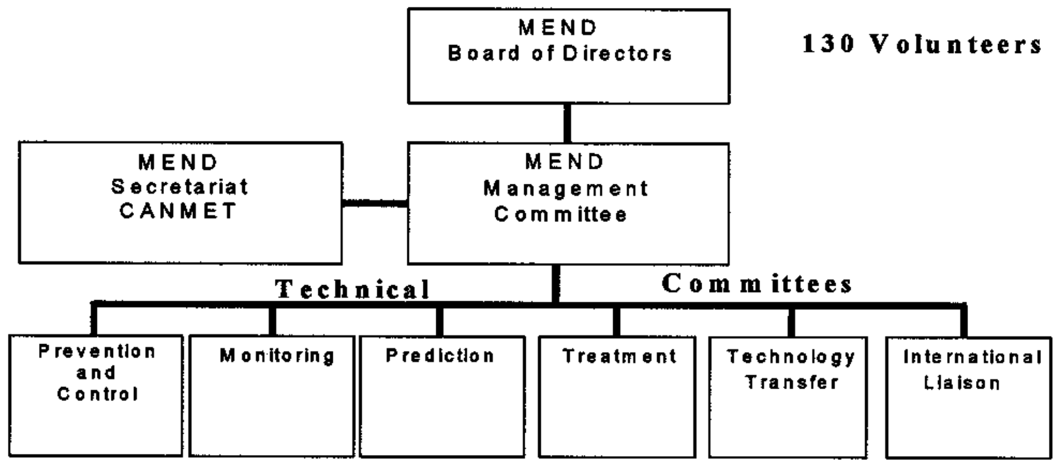
To ensure transparency of the program, it was also recommended that all research reports produced be made available to the partners and the public. Prior to their release the reports were critically reviewed and edited to enhance credibility and provide quality.

MEND adopted an organizational structure that included a Board of Directors, a Management Committee and several technical committees and a coordinating secretariat (Figure 1). The roles of these components were as follows:

- The Board of Directors provided vision and approval of yearly plans and budgets;
- The Management committee provided day-to-day management of the program; and
- The technical committees addressed technological issues and solutions.

The Secretariat ensured coordination of the elements within, and external to MEND. An important role of the Secretariat was to provide program and project management.

Figure 1: **MEND Organization**



MEND relied heavily on the 130 volunteer representatives of the different participating agencies: regulators, mining company managers and engineers, non-government organizations (NGOs) and government officials and scientists.

MAJOR ELEMENTS AND RESULTS OF THE CANADIAN RESEARCH

MEND organized its work into four technical areas: prediction, prevention and control, treatment and monitoring. The four technical committees were also involved in technology transfer and international activities.

Over 200 projects were completed. Some of the highlights of the MEND Program include:

Prediction and Modelling

- **Field studies of several waste rock piles provided important understanding for development of prediction techniques**. One of the most important observations was that waste rock piles accumulate extensive quantities of oxidation products and acidity that can be released to the environment in the future (MEND 1.14.3; MEND 1.41.4).

- **Geochemical and physical characteristics of a waste rock pile, from its origin in underground workings to its disassembly and placement underwater in a nearby lake was completed.** This study provided qualitative and quantitative information on mass transport and water infiltration within a waste rock pile. Geochemical processes were dependent on physical factors such as channeling or stratification within the dump (MEND 1.44.1).

- **Laboratory and field prediction tests for waste rock and tailings have been investigated and further developed**. These tests include static and kinetic tests, mineralogical evaluations and oxygen consumption methods.

- **An "Acid Rock Drainage Prediction Manual" for the application of chemical evaluation procedures for the prediction of acid generation from mining wastes was produced (MEND 1.16.1b)**.

- **Advances in the prediction of drainage quality for waste rock, tailings and open pit mines have been made**. A tailings model (RATAP) was distributed and a geochemical pit lake model was developed (MINEWALL). A critical review of geochemical processes and geochemical models adaptable for prediction of acidic drainage was completed (MEND 1.42.1).

- **Models that will predict the performance of dry and wet covers on tailings and waste rock piles are available (WATAIL, SOILCOVER).**

Prevention

- **Prevention has been determined to be the best strategy.** Once sulphide minerals start to react and produce contaminated runoff, the reaction is self-perpetuating. Also, at some mine sites, acidic drainage was observed many years after the waste pile had been established. With many old mine sites, there may be no “walk-away” solution;

- **In Canada, the use of water covers and underwater disposal are being confirmed as the preferred prevention technology for unoxidized sulphide-containing wastes.** A total of 25 reports and/or scientific papers have been prepared on subaqueous disposal (MEND 2.11). A generic design guide was developed (MEND 2.11.9). The guide outlines the factors involved in achieving physically stable tailings, and discusses the chemical parameters and constraints that need to be considered in the design of both impoundments, and operating and closure plans.

- **Underwater disposal of mine wastes (tailings and waste rock) in man-made lakes is presently an option favored by the mining industry to prevent the formation of acidic drainage.** At the Louvicourt Mine (Québec) fresh, sulphide-rich tailings have been deposited in a man-made impoundment since 1994. Laboratory and pilot-scale field tests to parallel the full-scale operation and evaluate closeout scenarios are ongoing (MEND 2.12.1).

- **The use of water covers to flood existing oxidized tailings can also be a cost effective, long lasting method for prevention of acid generation**. Both the Quirke (Elliot Lake, Ontario) and Solbec (Québec) tailings sites were subjects of MEND field and laboratory investigations (MEND 2.13.1 (Quirke); MEND 2.13.2 (Solbec)). These sites were decommissioned with water covers and are presently being monitored. Where mining wastes are significantly oxidized, laboratory results have shown that the addition of a thin sand or organic-rich layer over the sulphide-rich materials can prevent or retard diffusion of soluble oxidation products into the water column.

Control

- **Dry covers are an alternative where flooding is not possible or feasible**. MEND has extensively investigated multilayer earth covers for tailings and waste rock (e.g. Waite Amulet and Les Terrains Aurifères (tailings) and Heath Steele (waste rock): 3-layer systems). These type of covers use the capillary barrier concept and although they are effective, they are also costly to install in many areas of Canada.

- **Innovative "dry" cover research is indicating that a range of materials, including low cost waste materials from other industries (crude compost, lime stabilized sewage sludge, paper mill sludge) may provide excellent potential for generating oxygen-reducing surface barriers.** This technology would see the application of one waste to solve a problem of other wastes.

- **Non acid-generating tailings can be used as the fine layer in composite moisture-retaining surface barriers**. Laboratory studies have confirmed that sulphide-free fine tailings offers some promising characteristics as cover materials (MEND 2.22.2). Barrick's tailings site in Northwest Québec, Les Terrains Aurifères, is the first full-scale demonstration project of using tailings in a cover system (MEND 2.22.4). A second site, Québec crown-owned Lorraine, has also been rehabilitated using the same closure technique.

- **The first full-scale application in Canada of a geomembrane liner for close-out was completed in 1999 at Mine Poirier in Northwest Québec.** Performance monitoring of the close-out scenario to evaluate the liner is ongoing (Lewis and Gallinger 1999).

Disposal Technologies

- Several other disposal technologies that will reduce acid generation and have been investigated include:

- **Permafrost in northern environments.** Permafrost covers approximately 40% of Canada, and cold conditions inhibit oxidation. Predictive methods have been researched. Although acid generation is common in cold environments, it occurs when exposed sulphides are warmed to temperatures above freezing (MEND 1.61.1-3; MEND 1.62.2).

- **Blending and segregation (or layering).** Technology is defined as the mixing of at least two rock waste types with varying acid generation potential, neutralization potential and metal content to produce a pile that has seepage water quality acceptable for discharge without additional measures (MEND 2.37.1; MEND 2.37.3).

- **Elevated water table in tailings.** This technique offers a method of inhibiting the oxidation of sulphides through the effective saturation of pore spaces. It may be applied as one component of a multi-component reclamation strategy (MEND 2.17.1).

- **In-pit disposal following mining.** Mined-out pits can provide a geochemically stable environment for wastes and can be a focal point in mine rehabilitation. The addition of buffering material may be required (MEND 2.36.1).

- **Depyritized tailings as cover materials**. Laboratory and field tests are showing that depyritized tailings have excellent potential as covers. Economic analyses have indicated that hydraulic placement will be necessary to be cost effective (MEND 2.22.3).

<u>Lime Treatment</u>

- **Studies conducted to date support the view that sludges will remain stable if properly disposed**. Concerns had been raised with regard to the long-term chemical stability and the potential liability arising from dissolution of heavy metals contained in the sludge (MEND 3.42.2). Other findings include:
  - Optimum conditions will depend on site-specific factors e.g. pH, metal loading chemistry;
  - Modifications to the treatment process (e.g., lime slaking, pH adjustment, mixing, aeration, flocculent addition) can influence operating costs, sludge volumes, and metal release rates (Zinck and Aubé 1999);
  - The method of disposal of the sludge will affect its long-term stability. Aging can promote recrystallization which improves sludge stability;
  - Codisposal of sludges with other mining wastes requires further study; and
  - Leach test protocols need to be developed specifically for lime treatment sludges.

- **The status of chemical treatment and sludge management practices was summarized in a reference document (MEND 3.32.1)**.

Passive Treatment

- **In Canada, experience indicates that passive systems do have specific applications for acid mine drainage (AMD) treatment.** These applications range from complete systems for treating small seeps to secondary treatment systems such as effluent polishing ponds. Alone, they cannot be relied upon to consistently meet AMD discharge standards. Large-scale passive systems capable of handling the low winter temperatures, high metal loads, and fluctuations in flow rates associated with the spring freshet have yet to be implemented.

- **The status of passive systems for treatment of acidic drainage was summarized in a reference document (MEND 3.14.1)**.

Monitoring

- **Several guides are available to assist in the development of acidic drainage monitoring programs.** An important MEND deliverable is MEND 4.5.4, *Guideline Document for Monitoring Acid Mine Drainage.* This document is designed to serve as a single source introductory guide to a wide range of AMD monitoring concerns, while also providing users with information on literature sources for site-specific concerns and emerging monitoring techniques. Monitoring requirements are addressed for both source and receiving environments, with receiving environment concerns restricted to freshwater systems.

  Other guideline documents include a field sampling manual (MEND 4.1.1) that presents an approach to assist people in selecting the appropriate methodologies for the sampling of tailings solids, liquids and pore gas. A comprehensive list and description of sampling techniques, and a guide to waste rock sampling program design for the exploration, operation and closure phases of a mining project is produced in MEND 4.5.1-1. Available sampling techniques for waste rock is given in MEND 4.5.1-2.

At the conclusion of the MEND program, a “tool box” of technologies has been developed to assist the mining industry in addressing its various concerns related to acidic drainage, and in significantly reducing its estimated liability. A particularly important outcome has been the development of a common understanding among participants, inasmuch as it has allowed operators to take actions with greater confidence and to gain multi-stakeholder acceptance more rapidly.

NEW IDEAS

In 1992, a Task Force was formed to solicit and nurture innovative new ideas. An additional goal was to encourage researchers from outside the general area of mining environment to becoming involved in acid drainage research. The resulting technology would need to be reliable, inexpensive, permanent, and widely applicable. The innovator had to demonstrate the relevance of the idea at the concept level, which would then be the basis for proceeding to a more detailed development project.

A two-page proposal format was developed and distributed across Canada. New ideas were solicited in two rounds. A total of 135 proposals were received and 18 were funded. Up to $10,000 was provided for the review and the development of the concepts. Table 1.4-1 lists the new ideas projects funded by the Task Force.

Table 5
New Ideas Projects

New Idea #	Project Title
01	Status of AMD Research in the U.S.
02	Permafrost to Prevent AMD
03	Low Cost Wax Cover
04, 21	Chemically Modified Clay Cover
05	Ferric Phosphate Coating of Pyrrhotite
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07	Formation of Hardpan in Pyrrhotite Tailings
08	Passivation of Sulphide Minerals
09, 22	Selective Ion Exchange Resin
10	Chelating Ribbons
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23	Sprayed Polyurethane Covers
24	Literature Review: Foam Flotation
25	Limestone Precipitation Layer in Coal Wastes
26	Ion Flotation for Zinc Recovery

The eighteen New Ideas projects cost about $230k. Although most of the new ideas were innovative and applicable and provided useful information, they did not achieve the objective of providing a solution to the problem of acidic drainage. However, at least three had potential applications (sprayed polyurethane, modified clay and permafrost) and three yielded useful state-of- art reviews (U.S. research, foam flotation and Japanese technology). Also, a large number of additional researchers were made aware of the acidic drainage problem, and some may in due course make useful contributions. Finally, the endeavour increased MEND's certainty that a magic answer was not overlooked.

TECHNOLOGY TRANSFER

Technology transfer activities were expanded in the later years of the program. The dissemination of information on developed technologies to the partners and the public was a major function of the program. A MEND 2000 Internet site (**http://mend2000.nrcan.gc.ca**) was established and is updated with current information on technology developments. The site provides report summaries, the MEND publication list, information on liabilities, case studies, and conference and workshop announcements. MEND and MEND 2000 hosted several workshops per year at various locations across Canada. Proceedings for the workshops on chemical treatment, economic evaluations, in-pit disposal, dry covers, monitoring, case studies of Canadian technologies, research work in Canada, and risk assessment and management are available from the MEND Secretariat.

MEND participated in the organization of several International Conferences on the Abatement of Acid Rock Drainage (ICARDs) held in 1991 (2^{nd} – Montreal), 1994 (3^{rd} - Pittsburgh) and 1997 (4^{th} -Vancouver).

Other technology transfer initiatives included:

- MEND videos are available in English, French, Spanish and Portuguese. They describe technological advances relating to the prediction, prevention and treatment of acidic drainage from mine sites;
- The MEND Manual that summarizes all of the MEND and MEND-associated work on acidic drainage from mine wastes;
- The Proceedings of the 4^{th} International Conference on Acid Rock Drainage are available on CD-ROM;
- About 200 reports completed during MEND and MEND 2000;
- MEND reports on CD-ROM. A project to have the key MEND reports available on CD-ROM will be completed in 2001; and
- National case studies on acidic drainage technologies.

THE MEND MODEL

MEND has been described as a model way for governments and industry to cooperate in technology development for advancing environmental management in the mining industry.

Decisions are now being made based on findings from scientific research. Reasons for this include:

- The high return on the investment targeted and achieved, in terms of knowledge gained and environmental and technical awareness of the scope of the acidic drainage problem and credible scientific solutions;
- The partnership and improved mutual understanding developed between the two levels of government, the mining industry and NGOs in search of solutions to a major environmental problem;
- The secretariat group which coordinated activities, managed the accounting, reporting and technology transfer;
- The peer review process that was both formal and informal, and resulted in enhanced credibility of the information base; and
- The approach taken for transferring the knowledge gained during MEND.

In large part as a result of MEND, it was shown that new mines are able to acquire operating permits faster and more efficiently than before since there are now accepted acidic drainage prevention techniques. As an example, the Louvicourt mine in northwest Québec adopted MEND subaqueous tailings disposal technology and has been able to progress from the exploration phase to an operating mine within 5 years, with a reduced liability of approximately $10 million for the tailings impoundment. Similar benefits are reported for existing sites in the process of decommissioning. MEND has also fostered working relationships with environmental groups, ensuring that they are an integral part of the process.

MEND 2000

MEND concluded on December 31, 1997. However, the partners agreed that additional cooperative work was needed to further reduce the acidic drainage liability and to confirm field results of MEND-developed technologies. MEND 2000 was a three-year program that officially started in January 1998. The program was funded equally by the Mining Association of Canada (MAC) and Natural Resources Canada, a department of the Canadian government. The objectives of MEND 2000 were to:

- Transfer and disseminate the knowledge gained from MEND and other related acidic drainage projects;
- Verify and report the results of MEND developed technologies through long-term monitoring of large scale field tests;
- Maintain links between Canadian industry and government agencies for information exchange and consensus building; and

- Maintain linkages with a number of foreign government and industry driven programs (e.g. International Network on Acid Prevention (INAP), the Mitigation of the Environment Impact from Mining Waste (MiMi - Sweden), and the Acid Drainage Technology Initiative (ADTI - USA).

An important function of MEND is technology transfer. All research results must be effectively communicated to industry, government agencies and the public if the program is to continue to achieve the desired results.

CONCLUSION

The benefit of the MEND programs has come through the sharing of experiences, the thorough evaluation of technologies and their incremental improvement. Mining companies and consultants have acquired more capabilities to deal with water contamination from mine wastes, including acid generation. No dramatic technological breakthrough other than water covers has been achieved. Nonetheless, Canadian industry reports that a significant reduction in liability is predicted. An evaluation of MEND in 1996 concluded that the estimated liability had been reduced by $340 million, for five Canadian mine sites alone (MEND 5.9). It is also acknowledged that the reduction in liability is significantly higher than this quoted value, with a minimum of $1 billion commonly accepted. The same study concluded:

- There is a much greater common understanding of acidic drainage issues and solutions;
- The research has led to reduced environmental impact;
- There is increased diligence by regulators, industry and the public; and
- The work should continue with strong international connections.

As a result of MEND and associated research, technologies are in place to open, operate and decommission a mine property in an environmentally acceptable manner, both in the short and long term.

MEND is an example of a successful, multi-stakeholder program addressing a technical issue of national importance, and has been a model for cooperation between industry, environmental groups and various levels of government.

MEND AND RELEVANT PUBLICATIONS

MEND Reports are cited in Appendix C.

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Harries, J. 1997. *Estimating the Liability for Acid Mine/Rock Drainage in Australia*. In: Proceedings of the 4th International Conference on Acid Rock Drainage. Vancouver, British Columbia. 31 May to 6 June. Vol 4, pp. 1905-1918.

Lewis, B.A. and Gallinger, R.D. 1999. *Poirier Site Reclamation Program*. In: Sudbury '99 – Mining and the Environment II Conference Proceedings. Sudbury, Ontario. 13 to 17 September. Vol 2, pp. 439-448.

Murray, G.S.C., Ferguson, K.D. and Brehaut, C.H. 1995. *Financial and Long Term Liability Associated with AMD*. In: Proceedings of the 2nd Australian Acid Mine Drainage Workshop. Charters Towers, Queensland. 28 to 31 March. pp. 165-177.

Price, B. 1995. *Defining the AMD Problem: II. An Operator's Perspective*. In: Proceedings of the 2nd Australian Acid Mine Drainage Workshop. Charters Towers, Queensland. 28 to 31 March. pp. 17-19.

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Skousen, J. 1995. *Prevention of Acid Mine Drainage*. In: Acid Mine Drainage Control and Treatment. (Eds. J.G. Skousen and P.F. Ziemkiewicz). West Virginia University and the National Mine Land Reclamation Center, Morgantown, West Virginia. pp. 13-14.

Zinck, J.M. and Aubé, B. 1999. *Optimization of Lime Treatment Processes*. In: Proceedings of the 31st Annual Meeting of the Canadian Mineral Processors. Ottawa, Ontario. 19 to 21 January. pp. 75-93.

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4.0 PREVENTION AND CONTROL

This volume describes the research completed under the auspices of MEND, and by others, in acid generation prevention and control. MEND made significant contributions in this area. MEND expenditures in the prevention and control of acidic drainage amounted to over $9,000,000. This substantial expenditure was beneficial as it led to the development and applications of numerous technologies. These applications are not necessarily universally applicable, and as such site-specific conditions need to be taken into account. Recent experience in applying these technologies has also demonstrated the need for good quality, long-term data.

Several MEND studies have assessed the suitability of acid generation prevention and control technologies for different types of sites and conditions. These studies have shown that the most suitable acid prevention and control technology is likely to depend on site-specific conditions and costs as well as expected benefits and drawbacks, and other environmental/regulatory considerations. The process of selecting the most suitable acid prevention or control technology should in most cases include an assessment of available technologies. At some sites, it may be possible to develop a cost-effective acid prevention or control strategy that involves more than one acid prevention and control technology.

While a tremendous amount of work has been completed in this area, there is a need for further research and development to address areas of uncertainty and reduce costs that in some cases can be excessive and unaffordable. Long-term monitoring has been commenced at a number of sites to assist in confirming the performance of available technologies. The key results of selected projects are presented in this volume.

4.1 INTRODUCTION

4.1.1 BACKGROUND AND CONTEXT

As a result of MEND, the knowledge base on acidic drainage has grown considerably to include a fundamental understanding about the acid generation process and measures that may be taken to prevent and control acid generation. And while the present knowledge base can be generally described as reasonable and adequate, we do not have all of the answers.

Recent experience in applying acidic drainage technologies have included numerous projects that are potential success stories, and others that have simply provided valuable information. The application of new technologies to prevent and control acidic drainage should be considered a work in progress, and where valuable information has already been generated.

Experience in the closure of acid-generating mine waste sites has shown that the prevention of acidic drainage should be a first objective when this is achievable and affordable. At sites where acidic drainage occurs, active and passive treatment technologies are available to treat the drainage prior to its release or recycle – these technologies are reviewed in Volume 5.

Readers are encouraged to use this volume as a reference to the MEND reports and other research documents, and to follow-up as necessary. Readers are also reminded that, for the many reasons given in the manual, this is not a *"how to"* manual.

The prevention and control of acidic drainage is a technically complex area, and one that typically requires the involvement of experts from numerous technical disciplines. Site-specific factors and conditions add to this complexity, and often necessitate site-specific research. As a result, acidic drainage technologies are not universally applicable. Prohibitive cost or other factors may negate the application of a particular technology to a site.

The Guide to the Management of Tailings Facilities (MAC 1998) stresses that tailings management facilities are site-specific, with each site having its unique setting both physically and environmentally. Industry is strongly encouraged to ensure that their mine waste management facilities, including the waste containment structures, water management and control programs, and monitoring and facility audit systems are confidently designed, constructed, operated and monitored with qualified technical personnel. In addition, mine waste management programs should be an integral and important part of an overall environmental management system for the organization. An effective environmental management system addresses all environmental activities within an organization, including the organizational structure, planning activities, responsibilities, procedures, processes, and a commitment to provide required resources. The ISO 14001 International Standard (ISO 1996) is a useful reference in this area.

Recent developments affecting acidic drainage monitoring include environmental effects monitoring and the Metal Mining Effluent Regulations – these aspects are discussed in Volume 6, Section 6.3. Useful reference reports include the Synthesis Report on Monitoring Methods issued by the Aquatic Effects Technology Evaluation Program (AETE 4.1.4), and MEND workshop notes.

While MEND research has focused on acidic drainage issues, it also assessed related issues such as risk assessment. Specifically, MEND 2.11.9 reviewed the risk assessment process as it applies to the subaqueous (underwater) disposal of tailings in engineered impoundments. MEND held a workshop on risk assessment in Vancouver in 1998 (MEND BC.01) and another in Sudbury in 1999 (MEND ME.01). The main objective of the workshops was to communicate the application of risk management techniques to mine waste management.

All activities and actions undertaken by individuals and groups in society have associated levels of risk, where a risk, in general terms, represents the possibility of an unfavourable condition or event occurring. From an engineering or environmental perspective, risk can be defined as the mathematical product of the probability of an event occurring and the consequences of that event. Risk assessment has become an important component of mine waste management facility planning and operation.

In recent years, risk assessment approaches have evolved to be more comprehensive and viewed as the risk management component of decision-making processes (MEND BC.01, MEND ME.01, SENES 2000). Probabilistic methods can be used to assess and present a range of confidence for the various elements (i.e. hazards, likelihood, and consequences) in the risk assessment process. These techniques can be applied to identify impacts or benefits associated with proposed actions, and determine the sensitivity of outcomes with respect to the underlying assumptions. Risk assessment techniques used by industry range from screening level studies to detailed and sophisticated modelling studies.

Risks associated with acidic drainage extend from the risk of acid generation, to the risk of adverse impacts to the receiving environment and related liabilities. Considerable work has been completed in the area of characterizing mine wastes to determine whether or not a waste will become acid generating. The regular sampling of mine wastes as they are produced, and monitoring are two methods that are used to address uncertainty in the net acid generation potential of wastes.

Risk assessments of options to mitigate acidic drainage concerns have largely focused on external drivers related to acid generation and potential impacts to the environment. Risk assessment tools provide a framework to identify and communicate risks associated with a particular acidic drainage prevention or control initiative. As an example, a risk assessment of the use of a water cover over tailings should identify: what could potentially go wrong (i.e. loss of tailings containment dam, seismic events, storm events, effects of droughts, etc.)?; how often are these events likely to happen?; and, if they were to occur, what would the consequences be? This example again indicates that acidic drainage is only one component of mine waste management.

SENES (2000) reports that the selection of an appropriate level of risk can be weighed, as is the case with the U.S. EPA, against the size of populations exposed and what is reasonably achievable. However, risk acceptance will vary with an individual's background (e.g. technical, social, etc.). A risk assessment will not necessarily identify the "best" option for all attributes. As an example, a waste management option having a low probability of acidic drainage release may also be cost prohibitive or otherwise unreasonable. In such as case, the "best overall approach" may be the most appropriate.

4.1.2 STRUCTURE AND SUBJECT AREAS

Prevention and control technologies are described under the following subject headings.

Section	Subject Heading
4.2	Water Covers
4.3	Saturation
4.4	Dry Covers
4.5	The Co-Disposal of Tailings and Waste Rock
4.6	Blending and Layering of Waste Rock
4.7	Separation and Segregation
4.8	Permafrost and Freezing
4.9	Backfilling

4.1.2.1 Water Covers

MEND research showed that the oxidation of sulphide minerals can be inhibited by the presence of a water cover, as the water acts as a barrier to the diffusion of oxygen from the atmosphere to the submerged sulphides. Potential disposal options include: 1) the subaqueous disposal of unoxidized sulphidic wastes under a water cover; and 2) the flooding of oxidized wastes.

In Canada, the use of water covers and underwater disposal are being confirmed as the preferred prevention technology for unoxidized sulphide-containing wastes. These aspects were addressed through initial investigations of historically submerged tailings in natural basins with extensive and detailed geochemical investigations. A total of 25 reports and/or scientific papers have been prepared on subaqueous disposal (MEND 2.11), and a generic design guide (MEND 2.11.9) is available. The guide outlines the factors involved in achieving physically stable tailings, and discusses the chemical parameters and constraints that need to be considered in the design of impoundments and operating and closure plans.

Underwater disposal of unoxidized mine wastes (tailings and waste rock) in man-made lakes is presently an option favored by the mining industry to prevent the formation of acidic drainage. At the Louvicourt Mine (Québec) fresh, sulphide-rich tailings have been deposited in a man-made impoundment since 1994. Laboratory and pilot-scale field tests to parallel the full-scale operation and evaluate closeout scenarios have been investigated (MEND 2.12.1).

The underwater disposal of mine wastes can be challenging and may not be an acceptable disposal method at some sites. MEND addressed a range of issues related to the subaqueous disposal of mine wastes including the effects on the sulphide oxidation process, and concerns for aspects such as: the effect of water depth, oxygenated water, currents, waves; impacts to/from pore water; biological influences; water cover quality; and potential benefits of diffusion barriers at the waste/water cover interface.

The key challenge for flooding oxidized wastes is that contaminated pore water and soluble mineral phases can be released, resulting in unacceptable contaminant concentrations and loadings to the water cover and to the receiving aquatic environment. The flooding of oxidized wastes can also necessitate the treatment of excess pond water prior to its release.

Both the Quirke (Ontario) and Solbec (Québec) tailings sites were subjects of MEND field and laboratory investigations (MEND 2.13.1 (Quirke); MEND 2.13.2 (Solbec)). These sites were decommissioned with water covers and are presently being monitored. Where mining wastes are significantly oxidized, laboratory results have shown that the addition of a thin sand or organic-rich layer over the sulphide-rich materials can prevent or retard diffusion of soluble oxidation products into the water column.

Water covers have been applied at many sites, but are not universally applicable. Related issues such as the ability to maintain a water cover over the long-term, the integrity of the containment structures, locality and site-specific potential risks due to seismic events, severe storm events, etc. can negate the use of this technology. However, under suitable conditions, the present state of knowledge is sufficient to allow for the responsible design, operation and closure of waste management facilities using water covers for both fresh and oxidized tailings and waste.

4.1.2.2 Saturation

Saturation refers to the moisture saturation of tailings pore spaces to make good use of the low rate of oxygen diffusion through water-filled pore spaces in comparison to those that are gas-filled. In this light, MEND investigated the innovative use of elevated water tables in tailings as a means of saturating the tailings while eliminating the need to maintain a water cover (MEND 2.17.1). The use of an elevated water table by itself does not prevent acid generation, as there may be zones of near-surface exposed and drained tailings that remains available for oxidation. The use of an elevated water table can, however, significantly reduce the inventory of sulphide tailings available for oxidation. Technologies that can be applied to make use of elevated water table conditions, including the use of thickened or paste tailings, were reviewed by MEND. The use of elevated water concepts may be cost advantageous when applied in tandem with other approaches to prevent and control acid generation.

4.1.2.3 Dry Covers

Dry cover systems are commonly used to decommission waste rock piles and tailings impoundments at sites around the world. The key objective of dry cover systems is to provide a barrier that minimizes the influx of atmospheric oxygen to the mine waste, and to limit moisture infiltration. Apart from these functions, dry covers are also expected to be resistant to erosion, and provide a media for vegetation.

Dry covers can range from a single layer of earthen material to several layers of different material types, including native soils, non-reactive tailings and/or waste rock, geosynthetic materials, and oxygen consuming materials. Multi-layer cover systems utilize the capillary barrier concept to keep one (or more) of its layers near saturation under all climatic conditions. This creates a “blanket” of water over the reactive waste material, which reduces the influx of atmospheric oxygen and subsequent production of acidic drainage.

MEND research on dry covers commenced in 1988. Multi-layer soil cover systems for tailings and waste rock were extensively investigated (e.g. Waite Amulet (MEND 2.21.2 – tailings), Les Terrains Aurifères (MEND 2.22.4 - tailings) and Heath Steele (MEND 2.31.1 - waste rock)). A review of soil cover technologies (MEND 2.21.3) is available, while a design guide for earthern covers is in preparation (MEND 2.21.4). These MEND reports are likely to serve as useful references as they address cover design, selection of cover materials, construction aspects, instrumentation. They will also review the performance monitoring of covers that are exposed to Canadian climatic conditions.

Supplying suitable and sufficient quantities of cover materials can be challenging and cost prohibitive at some sites. MEND addressed this aspect through evaluations of the use of alternate cover materials, such as sulphide-free tailings and organic waste materials, instead of natural soils.

Laboratory studies have shown that sulphide-free fine tailings offer promising characteristics for use in dry cover (MEND 2.22.2). Barrick’s tailings site in Northwest Québec, Les Terrains Aurifères, provided the first full-scale demonstration project of using tailings in a cover system (MEND 2.22.4). A second site, Québec crown-owned Lorraine, was also been rehabilitated using the same closure technique.

Innovative dry cover research has noted that a range of materials, including low cost waste materials from other industries (crude compost, lime stabilized sewage sludge, paper mill sludge) may provide excellent potential for generating oxygen-reducing surface barriers (MEND 2.20.1). This approach would see the application of one waste to solve a problem of another waste.

MEND 2.20.1 evaluated the use of alternate materials, including geomembranes, in constructing dry covers. The first full-scale application in Canada of a geomembrane liner for close-out was completed in 1999 at La Mine Poirier in Northwest Québec. Performance monitoring of the close-out scenario to evaluate the liner is ongoing (Lewis and Gallinger 2000).

4.1.2.4 The Co-Disposal of Tailings and Waste Rock

As its name implies, co-disposal refers to the combined disposal of waste rock and tailings. MEND investigated this concept with a study into the possibility of injecting tailings into waste rock piles.

Co-disposal is expected to remain an active area of interest for a number of reasons including: the potential to minimize the footprints of mine waste management areas; potential opportunities to establish and create elevated water table conditions within waste rock piles; and the possible elimination of tailings containment structures at some sites.

4.1.2.5 Blending and Layering of Waste Rock

The blending and layering of waste rock is an approach that is founded on the geochemistry of sulphide oxidation. In concept, a net acid generating waste could be blended with, or layered between, alkaline material to produce a non-acid generating waste that has seepage water quality acceptable for discharge without additional measures (MEND 2.37.1; MEND 2.37.3). This is a challenging area that continues to be under development given the potential benefits.

4.1.2.6 Separation and Segregation

MEND had a key role in sulphide tailings separation research where a key driver was to identify techniques to separate sulphide solids for disposal, or produce reduced sulphur content tailings for use in site rehabilitation. MEND identified suitable methodologies in this regard, however, high cost remains a major drawback.

Laboratory and field tests are showing that depyritized tailings have excellent potential for use in dry covers. Flotation has been shown to be an effective, albeit costly, method of reducing the sulphide content of tailings prior to their discharge. The extent to which tailings sulphur levels need to be reduced to prevent acidic seepage was also investigated under MEND. Economic analyses indicate that the hydraulic placement of desulphurized tailings would be more cost effective in comparison to mechanical placement (MEND 2.22.3).

4.1.2.7 Permafrost

The potential for making best use of cold climatic conditions and permafrost was investigated under MEND. Research has shown that the sulphide oxidative process slows considerably as the temperature of the waste drops and approaches 0ºC. MEND projects have also assessed the effects of freezing on metal leaching, and conceptual acid control strategies for use in cold climatic conditions.

Permafrost in continuous and discontinuous forms is present across northern Canada, and covers about 40% of Canada's land mass. A reasonable and sufficient knowledge base exists with regards to the construction of a variety of structures over permafrost. However, knowledge and experience in the use of permafrost, including the natural and assisted freezing of sulphide wastes, is an area that continues to be developed.

MEND 6.1 reviews concepts for the prevention of acidic drainage through the disposal of tailings in permafrost conditions. This report would serve as a useful introduction for readers. Section 4.8 of this volume provides an overview of MEND research in permafrost, and a discussion on the general approach that is being taken in planning mine waste disposal in permafrost.

4.1.2.8 Backfilling

In recent years, sulphide tailings and waste rock have been disposed in mined-out open pits with the specific objectives of preventing or controlling acid generation. While this practice has been extensively applied and there is a sound technical base for in-pit disposal programs, few sites provide scientific databases that can be used to assess the performance of these technologies.

MEND 2.36.1 investigated the backfilling of open pits and identified a range of approaches for the in-pit disposal of mine wastes. The project determined that not all pits are suitable for in-pit disposal. In-pit disposal has been used on many occasions, however, there have been few scientifically monitored pits. Under suitable conditions, in-pit disposal can be both effective and affordable.

The backfilling of mines has been extensively practised internationally. Only recently, however, have backfilling programs been designed specifically for the disposal of sulphide tailings or waste rock to inhibit acidic drainage.

4.1.3 MEND AND RELEVANT PUBLICATIONS

MEND 2.11.9 1998. Design Guide for the Subaqueous Disposal of Reactive Tailings in Constructed Impoundments. March.

MEND 2.12.1 2001. Evaluation of Man-Made Subaqueous Disposal of Reactive Tailings in Constructed Impoundments. To be published.

MEND 2.13.1a 1998. Wet Barriers on Pyritic Uranium Tailings - Part I and II - Laboratory Lysimeter Studies of Oxidation and Limestone Neutralization Characteristics of Uranium Tailings and Waste Rock. January.

MEND 2.13.1b 1998. Wet Barriers on Pyritic Uranium Tailings - Part III - Laboratory Diffusion Lysimeter Studies of Uranium Tailings Deposited under a Shallow Water Cover. January.

MEND 2.13.2a 1993. Expérimentation de l'inondation du parc à résidus miniers – Solbec-Cupra (Phase IV). Rapport sommaire. Décembre.

MEND 2.13.2b 1994. Flooding of a Mine Tailings Site – Solbec-Cupra. Suspension of Solids: Impact and Prevention. March.

MEND 2.13.2c 1994. Inondation artificielle du parc à résidus miniers Solbec-Cupra: Études microbiologique et chimique - Rapport final. Mars.

MEND 2.13.2d 2001. Suivi environnemental du pars à résidus minier Solbec - 1994-1997.

MEND 2.17.1 1996. Review of Use of an Elevated Water Table as a Method to Control and Reduce Acidic Drainage from Tailings. March.

MEND 2.20.1 1994. Evaluation of Alternate Dry Covers for the Inhibition of Acid Mine Drainage from Tailings. March.

MEND 2.21.2 1993. Field Evaluation of the Effectiveness of Engineered Soil Covers for Reactive Tailings. October.

MEND 2.21.3a 1997. Review of Soil Cover Technologies for Acid Mine Drainage – A Peer Review of the Waite Amulet and Heath Steele Soil Covers. July.

MEND 2.21.4 2001. Design, Construction and Monitoring of Earthen Covers for Waste Rock and Tailings. To be published.

MEND 2.22.2a 1996. Évaluation en laboratoire de barrières sèches construites à partir de résidus miniers. Mars.

MEND 2.22.3 1998. Valorisation des résidus miniers – une approche intégrée. Avril.

MEND 2.22.4a 1999. Construction and Instrumentation of a Multi-Layer Cover Les Terrains Aurifères. February.

MEND 2.22.4b 2000. Field Performance of the Les Terrains Aurifères Composite Dry Covers. March.

MEND 2.31.1a 1992. Heath Steele Waste Rock Study, Phases I to III. March.

MEND 2.31.1b 1996. Monitoring Report Phase IV – Composite Soil Cover Acid Waste Rock Study Heath Steele Mines. March.

MEND 2.31.1c 1998. Monitoring Program 1995-96, Phase V Composite Soil Cover on Waste Rock Pile 7/12 at Heath Steele, New Brunswick. February.

MEND 2.36.1 1995. Review of In-Pit Disposal Practices for the Prevention of Acid Drainage – Case Studies. September.

MEND 2.37.1 1998. Blending and Layering Waste Rock to Delay, Mitigate or Prevent Acid Rock Drainage and Metal Leaching: A Case Study Review. April.

MEND 2.37.3 1997. Control of Acidic Drainage in Layered Waste Rock: Laboratory Studies and Field Monitoring. September.

MEND 6.1 1993. Preventing AMD by Disposing of Reactive Tailings in Permafrost. December.

MEND BC.01 1998. 5th Annual BC MEM – MEND 2000 Metal Leaching/ARD Workshop. Risk Assessment and Management. Vancouver, British Columbia. December.

MEND BC.03 2000. 7th Annual BC MEM – MEND 2000 Metal Leaching/ARD Workshop (Dry Covers, Predicting Solubility Constraints, Research and New Developments). Vancouver, British Columbia. November.

MEND ME.01 1999. Assessment and Management of Risks Associated with Metal Leaching and Acid Rock Drainage at Mine Sites. Workshop Notes. Sudbury, Ontario. September.

AETE 4.1.4 1999. AETE Program Synthesis Report of Selected Technologies for Cost-Effective Environmental Monitoring of Mine Effluent Impacts in Canada. March.

ISO – International Organization for Standardization 1996. ISO 14001, Environmental Management Systems – Specification with Guidance for Use. First Edition. 1 September.

Lewis, B.A., Gallinger, R.D. and Wiber, M. 2000. *Poirier Site Reclamation Program*. In: Proceedings of the 5th International Conference on Acid Rock Drainage. Denver, CO. 21 to 24 May. Vol 2, pp. 959-968.

MAC – The Mining Association of Canada 1998. A Guide to the Management of Tailings Facilities. September.

SENES Consultants Limited 2000. State-of-the-Art of Risk Assessment Application to ARD. Report prepared for INAP - The International Network of Acid Prevention. April.

4.2 WATER COVERS

Research has demonstrated that the oxidation of sulphides in mine tailings and waste rock is inhibited by placing the mine wastes under a water cover. The suitability of this method is subject to site specific factors, and as such is not universally applicable. However, for sites where water covers can be used, the method offers one of the best solutions for preventing sulphide oxidation and acid generation over the long term.

The practice of disposing of sulphide rich tailings under water is based on inhibiting oxidation, and acid generation and the potential for the release of metals into the aquatic system. Primary concerns that have been addressed are the potential deterioration of receiving water quality and the impact on biological resources from excessive concentrations of dissolved metals.

Potential disposal options that involve the use of a water cover include:

- Use of water covers to flood existing tailings facilities;
- Disposal in an engineered tailings management facility that includes a water cover that submerges the mine wastes; and
- Disposal in a natural lake or marine environment.

Research projects on water covers have been conducted under the auspices of the BC AMD Task Force and the MEND program since 1988. For the purpose of this section, the applications of water covers to fresh and oxidized tailings are differentiated as follows:

- **Subaqueous disposal** refers to the placement of fresh tailings under a water cover.
- **Flooding** refers to the submergence of oxidized tailings where the water cover is typically contained by engineered structures.

At sites where fresh tailings are subaqueously disposed within the confines of an engineered tailings management facility, also referred to as a man-made tailings impoundment, the sulphide materials are typically disposed beneath the water cover at the earliest opportunity and kept submerged.

The impoundment of a water cover presents a level of risk that is associated with the probability of failure of the geotechnical containment structures and the pond level control measures. As such, the risk of various failure modes needs to be assessed with input from both geotechnical and risk experts. Minimizing the depth of a water cover assists in reducing the risk of failure of containment structures. However, too shallow a water cover may allow tailings solids resuspension with wind and wave action.

Previously oxidized wastes can in some instances be disposed under a water cover, but only after the wastes have been well-characterized and the potential effects of the oxidation products on tailings pore waters, containment structure seepage, groundwater recharge, and impacts to the quality of the water cover and effluent treatment requirements have been thoroughly assessed. This is to allow measures to be planned in advance and taken to deal with the possible release of contaminated pore water and oxidation by-products to seepage, the water cover and subsequently to the environment. Oxidized wastes have successfully been disposed underwater through subaqueous disposal and by flooding – however such programs may require the treatment of excess water over the short term.

The literature related to water covers is now quite extensive. The overall cost for the MEND water covers program, both for fresh and oxidized mine wastes, totals about $3.5 million. This expenditure has resulted in excess of fifty reports and scientific papers. Key MEND reports, related to water covers, are listed in Table 4.2-1.

MEND 2.18.1 involved a comprehensive review of water cover applications and research with the objectives of documenting the design and performance of water covers used to prevent acid generation in sulphide-bearing mine wastes and compiling the results of water cover research completed in Canada, Norway and the U.S.. MEND 2.18.1 is recommended as a useful reference document for readers involved in the planning or review of a water cover program.

MEND 2.18.1 indicates that it is apparent from laboratory and field experiments that subaqueous disposal represents the most successful method presently known for preventing and controlling acidic drainage. However, in many cases the water covering the oxidized tailings may require treatment prior to discharge to meet regulatory standards. It is not currently possible to accurately predict either the nature or duration of treatment prior to actual deposition and observation.

A permanent water cover remains one of the most effective approaches that can be applied to provide a stable geochemical environment for most sulphide tailings. Additional enhancement is accomplished at some tailings ponds through the use of substrate covers over the submerged tailings using oxygen consuming materials to reinforce reducing conditions in the sediments, or the use of a non-acid generating sand layer to form a diffusion barrier. Research into diffusion barriers is ongoing.

Table 4.2-1
Projects on Water Covers

MEND Project	Title	Fresh/ Oxidized Tailings
2.11.1a	Subaqueous Disposal of Reactive Mine Wastes: An Overview	Fresh
2.11.1a Ben	A Preliminary Assessment of Subaqueous Tailings Disposal in Benson Lake, British Columbia	Fresh
2.11.1a And	A Preliminary Assessment of Subaqueous Tailings Disposal in Anderson Lake, Manitoba	Fresh
2.11.1a Man	A Preliminary Assessment of Subaqueous Tailings Disposal in Mandy Lake, Manitoba	Fresh
2.11.1b	Geochemical Assessment of Subaqueous Tailings Disposal in Buttle Lake, British Columbia	Fresh
2.11.1b And	Geochemical Assessment of Subaqueous Tailings Disposal in Anderson Lake, Snow Lake Area, Manitoba	Fresh
2.11.1b Man	Geochemical Assessment of Subaqueous Tailings Disposal in Mandy Lake, Flin Flon Area, Manitoba	Fresh
2.11.1c	A Preliminary Biological and Geological Assessment of Subaqueous Tailings Disposal in Benson Lake, British Columbia	Fresh
2.11.1c Keo	Chemical Diagenesis of Submerged Mine Tailings in Benson Lake and Natural Sediments in Keogh Lake, Vancouver Island, British Columbia	Fresh
2.11.1d	A Critical Review of MEND Studies Conducted to 1991 on Subaqueous Disposal of Tailings	-
2.11.1e	Review of MEND Studies on the Subaqueous Disposal of Tailings (1993-95)	-
2.11.2a	Literature Review Report: Possible Means of Evaluating the Biological Effects of Subaqueous Disposal of Mine Tailings	-
2.11.3abc	Geochemical Assessment of Subaqueous Tailings Disposal in Anderson Lake, Manitoba, 1993-1995 Study Program	Fresh
2.11.4a	Geochemical Assessment of Subaqueous Tailings Disposal in Buttle Lake, British Columbia, 1993 Study Program	Fresh
2.11.5ab	Shallow Water Covers – Equity Silver Base Information Physical Variables	Fresh
2.11.5c	Geochemical Assessment of the Equity Silver Tailings Pond	Fresh
2.11.9	Design Guide for the Subaqueous Disposal of Reactive Tailings in Constructed Impoundments	Fresh
2.12.1a*	Evaluation of Man-Made Subaqueous Disposal Option as a Method of Controlling Oxidation of Sulphide Minerals – Synopsis Report	Fresh
2.12.1b*	Evaluation of Man-Made Subaqueous Disposal Option as a Method of Controlling Oxidation of Sulphide Minerals (Phase 1) – Background and General Description	Fresh
2.12.1c*	Subaqueous Disposal of Reactive Mine Tailings – Louvicourt Mine Test Cells – Geochemical Sampling and Analysis – Final Report	Fresh
2.12.1d*	Reactivity Assessment and Subaqueous Oxidation Rate Modelling for Louvicourt Tailings	Fresh
2.12.1e*	Evaluation of Man-Made Subaqueous Disposal Option as a Method of Controlling Oxidation of Sulphide Minerals: Column Studies – Final Report	Fresh
2.13.1a*	Wet Barriers on Pyritic Uranium Tailings - Part I and II – Laboratory Lysimeter Studies of Oxidation and Limestone Neutralization Characteristics of Uranium Tailings and Waste Rock	Oxidized

Table 4.2-1 (Cont'd)
Projects on Water Covers

MEND Project	Title	Fresh/ Oxidized Tailings
2.13.1b*	Wet Barriers on Pyritic Uranium Tailings - Part III - Laboratory Diffusion Lysimeter Studies of Uranium Tailings Deposited Under a Shallow Water Cover.	Oxidized
2.13.2a	Expérimentation de l'inondation du parc à résidus miniers - Solbec Cupra. Rapport Sommaire,	Oxidized
2.13.2b	Flooding of a Mine Tailings Site - Solbec Cupra. Suspension of Solids: Impact and Prevention.	Oxidized
2.13.2c	Inondation artificielle du parc à résidus miniers Solbec-Cupra : Études microbiologique et chimique - Rapport final.	Oxidized
2.13.2d*	Suivi environnemental du parc à résidus miniers Solbec-Cupra – 1994-1997.	Oxidized
2.13.3a**	Preliminary Hydrogeochemical Investigation of the Falconbridge New Tailings Impoundment – Fall 1990 – Final Report.	Oxidized
2.13.3b**	WATAIL 1.0 User's Manual - A Tailings Basin Model to Evaluate Transient Water Quality of Acid Mine Drainage.	Oxidized
2.15.1a	Flooding of Pre-Oxidized Mine Tailings: Mattabi Case Study	Oxidized
2.15.b*	Laboratory Studies of Shallow Water Cover on Reactive Tailings and Waste Rock: Part 1- Oxidation and Leaching Characteristics	Oxidized
	Laboratory Studies of Shallow Water Cover on Reactive Tailings and Waste Rock: Part 2 - Roles of Oxygen Limiting Barriers and Water Flows	Oxidized
	Laboratory Studies of Shallow Water Cover on Reactive Tailings and Waste Rock: Part 3 - Mobility of Mine Tailings Under Wave Action	Oxidized
2.15.3	Laboratory Study of Particle Resuspension, Oxidation and Metal Release in Flooded Mine Tailings.	Fresh
2.18.1	Review of Water Cover Sites and Research Projects.	Both
Associate Project APC-1	Subaqueous Deposition of Tailings in the Strathcona Tailings Treatment System.	Oxidized
W.012	Water Covers to Prevent Acid Mine Drainage. Vancouver, British Columbia. 1996.	Both
IW.01	Water Covers and Underwater Disposal. Vancouver, British Columbia. 1997.	Both
ME.02	Case Studies on Wet and Dry Covers for Tailings and Waste Rock. Sudbury, Ontario. 1999.	Both
BC.02	Case Studies, Research Studies and Effects of Mining on Natural Water Bodies. Vancouver, British Columbia. 1999.	Fresh

* Not available at time of printing

** Open file report

Gallinger and Nicholson (1999) summarized the present state of affairs regarding the use of water covers. Firstly, water covers have been shown to be effective in controlling acid generation, and providing significant benefits in terms of improved water quality. Secondly, the planning and application of water covers at numerous sites have demonstrated that the key design considerations are geochemistry, hydrology, hydrogeology, and engineering design including sound geotechnical designs. As examples, these techniques can be applied to:

- Determine the optimum depth of water cover for a mine waste management facility;
- Predict water cover quality during and after waste disposal - either by subaqueous disposal or flooding;
- Monitor and assess the performance of waste management facilities with water covers; and
- Assess the risk associated with the use of a water cover and to manage the facility using an environmental management system.

The present state of knowledge is sufficient to allow for the responsible design, operation and closure of a waste management facility with a water cover. Knowledge in these areas is expected to increase and provide additional information respecting:

- Interactions due to pre-submergence supernatant (pore water) quality;
- Potential effects of localized mixing, tailings solids resuspension and hydrology interactions;
- Natural sedimentation/biological effects;
- Chemo/thermocline mixing effects; and
- Post-closure water management timing needs.

Price (1996) assessed information needs for underwater disposal and indicated that from a regulator's perspective, the following questions, issues and options need to be addressed by operators:

- What are the impacts of the predicted rates of release on the quality of the overlying water cover and on the off-site downstream receiving environment? Contaminant release indicates a concern, but is *not* in itself a problem. The significance of contaminant release will depend on the loadings and on the available dilution, attenuation and sensitivity of the receiving environment. Operators and researchers should always try to determine the "big picture" environmental significance of their predictions.
- Various methods of deposition, pond management and mitigation strategies can be used to avoid unacceptable impacts. These actions may also be the most efficient means of avoiding unacceptable uncertainty or research costs. All stages of research and planning should

include a consideration of cost effectiveness, and of the limitations and opportunities provided by pond management and mitigation.

- Contaminant release may occur through a variety of mechanisms. Potentially significant mechanisms, in addition to those studied to date, include: biological uptake through plant absorption and faunal ingestion, and oxidation of resuspended particles. Deposition, pond management and mitigation strategies may be used to avoid uncertain but potentially unacceptable impacts.
- Long-term ecological development will alter many of the features which presently contribute to or prevent contaminant migration into the overlying water and/or vegetation. One common process of potential importance is natural sedimentation. Other factors include changes in the overlying water quality and biological invasion. While the use of man-made impoundments avoids a short-term direct impact on natural watercourse, any impoundment will eventually become an integral part of the ecology, contributing habitat, food and drainage.
- Physical and compositional differences, potentially affecting contaminant release, result from the differential settling observed in most conventional tailings impoundments.

4.2.1 WATER COVERS OVER FRESH TAILINGS

Subaqueous tailings disposal is the placement of fresh sulphide-rich tailings under a water cover to prevent the oxidation of the sulphide minerals. The MEND 2.11 series of reports evaluated various Canadian tailings deposits to determine their geochemical stability, and to better understand the geochemical mechanisms that are active in preventing the leaching of trace metals from these deposits.

4.2.1.1 Discussion of Theory

The following discussion has been extracted from MEND 2.11.1a, b, c.

Finding an environmentally sound, yet cost effective, method for disposal of sulphide-containing mine wastes has been a challenge for decades. Given the critical role played by oxygen in the process of acid generation, thoughts towards abatement of this problem have focused around elimination of oxygen as a reactant. Consequently, arguments for subaqueous disposal arose from the premise that acid generation from sulphides could be suppressed when submerged underwater where oxygen concentrations are greatly diminished relative to the atmosphere. In other words, lowering the concentration of one of the principal reaction ingredients (oxygen) would lower the oxidation reaction rate, hence the rate of generation of acid and dissolved metals.

The efficiency of subaqueous disposal prior to the 1980's was largely unproven and supported by a few scientific studies. In order to address the paucity of relevant data, a suite of projects created through the BC ARD Task Force and MEND were designed, and involved field work in a series of lakes where mine tailings had been deposited. The projects utilized a variety of sampling, analytical and interpretative techniques designed to measure the reactivity and short and long-term chemical stability of subaqueous tailings deposits (MEND 2.11).

1. <u>Background Chemistry</u>

The instability or reactivity of metal sulphides arises from their mode of formation. Sulphides are formed in reducing environments (in the absence of oxygen). They are unstable and susceptible to chemical reaction in the oxygen-rich environment of the earth's surface. Accordingly, the most stable environment in which to store sulphide-rich mine tailings is one devoid of oxygen - one that mimics their environment of genesis.

Subaqueous systems are an effective first approximation of a stable environment for sulphides not because they are devoid of oxygen (indeed, subaqueous environments most often have measurable concentrations of dissolved oxygen), but rather because they contain low oxygen levels even in their most saturated state. The maximum concentration of dissolved oxygen found in natural waters is at a minimum 30 fold lower than that in the atmosphere. Because the rate of sulphide oxidation is in part dependent on the concentration of oxygen, it is readily apparent that the generation of acid and dissolved metals can be dramatically minimized underwater. More importantly, however, is the rate at which oxygen is replenished to the reaction site. Once the small inventory of dissolved oxygen in the water is consumed, it is typically replaced by processes of molecular diffusion and small-scale turbulence; the transfer of oxygen in water is nearly 10,000 times slower than similar transfers in air. Consequently, storage under permanent water cover is perhaps the single most effective measure that may be taken to inhibit acid generation from sulphidic tailings.

Sediments recreate an environment stable to sulphide minerals even more effectively than a water cover, in part because of the low concentrations of dissolved oxygen but more importantly because of a natural tendency for sediments to become chemically reducing. To understand why the sedimentary environment is an appropriate site for the storage of sulphidic mine tailings, it is first necessary to outline some of the natural chemical processes found in that environment.

Natural sediments typically contain a spectrum of components ranging from eroded rocks and soils of local origin to unique substances formed within the deposits. However, of all the components found in natural sediments, the remains of plants and animals (organic matter) are perhaps the most important as they are considered to be the fuel for almost all chemical reactions that occur within the deposits. This is because organic matter (like sulphides) is unstable in the presence of oxygen; it has a natural tendency to decompose into its constituent elements (mostly simple molecules containing the elements carbon, nitrogen, phosphorus, sulphur and hydrogen). In other words, organic matter consumes or reacts with the oxidant oxygen to form carbon dioxide and a suite of simple, biological by-products. This reaction is accelerated by a host of bacterial species which catalyse the reaction to derive energy for their own needs. Because the concentration of oxygen in natural waters is initially low, it is often rapidly depleted within the surface layers of sediments. When oxygen is no longer available to react with the organic matter, secondary oxidants are utilized in its place by the bacterial community. The oxidants are in order of preference: nitrate, Mn-oxide, Fe-oxide, sulphate and carbon dioxide; once one secondary oxidant is consumed (i.e., nitrate) the next most favoured is consumed (i.e., Mn-oxide) until all are exhausted. A by-product of the reaction between sulphate and organic matter (in the absence of more favourable oxidants) is hydrogen sulphide. The natural tendency in sediments is toward the creation of an environment in which sulphides may form naturally.

2. Methods of Examination

There are two principal ways in which to assess whether or not sulphidic mine tailings are reacting or releasing acid and metals to the subaqueous environment. The first is direct microscopic or petrographic observation of the submerged tailings particles. A far more sensitive, effective and elegant approach is to look for direct effects of sulphide oxidation such as a drop in pH, an increase in sulphate or the most direct indicator of all, an increase in dissolved metals in the pore water adjacent to the tailings solids. Since dissolved metals are the parameters of environmental concern and because they exist at very low concentrations naturally, measuring their distribution within sediment pore waters (the water surrounding the deposited sediment or tailings particles) yields a very sensitive indication of tailings reactivity as well as potential environmental impact.

The distribution of dissolved metals in pore waters has been determined by two proven approaches. Within the MEND projects, sampling of pore waters was accomplished utilizing sediment coring and the deployment of dialysis arrays (peepers). Sampling pore waters by core involves the collection of sediment with a specialized, light-weight, gravity corer. The pore waters are separated from the sediment solids by placing sequential slices of sediment into a centrifuge; the resulting fluid fraction is filtered and analysed for dissolved metals. Peepers sample pore waters much more passively. Peepers consist of an array of depressions or wells in a plexiglass plate. The wells are filled with ultra-pure water and covered with a filtration membrane. The peeper is inserted vertically into the sediments, usually by divers, and allowed to equilibrate within the sediments for 10 to 14 days. During that period, dissolved metals move across the membrane into the sample wells while solids are excluded. After 10 to 14 days the water within the sample wells is chemically indistinguishable from that of the adjacent pore waters. The peepers are retrieved and brought to the surface, transferred to the boat and immediately inserted into nitrogen-flushed storage boxes for transport.

To avoid oxidization of the samples through contact with the atmosphere, all sample handling of both cores and peepers after collection is carried out in nitrogen-filled, plastic glove bags. Once the pore waters have been filtered (again, under nitrogen), they are “preserved” for subsequent analysis by the addition of a small amount of ultra-pure acid.

3. Chemical Manifestations of Dissolved Metals in Pore Waters

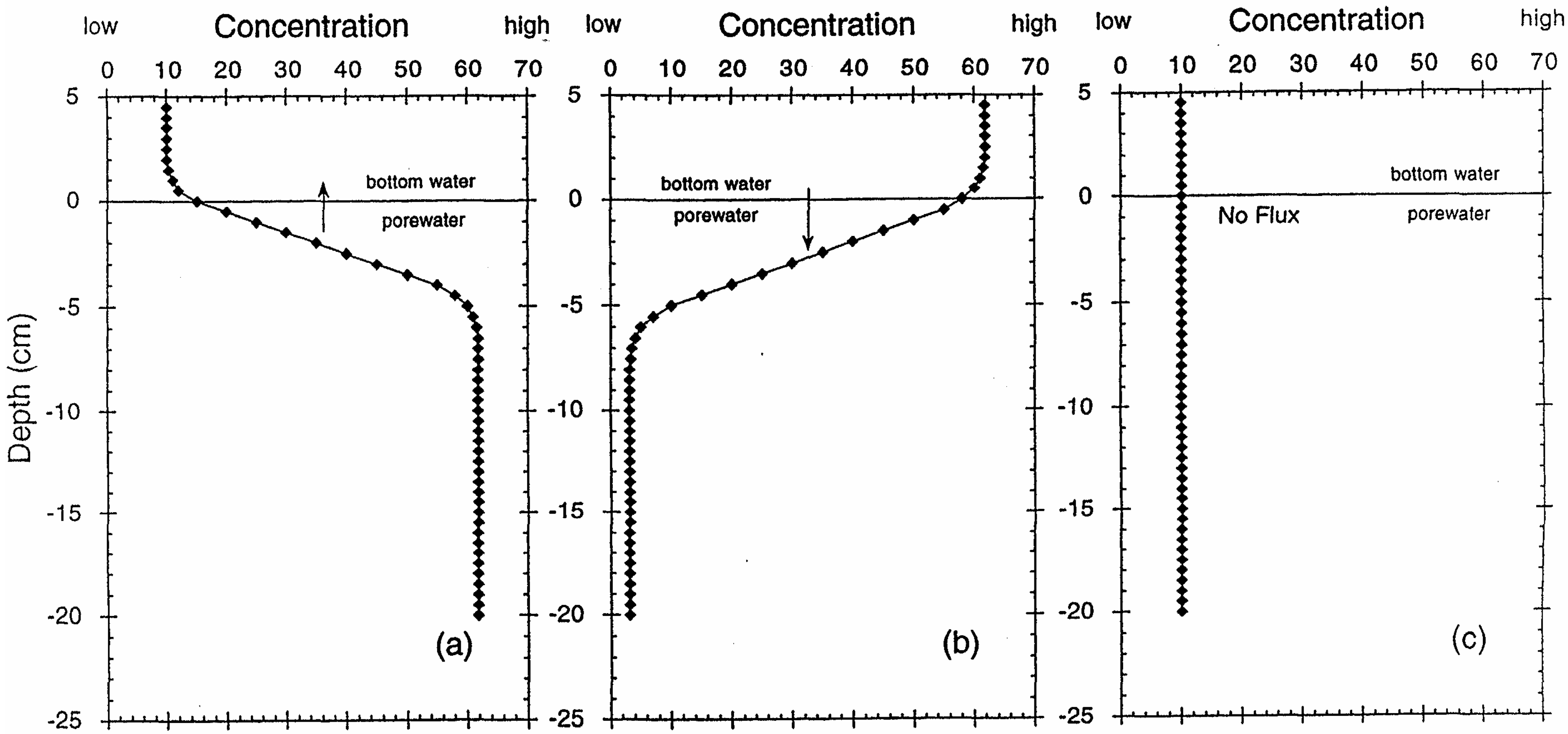
Upon their formation, sediment pore waters are no more than lake water trapped between sediment particles; in the absence of chemical reactions, the composition of pore waters would be identical to the overlying lake water (MEND 2.11.9). If tailings are reactive and release dissolved metals to the environment, the most sensitive manifestation will be locally elevated concentrations of dissolved metals within shallow pore waters (Figure 4.2-1(a)). Conversely, precipitation or consumption of dissolved metals within the tailings are characterized by concentrations that decrease with depth (Figure 4.2-1(b)).

The release or consumption of dissolved metals results in the formation of adjacent zones of differing concentrations. The difference in dissolved metal concentration between a high and a low define a concentration *gradient* and results in net migration of dissolved metals from the zone of high concentration to the zone of low concentration. In sediments, this process occurs through the random motion associated with all dissolved molecules and is termed molecular diffusion. The amount of dissolved metals that migrates down a concentration gradient (from high to low concentration) is termed the flux and is proportional to the steepness of the gradient. In other words, a greater flux (e.g., a greater transport of dissolved metals) occurs where a very high concentration is immediately adjacent to a very low concentration.

When a concentration gradient extends across the sediment water interface, metals can be said to be diffusing out of or into the sediments (to or from the water cover) depending on the direction of the gradient. Lower concentrations of dissolved metals in pore water relative to the overlying lake water indicates a flux of metals into the sediments from lake water (Figure 4.2-1(b)). Conversely, higher concentrations in pore waters than lake water infers a flux in the opposite direction (Figure 4.2-1(a)).

In the majority of the MEND projects undertaken metals have been observed to diffuse into the sediments from the overlying lake water. This has occurred in part because some of the lakes contained elevated concentrations of dissolved metals, but more importantly because of the natural tendency for sediments to create an environment stable to sulphides as discussed previously. When sulphate is utilized as a oxidant in the decomposition of organic matter within

Figure 4.2-1 **Three Hypothetical Concentration Profiles for Dissolved Metals in Tailings Deposits or Natural Sediments. a) release to pore solution; b) consumption by the deposits; c) zero net transport across the sediment-water interface**



<u>Note</u>: Arrows indicate the direction of diffusive transport of a dissolved metal along a concentration gradient
Source: MEND 2.11.9

the sediments, a natural by-product is hydrogen sulphide. Hydrogen sulphide is highly reactive with most dissolved metals (such as Cd, Cu, Hg, As, Mo, Ni, Fe, Pb, Zn and others) resulting in rapid precipitation of those metals as insoluble, solid metal sulphides. Because sulphate reduction (sulphide formation) typically occurs at shallow depths within sediments, there is a commensurate zone of localized metal consumption with the establishment of a dissolved metal concentration gradient from lake water into the sediments. The result is a flux or transport of dissolved metals into the surface sediments from the overlying lake water with the tailings acting as a sink for dissolved metals rather than a source. The concentration profile characteristic of such a case is shown in Figure 4.2-1(b).

In some instances, dissolved metals have been observed to be released from sediment solids to the pore waters. At first glance, this might suggest that the tailings are releasing dissolved metals to the overlying lake water, particularly if the concentration gradient extends to the sediment-water interface. However, there are several complicating factors that must be considered when such profiles are observed.

First, several metals (such as Cd, Cu, and Zn) are released to near-surface pore waters naturally as they are often associated with organic matter - they are not tailings-derived. As the organic material decomposes or oxidizes, those associated metals are released in dissolved form and may indeed migrate back into the overlying lake water. This most commonly occurs in sediments where oxygen has not been sufficiently depleted (or more specifically, where sulphide precipitation is absent). Such release is a natural phenomenon and accounts for much of the natural cycling of certain trace metals in many natural environments.

The second factor is that even though there may be some release of metals from tailings to pore waters in certain cases, a process referred to as *oxide blocking* or *oxide scavenging* can intercept much of the upward flux of those metals before the dissolved species cross the sediment water interface into the lake water. Such scavenging involves oxides of iron and manganese. Where dissolved oxygen is present, Fe and Mn oxides exist as solids whose surfaces strongly adsorb many trace metals. When they are utilized in subsurface sediments as secondary oxidants in the absence of oxygen, they revert to dissolved Fe and Mn creating concentration gradients. As dissolved Fe and Mn diffuse upward toward the sediment-water interface, they eventually encounter dissolved oxygen and revert back to their original solid, oxide form. Iron and manganese oxides are both efficient in adsorbing a broad range of dissolved metal ions. Thus, their continuous formation in the near-surface sediments results in the establishment of an effective “blocking mechanism” that inhibits dissolved metals from entering the water column.

One final barrier to *all* metal release from tailings within lake sediments is time. The burial of tailings by natural sediments or more recently deposited tailings occurs progressively with time and has a profound effect on the ability of even the most reactive substances to affect lake water quality. As the dominant transport mechanism of dissolved metals in sediments is diffusion, and because mass transport by diffusion is effective only over short distances (i.e. a few centimetres), accumulation of a relatively thin layer of sediments over an abandoned tailings deposit may be sufficient to isolate tailings chemically from the water column.

4. Summary

Based on the results of the MEND study on tailings in natural lakes, several generalizations can be drawn to both natural lakes and tailings ponds.

- The diminished concentration of oxygen dissolved in water is the single-most effective inhibitor to tailings oxidation; low concentrations of oxygen translate into low oxidation reaction rates. The presence of a permanent water cover not only minimizes the maximum concentration of oxygen to which the tailings may be exposed, but it also inhibits the rate at which that oxygen may be resupplied.
- Even though tailings ponds are typically impoverished in organic carbon, they still present conditions suitable to long-term storage of sulphide-rich material.
- Time itself is an effective component in allowing the establishment of a physical barrier, which prevents the release of metals to the overlying lake waters. The accumulation of a veneer of natural sediments (a few centimetres thick) effectively isolates the tailings. Subaqueous disposal is at worst a relative short-term risk that decreases with time to yield a stable, passive but effectively final control system.

As indicated above, the oxidation of sulphide solids in tailings submerged under a water cover is naturally attenuated by several orders of magnitude because of the considerable reduction in the rate of oxygen diffusion through the water and the reduced oxygen flux at the water/tailings interface. The reduced rate of oxidation can be further mitigated by the presence of organic sediments beneath and over the tailings. Benthic communities assist in the development of anoxic conditions just below the water cover/tailings interface.

In general, anoxic tailings conditions would be expected to remain as long as the water cover is maintained. Sulphate reducing conditions in the organic layers allow dissolved metals to be precipitated and in the process coprecipitate other metals. The process includes the consumption of SO_4 in the reaction between sulphate and natural organic matter and the formation of hydrogen sulphide. The solubilities of sulphide particles are very sparse and as such sulphide solids maintained underwater and in an anoxic condition would be expected to remain stable and hardly contribute to the metal concentrations of surrounding freshwater. However, metal concentrations arising from the dissolution of oxidation products can be significant and of concern. As such, it is preferable to place fresh sulphide solids under a water cover before they oxidize.

4.2.1.2 Discussion of MEND Research

International research has been carried out, and continues to be undertaken, to better understand acidic drainage and develop methods for prevention of this natural process. Two Canadian co-ordinated joint industry/government programs to address acidic drainage issues have been initiated since 1988: the National Mine Environment Neutral Drainage (MEND) Program and the British Columbia Acid Mine Drainage Task Force. The subaqueous tailings research project (MEND 2.11) was co-sponsored by these two programs.

Subaqueous tailings disposal is the placement of tailing under a water barrier which is proposed as an effective medium to prevent oxidation of the sulphide minerals. Similar methods may apply to waste rock but to date limited work has been conducted by researchers in this area. Expenditures for the MEND subaqueous disposal program total approximately $1,600,000. A total of 24 reports and/or scientific papers have been prepared on this project.

Peer reviews have reaffirmed that the application of this method to natural lakes requires the acquisition of comprehensive biological information, much of which would be site-specific, and which was not technically or financially feasible in this generic research project. Consequently, subsequent design work completed under MEND focussed specifically on the application of the results to future man-made tailings ponds and small headwater lakes. A generic design guide (MEND 2.11.9) has been developed for applying this technology.

Subaqueous studies conducted under the MEND program are summarized below along with selected findings. Additional information is given in Part III. Applications and Limitations.

MEND 2.11.x MEND Subaqueous Studies

MEND 2.11 assessed historic Canadian tailings deposits to determine their geochemical stability and better understand the geochemical mechanisms which are active in preventing metals leaching. The project consisted of three major phases that extended over several years (Robertson *et al.* 1997).

Phase 1: <u>Preliminary Studies to Scope Several Sites (1988 to 1992)</u>

Phase One consisted of a literature review of potential sites in Canada in 1988. Despite the difficulty of finding ideal, fully flooded tailings sites (without beached tailings), four lakes were selected and scoping studies were conducted between 1989 and 1991. These studies included sediment core sampling using the Pedersen corer, pore water samples, lake water quality, limited fish inventory and sampling for trace metal content. The work did not address the complex biological issues related to the deposition of reactive tailings in a natural lake.

Phase 2: Extensive Detailed Geochemistry Studies at Two Specific Sites (1992 to 1995)

Phase Two consisted of performing more intense studies on two lakes (Anderson Lake, Manitoba and Buttle Lake, British Columbia) to further examine the sediment geochemistry, water column chemistry, and exchange across the sediment/water interface. In following one of the recommendations of the first Peer Review (MEND 2.11.1d), the additional evaluations were conducted under different seasons to examine the variability of the sediment stability under different environmental conditions. Dialysis arrays (peepers) were employed for parts of these programs and an extremely rigorous QA/QC program was implemented for laboratory analysis.

Phase 3: Design Manual and Validation of Criteria (1995 to 1997)

Phase Three consisted of preparing generic design guidelines for the application of this technology to future mines. The first objective was to develop a technology that determines the governing physical criteria which maintain sediments in a stable condition under different wind and wave actions. The second objective was to outline the chemical criteria which maintain the metals in a chemically stable condition within sediments.

Limited validation studies to verify the theoretical physical criteria have been completed by work at the Equity Silver Mine tailings pond in B.C. (MEND 2.11.5ab and c). A design guide was completed in 1998 (MEND 2.11.9).

MEND 2.11.1a Subaqueous Disposal of Reactive Mine Wastes: An Overview

MEND research in water covers commenced with MEND 2.11.1a which comprised a literature review of the practice of subaqueous deposition with an emphasis on the physical, chemical, geochemical, microbiological, and macrobiological implications for freshwater environments. MEND 2.11.1a is a useful reference for the theoretical basis of subaqueous tailings disposal.

The following discussion regarding sulphide oxidation reaction products and the solubilities of sulphides and oxidation products is based on a more detailed discussion presented in MEND 2.11.1a. Sulphide reactions lead to diverse alternation products typically amorphous oxyhydroxides, and carbonates and sulphates - the latter two having much greater solubilities than the source sulphides. Table 4.2-2 lists solubility products for seven monosulphide minerals in freshwater. Equilibrium solubility constants for oxides and hydroxides, and carbonates and hydroxyl carbonates are also listed. As the sulphides and oxidation products have different reaction stoichiometries their solubility constants should not be directly compared. Rather it is preferable to compare the concentrations of free metal ions in solution which are in equilibrium with their solid phases.

Table 4.2-2

Solubility Products and Constants for Selected Solubility Equilibria

Sulphides	log K, 25ºC, I = 0
$MnS(s) = Mn^{2+} + S^{2-}$	-13.5
$FeS(s) = Fe^{2+} + S^{2-}$	-18.1
$ZnS(s) = Zn^{2+} + S^{2-}$	-24.7
$CdS(s) = Cd^{2+} + S^{2-}$	-27.0
$CuS(s) = Cu^{2+} + S^{2-}$	-36.1
$PbS(s) = Pb^{2+} + S^{2-}$	-27.5
$HgS(s) = Hg^{2+} + S^{2-}$	-52.7
Oxides and Hydroxides	
α-$FeOOH(s) + 3 H^+ = Fe^{3+} + 2 H_2O$	$*K_sO = 0.5$
(am) $FeOOH(s) + 3 H^+ = Fe^{3+} + 2 H_2O$	$*K^sO = 2.5$
$ZnO + 2 H^+ = Zn^{2+} + 2 H_2O$	$*K_sO = 11.14$
(am) $Zn(OH)_2 + 2 H^+ = Zn^{2+} + 2 H_2O$	$*K_sO = 12.45$
$CuO(s) + 2 H^+ = Cu^{2+} + H_2O$	$*K_sO = 7.65$
Carbonates and Hydroxide Carbonates	
$Zn(OH)_{1.2} (CO_3)_{0.4}(s) + 2 H^+ = Zn^{2+} + 1.6 H_2O + 0.4 CO_2(g)$	$*K_{\rho sO} = 9.8$
$ZnCO_3(s) + 2 H^+ = Zn^{2+} + H_2O + CO_2(g)$	$*K_{\rho sO} = 7.95$
$Cu(OH) (CO_3)_{0.5}(s) + 2 H^+ = Cu^{2+} + 3/2 H_2O + 1/2 CO_2(g)$	$*K_{\rho sO} = 6.49$
$PbCO_3(s) = Pb^{2+} + CO_3^{2-}$	$*K_{\rho O} = -13.1$
$CdCO_3(s) + 2 H^+ = Cd^{2+} + H_2O + CO_2$	$*K_{\rho sO} = 6.44$
$MnCO_3(s) = Mn^{2+} \quad CO_3^{2-}$	$*K_{\rho O} = -10.4$

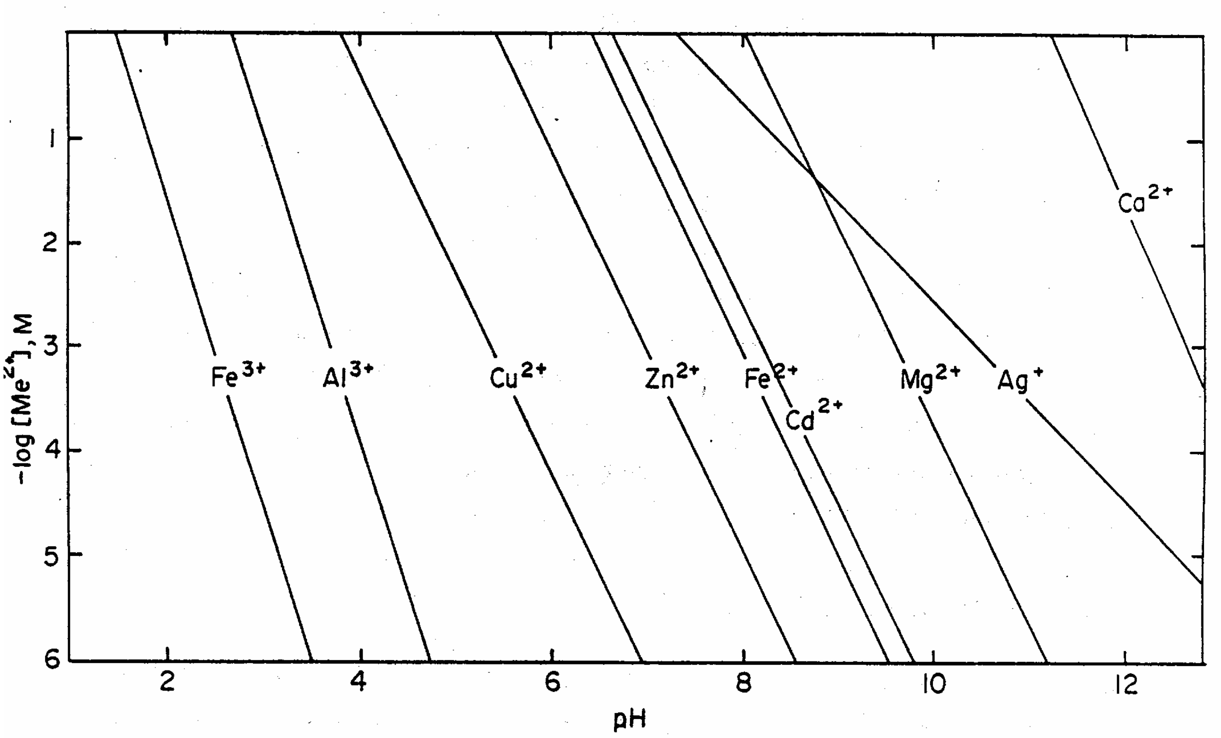
Note: Asterisked (*) constants represent the original terminology in the tables of constants published by L.G. Sillen and A.G. Martell, Stability Constants of Metal-Ion Complexes, Special Publications, Nos. 17 and 25, Chemical Society, London, 1964 and 1971.

Source: MEND 2.11.1a (after Stumm and Morgan 1981)

Figure 4.2-2 shows theoretical solubilities of oxides and hydroxides as a function of solution pH. Solubilities are represented as concentrations of free metal ions in and equilibrium with their respective oxide and hydroxide solid phases. The figure indicates that:

- Dissolved Fe^{3+} concentration would be expected to be very low at neutral range pH (e.g. in natural oxygenated surface waters); and
- A decrease in pH increases the solubility of all solid oxyhydroxide phases leading to increased metal ion concentrations. A one pH unit reduction generally leads to a two order-of-magnitude rise in dissolved metal concentration. Sulphide minerals are, in contrast, only sparingly soluble.

Figure 4.2-2 **Solubility of Selected Oxides and Hydroxides Plotted as Free Metal Ion Concentration in Equilibrium with Solid Oxides or Hydroxides versus pH**



Note: The plot is based on l = 0 and that the occurrence of hydroxo metal complexes has not been taken into account

Source: MEND 2.11.1a (after Stumm and Morgan 1981)

In a system where sulphide solids are naturally present under a water cover, sulphide oxidation would be expected to be inhibited as a result of:

- A significant reduction in atmospheric oxygen flux across the water/sediment interface;
- Relatively low levels of dissolved oxygen in the natural water cover - in comparison to the atmospheric level; and
- The continuous accumulation of organic and inorganic sediments. Bacterial activity could reduce the oxygen concentration in sediment pore water. This suggests that the layer of natural sediments which accumulate over submerged tailings can act as a protective barrier to oxidation.

MEND 2.11.1a suggests that the complexing capacity of natural sediments is relatively high due to the content of dissolved content of humic and fulvic acids in pore waters and available particle surfaces. Speciation in pore water and lacustrine water is a complex function of the type of inorganic and organic ligands and particulate matter present, and metal concentrations, pH, redox state, and flow dynamics.

MEND 2.11.1a notes that biological impact resulting from the submergence of mine wastes under fresh water will vary with the study organism, the physical and chemical nature of the waste, and the limnological characteristic of the water cover. In this area, the study concluded that:

- Macrobiological impacts need to be considered under the categories of: turbidity (affecting transparency and hence production, respiration, feed, etc.); sedimentation (with respect to the possible smothering of eggs and benthic organisms); toxicity (lethal, sub-lethal, and behaviour impacts of trace metals on biota); and potential contamination of the food chain; and
- The complex process of bioavailability of metals in lake bottom sediments and bioaccumulation in the freshwater food chain is an area that warrants additional investigation.

Geochemical studies have been undertaken on four natural and two constructed tailings impoundments across Canada including Anderson and Mandy Lakes in Manitoba, Benson and Buttle Lakes in British Columbia, Equity Silver in British Columbia, and Louvicourt in Québec. The subaqueous tailings deposits studied are summarized in Table 4.2-3 and a brief outline of the findings follows.

Table 4.2-3
Summary of Subaqueous Tailings Deposits Examined in MEND Studies

System Characteristics	Mandy Lake	Benson Lake	Buttle Lake	Anderson Lake	Equity Silver Tailings Pond	Louvicourt Tailings Pond
Mineral Mined	Cu, Zn	N/A	Cu, Pb, Zn	Cu, Pb, Zn	Au, Cu, Ag	Cu, Zn, Au, Ag
Tonnes of Tailings Deposited (t)	73,000	N/A	5,000,000	7,300,000+	35,000,000	Operating site
Type of System	Natural lake	Natural lake	Natural lake	Natural lake	Constructed pond	Constructed pond
Time of Tailings Deposition	Historical (1943-1944)	Historical (1962-1973)	Recent (1966-1984)	Recent (1979-1994; 1995-present)	Recent (1980-1994)	Ongoing (1994-present)
Maximum Water Depth	5.5 m	54 m	80 m	4 m	5 m	
Water Quality Rating	Good	Good	Moderate	Poor	Moderate	
Biological Productivity	Mid to high productivity	Low productivity	Low productivity	Mid to high productivity	Low productivity	Low productivity
Oxygen State in Water Column	Insufficient data	Oxic	Oxic	Seasonally anoxic	Unknown	
Trace Elements Studied	Cd, Co, Cu, Pb, Zn	Cd, Cu, Pb, Zn	As, Cd, Cu, Hg, Pb, Zn	As, Cd, Cu, Hg, Pb, Zn	As, Cd, Cn, Cu, Pb, Sb, Zn	
Metals Influx	Zn	Zn, Pb, Cd	None	Cd, Cu, Pb, Zn	Sb	
Metals Efflux	Cu, Pb	None	Cd, Cu, Pb, Zn	None	As	
Solid Tailings Characteristics	Sulphidic tailings	Sulphidic, pyrite-rich	Sulphidic tailings	Sulphidic, pyrite-rich	Sulphidic tailings	Sulphidic tailings
Supernatant Characteristics	Process water	Process water	Process water	Process Water	Neutralization sludges; CN-bearing process water; pressure oxidation leaching residues	Process water
Tailings Management	None	Flocculent added	Flocculent added	Lime addition	None	
Date of MEND Study	June 1990	Aug. 1991	Sept. 1989; Oct. 1993	Spring 1989;1991; April and Aug. 1993	Oct. 1995	Jan. 1994 – Mar. 1998
MEND Report	2.11.1a Man 2.11.1b Man	2.11.1a Ben 2.11.1c 2.11.1c Keo	2.11.1b 2.11.4b	2.11.1a And 2.11.1b And 2.11.3 abc	2.11.5ab 2.11.5c	2.12.1 (five reports)

Mandy Lake, Manitoba

Mandy Lake is located in central Manitoba near the Saskatchewan-Manitoba border, five kilometres south of Flin Flon. Originally a bay off the west side of the northwest arm of Schist Lake, Mandy Lake was enclosed when a causeway was built across the inlet to the Mandy Mine site. During its operations from 1943 to 1944, ore was milled on site while mine tailings were deposited directly into Mandy Lake. The tailings form a deposit that gradually slopes away from the east shore to a water depth of 1 m, before dropping off quickly into 5 m deep water. The tailings inputs consisted primarily of pyrite and appreciable quantities of zinc and copper. A study of Mandy Lake was conducted between June and September, 1990, with a focus on the geochemical behaviour of zinc, copper and lead in the tailings (MEND 2.11.1a Man, MEND 2.11.1b Man). Tailings cores were collected from submerged tailings in two locations and the pore waters extracted. It was found that tailings were widespread in the small lake and occurred in significant concentrations in the surface sediments (top 5 mm). This appeared to be the result of continuous slumping of the tailings fan.

Despite high concentrations in the solid phase, levels of zinc, copper and lead in pore waters were generally very low and decreased with depth in the deposit. Dissolved iron measurements made on pore water samples indicated that the tailings deposits were anoxic within several millimetres of the sediment water interface. A small benthic efflux of copper and lead was detected, but was regarded as negligible since it had no measurable impact on the dissolved metal inventory in the overlying lake water. Pore water profiles also indicated that dissolved zinc was diffusing into the deposits from the overlying water.

Benson and Keogh Lakes, British Columbia

The Coast Copper Mine discharged pyrite-rich mine tailings into Benson Lake on Vancouver Island, British Columbia, from 1962 to 1973. The lake consists of a single, tube-shaped basin, 2.2 km long, which reaches a maximum depth in the centre of approximately 54 m. The lake is fed from the eastern end by the Benson and Raging Rivers. The tailings in the central basin of the lake have been covered by a veneer of natural organic-rich sediments of dominantly terrestrial origins. Benson Lake is considered to be oligotrophic, and contains soft water (MEND 2.11.1a Ben, MEND 2.11.1c).

Keogh Lake was used as a control (reference) site for this study (MEND 2.11.1c Keo). The lake is located in an adjacent watershed and has never received any mine-related discharges. One sediment core from each lake was collected (August 1991 in Benson Lake; November 1991 in Keogh Lake). Both the extracted pore waters and the solid phase were analyzed for a suite of metals.

The Benson core consisted of about 30 cm of “natural” (but still copper-enriched) sediments overlying copper bearing sulphide-rich tailings, depleted in zinc and lead. The Keogh core consisted of primarily organic-rich natural deposits. A thin veneer of manganese and iron-oxyhydroxide-rich material mantled the sediments in both lakes. High dissolved iron concentrations in pore waters below 1.5 cm depth in Benson Lake, and at 1.5 cm depth in Keogh Lake, indicated that the sediments became suboxic to anoxic at very shallow depths in both basins. High-resolution profiles of dissolved cadmium, lead and zinc in pore waters of both lakes showed that the concentrations of these metals decreased from overlying bottom waters to the sediment. These data confirmed that at the time of sampling there was no efflux of these metals to the overlying water in either lake. Profiles in Benson Lake also showed that there was no copper efflux to the bottom waters, despite the fact that the “natural” sediments in Benson Lake contained higher concentrations of solid-phase copper than would be expected in a pristine basin.

Anderson Lake, Manitoba

Anderson Lake is located in northwest Manitoba near Snow Lake. The lake occupies a small (0.5 km wide by 6.3 km long), shallow (average depth 2.1 m) basin, which receives minimal inputs from runoff, swamp drainage and precipitation. The lake is mesotrophic to eutrophic, hosts natural sediments rich in organic matter and develops an anoxic water column during the ice-covered winter months. Tailings deposition into Anderson Lake commenced in 1979 when it began receiving inputs from the Snow Lake mill (Hudson Bay Mining and Smelting Co. Ltd.). The mill then processed copper-lead-zinc ores from seven local mines, producing tailings rich in sulphide and silicate gangue minerals. The water quality in the basin has been generally poor, and has suffered from sporadic episodes of depressed pH. To combat the acidic conditions that have occasionally developed, the mine periodically added NaOH solution to the central area of the basin. Between 1979 and 1994, tailings were deposited to the lake via a floating pipeline. Tailings discharge recommenced in 1995.

Geochemical assessments of Anderson Lake were performed on three occasions: summer 1990, winter 1993, and summer 1993 (MEND 2.11.1a And, MEND 2.11.1b And, MEND 2.11.3abc). Water column and sediment core samples (for analyses of solid-phase and pore water) were taken from two sites to characterize under-ice and mid-summer conditions in April and August, 1993, respectively. High concentrations of dissolved iron in the shallow pore waters indicated that the sediments become anoxic at shallow subsurface depths. There was no evidence of release of copper, cadmium or lead from the deposited tailings. In fact, decreases in metal (copper, cadmium, lead and zinc) concentrations across the sediment/water interface indicated that the sediments of Anderson Lake were acting as a sink rather than a source for these metals. Petrographic data from organic-rich natural sediments in Anderson Lake revealed significant framboidal pyrite formation, suggesting that precipitation of metal sulphides in the sediments was partially responsible for metal removal from the aqueous phase.

The studies concluded that the submerged tailings were not contributing to the high dissolved metal burdens in Anderson Lake. Rather, unbuffered discharges pumped from underground mine workings and acidic mine-site drainages appeared to represent the primary sources of metals and acidity to the basin. Of particular significance were acidic drainages from the northern shore of the lake where sulphide-rich wastes were used to build an access road in the late 1970's. The poor water quality in Anderson Lake was also attributed to the reductive dissolution of iron-oxides (and associated metals) during periods of water column anoxia. The large iron inventory in Anderson Lake appeared to be derived from ARD sources external to the lake.

The acid-generating roadway was excavated and removed from the northern periphery of the lake in summer and fall, 1995. Since that time, there has been a significant improvement in water quality in the lake despite the continued discharge of tailings. Dissolved zinc concentrations, for example, have decreased from ~ 350 µg/L (parts per billion) in the autumn of 1995 to < 100 µg/L by July 1997, while the pH averaged 7.6 through the period January to July 1997.

Buttle Lake, British Columbia

Buttle Lake is located in Strathcona National Park on Vancouver Island, British Columbia, and was used by Westmin Resources from 1966 to 1984 to dispose pyrite-rich tailings. The tailings deposit in Buttle Lake is restricted to the South Basin. Buttle Lake is large (35 km by 1 km) and deep (80 m deep), and is considered oligotrophic due to low nutrient levels. Despite its monomictic nature, Buttle Lake is able to maintain a significant inventory of dissolved oxygen throughout the water-column year-round. Buttle Lake water is soft and contains low concentrations of major ions.

Preliminary studies were conducted in late 1989 and early 1990 (MEND 2.11.1b), which consisted of taking sediment cores and water quality samples at four stations within the lake. A second, more intensive geochemical survey was conducted in October 1993 and involved establishing three stations to sample the water column, sediments and associated pore waters (MEND 2.11.4a).

The tailings, as of 1993, were covered by approximately 4 cm of oxic, natural sediments with a small proportion of admixed tailings (presumably introduced through bioturbation and slumping). Pore water data collected from natural sediments, indicated influxes of dissolved cadmium and zinc into the sediments. Pore water data from the two tailings sites suggested that cadmium, copper, lead and zinc were remobilized via the reductive dissolution of iron and manganese oxides at depth. Effluxes of these metals to the overlying water column appeared to be attenuated due to their reprecipitation with iron and manganese oxides in the surface oxic zone. Very little evidence, if any, supports the postulation that oxidation of sulphide particles was releasing metals to pore waters. Indeed, nitrate and sulphate concentrations decreased with depth at all sites, indicating that oxygen did not penetrate more than two or three centimetres into the sediments.

Equity Silver Tailings Pond, British Columbia

Placer Dome's Equity Silver Mine operated from August 1980 to January 1994. In this period, tailings were discharged to the Equity Silver tailings pond via a mobile floating platform. The tailings pond has an area of approximately 1.2 km^2 and a maximum water depth of 5 m in the central portion of the pond. Tailings consist predominantly of arsenopyrite, pyrite and pyrrhotite minerals. The neutralized sludge from an ARD treatment plant was also co-disposed with the tailings.

The subaqueous disposal project concluded with the development of a design document (MEND 2.11.9) which outlines basic guidelines and direction on where and how to apply this tailings disposal method. The field verification of this approach was completed at Equity Silver (MEND 2.11.5ab). It was found that the most important controllable variable at Equity was the water depth.

A geochemical survey was conducted at the Equity tailings pond in October 1995 (MEND 2.11.5c). Replicate peepers were used to collect pore waters from a shallow (1 m) and a deep (5 m) site within the tailings pond. Ancillary solid-phase and water column data were also collected. The distribution of most elements studied was indicative of small-scale lateral inhomogeneity. The data suggested that dissolved copper was neither released nor consumed by the tailings, while arsenic was released from pore waters to pond waters at both sites via the dissolution apparently of As bearing iron oxyhydroxide phases and an unidentified solid phase or phases. Conversely, antimony was being consumed rapidly within the surficial deposits, presumably by adsorption to an existing solid-phase. Low (< 0.08 mg/L) cyanide levels were detected in tailings pore waters.

Tailings Resuspension (Laboratory)

MEND 2.15.3 investigated the effect of tailings resuspension within a water cover and the oxidation of the suspended sulphide particles. The study involved a series of laboratory experiments in which columns loaded with unoxidized pyrrhotite tailings were flooded with 45, 60 and 80 cm deep water covers. The water covers were stirred at speeds ranging from 140 to 200 rpm and then monitored for pH, dissolved oxygen, conductivity, sulphate and metals concentrations. The suspended tailings, surficial submerged tailings, and undisturbed tailings solids and pore water were sampled and analyzed at the completion of the 126-day experiment. The resuspension of the tailings increased sulphide particle oxidation in comparison to a control column where the water covered tailings were not stirred. In the study, the tailings solids became suspended when the shear stresses created on the surface of the submerged tailings exceeded a critical shear stress. The researchers noted that as silicates tend to have a lower specific gravity in comparison to sulphide minerals, an experimentally derived critical shear stress value is conservative.

4.2.1.3 Application and Limitations

This section reviews the MEND Design Guide for the Subaqueous Disposal of Reactive Tailings in Constructed Impoundments, and examples of the application of water covers. The order of the subjects reviewed is shown in Table 4.2-4.

Table 4.2-4
Subjects Covered under Application and Limitations: Water Covers over Fresh Tailings and Waste Rock

Classification	MEND Report	Application	Page No.
Design Guide	2.11.9	Design guide for subaqueous disposal in engineered impoundments.	4-34
Subaqueous Disposal of Fresh Tailings and Waste Rock	2.11	Geochemical assessments of subaqueous tailings disposal in Benson and Buttle Lakes, British Columbia and Anderson and Mandy Lakes, Manitoba. Chemical diagenesis of submerged tailings.	4-44
	2.12.1a to e	Geochemical assessment (including laboratory and field investigations) of subaqueous disposal at the Louvicourt tailings area in Val d'Or, Québec (5 reports).	4-58
	-	Geochemical assessment of subaqueous disposal for the Voisey's Bay Project, Labrador.	4-66
	-	Geochemical assessment of proposed subaqueous disposal in Panama (Petaquilla Project).	4-71
	-	Subaqueous disposal of fresh waste rock - Eskay Creek	4-73

<u>MEND 2.11.9 Design Guide for the Subaqueous Disposal of Reactive Tailings in Constructed Impoundments</u>

MEND 2.11.9 is intended to assist users in making a preliminary evaluation of the applicability of the subaqueous disposal of sulphide-rich tailings in constructed, permanently-flooded tailings impoundments. The guide focuses on the prevention of acidic drainage, and inhibiting the potential migration of dissolved metals from tailings pore waters into water overlying the tailings. The immobilization of some specific metals may not, however, always be possible as metals such as arsenic and antimony have a unique, well-documented natural cycle that influences their movement into and from tailings sediments. As such, at some sites, the water discharged to maintain the water balance will require treatment for elements such as As and Sb.

The Design Guide is directed at a range of audiences including regulatory agencies in the developed and developing world, mining companies, environmental consultants, non-governmental organizations and the general public. Given that some of the groups are not highly experienced in technical issues related to tailings systems, geotechnical aspects, hydraulic design and risk assessment are discussed in general. The Design Guide could serve as a useful reference document for a mine proponent as a means of communicating the general basis of a subaqueous tailings disposal program to interested stakeholders. The guide indicates that:

- Subaqueous tailings disposal is not a universally applicable single solution; and
- Some tailings that include sulphide minerals may not have reactivities that warrant the use of permanently-flooded impoundments.

Sections of the Design Guide are summarized below.

<u>Design Guide Part I - ARD Geochemistry</u>

Part I summarizes the basic metal sulphide geochemical reactions that occur within tailings as revealed by MEND projects carried out from 1989 to 1996.

<u>Design Guide Part II - Data Collection and Database Development</u>

Part II addresses the need to collect quality data and develop an Integrated Chemical and Physical Database (ICPD) for use in the design, operations and closure of tailings systems. Typical data requirements are shown in Table 4.2-5.

Readers are reminded that Volumes 2.0 and 6.0 of this Manual deal with data collection and data management aspects.

<u>Design Guide Part III - Chemical Considerations for Tailings Pond Design</u>

Part III identifies and describes the range of chemical design data used to characterize tailings and input into an Integrated Chemical and Physical Database (ICPD). The use of a contaminant mass balance as a means of assessing each inflow to the tailings pond and developing a strategy to achieve a chemically stable tailings system is reviewed.

Part III provides an informative discussion on the benefits of mineralogical studies, and methods that are used to assess ore, waste rock and tailings for characterization purposes. Milling and metallurgical process are discussed in general terms, and important items to keep in mind when assessing subaqueous disposal vis-à-vis the milling/metallurgical process are identified. The chemical treatment of tailings pond inputs, such as ARD, is briefly reviewed. Other potential

Table 4.2-5

Types of Information Acquired for Tailings Pond Design and Operating and Closure Plans

Consideration	Type of Information Acquired
Water balance, tailings grain size, climate	• Minimum water depth • Pond volume • Pond residence time • Climatic range • Hydrology/hydrogeology
Ore and waste rock mineralogy	• Mineral abundance (e.g., sulphides, carbonates, sulphates) • Presence of reactive minerals (e.g., pyrrhotite) • Elemental abundance (e.g., % sulphur, metals) • Acid-generating potential
Tailings solids	• Tailings throughput • Elemental abundance • Percent solids • Types and abundances of secondary minerals • Depositional strategies • Mass loading to pond
Tailings supernatant	• Flow rate • Concentrations of dissolved metals (e.g., Cu, Pb, Zn) • Concentrations of CN, CN-complexes, thiocyanate, thiosalts, sulphate, ammonia • pH • Depositional strategies • Mass loading to pond
Treatment methods/by-products	• Treated effluent flow rates • Effluent composition (metals, pH, sulphate, cyanide, thiocyanate, chloride, etc.) • Hydroxide sludge composition (e.g., mineralogy, metal concentrations) • Ratio of sludge to tailings • Depositional strategies • Mass loading to pond
Other inputs: mine/pit waters, waste rock seepages	• Flow rates • Concentrations of dissolved metals, sulphate, pH • Mass loading to pond
Stability testwork	• Kinetics of the subaerial oxidation of ore, waste rock and tailings • Oxidative and non-oxidative subaqueous reactivity (long and short-term) • Hydroxide sludge stability (dissolution kinetics, problematic elements) • Depositional strategies
Structural integrity	• Geotechnical stability of foundations • Earthquake risks • Storm event criteria
Site selection	• Environmental impact • Natural hazards • Economics: cost and impacts

Source: MEND 2.11.9; MAC 1998

inputs to a tailings pond (i.e. mine site drainages, mine/pit water, natural runoff) and their possible influences on tailings ponds are reviewed.

Design Guide Part IV - Physical Considerations for Tailings Pond Design

Part IV identifies and describes the range of input data for an ICPD. It also defines the water cover design objective, which is to provide an adequate depth of water to ensure that the consolidate bed of tailings is not entrained or remobilized. The physical factors affecting submerged tailings bed stability under a body of water are reviewed as are sediment properties and their influence on the entrainment process.

Part IV presents an instructive Design Logic Chart to assist in the design of a tailings impoundment with a water cover. The Design Logic Chart is reproduced in Figure 4.2-3 and discussed in detail in Part IV of MEND 2.11.9.

Atkins *et al.* (1997) reviewed the factors associated with a water cover where the depth of the water cover is as shallow as possible yet adequate to inhibit acid generation in submerged tailings. The use of the shallowest possible depth of water is of key interest as it reduces the volume of water impounded. An adequate depth of water cover must, however, be sufficient to preclude the physical disturbance of the submerged tailings bed by wave action. The minimum depth of a water cover is dependent upon:

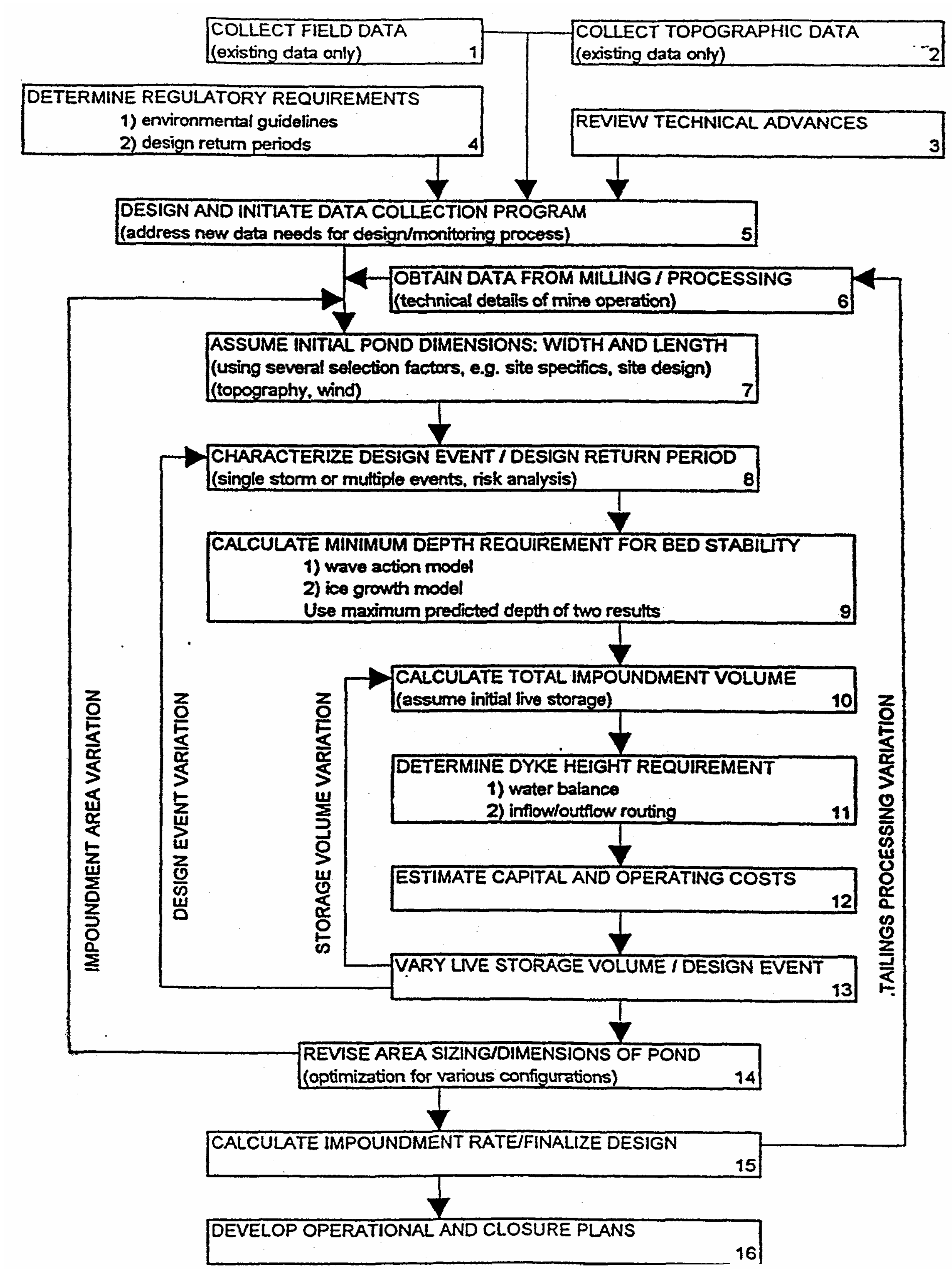
- The design wind event and return period;
- The size of waves generated by the design wind event; and
- The depth of water to which the wind-generated waves mobilize tailings at the surface of the tailings bed.

The key physical processes that affect the stability of submerged tailings as identified by MEND 2.11.9 are summarized in Table 4.2-6.

Table 4.2-6
Key Processes Affecting Submerged Tailings Bed

Process	**Comment**
Wave Action	Sufficiently large waves can generate velocities along the bed that can mobilize the bed material.
Ice Entrainment	The ice layer can ground on the bed and bond bed material by freezing.
Other Processes	• Seiching. • Seasonal turnover. • Currents.

Figure 4.2-3 **Tailings Pond Design Logic Chart**



Source: Atkins *et al.* 1997

MEND 2.11.9 reports that wave entrainment of sediments occurs when waves are large enough that the velocities associated with the passage of the wave are strong enough to move material at the bed as shown in Figure 4.2-4. Sediments that are entrained by wave motion may be lifted into the water column. Laboratory experiments suggest that when tailings are resuspended, whether by mechanical stirring or by intense wind and wave activity, sulphide oxidation is accelerated, the consequent precipitation of secondary iron oxyhydroxide minerals can adsorb or scavenge trace metals released during the primary oxidation reactions (MEND 2.15.3).

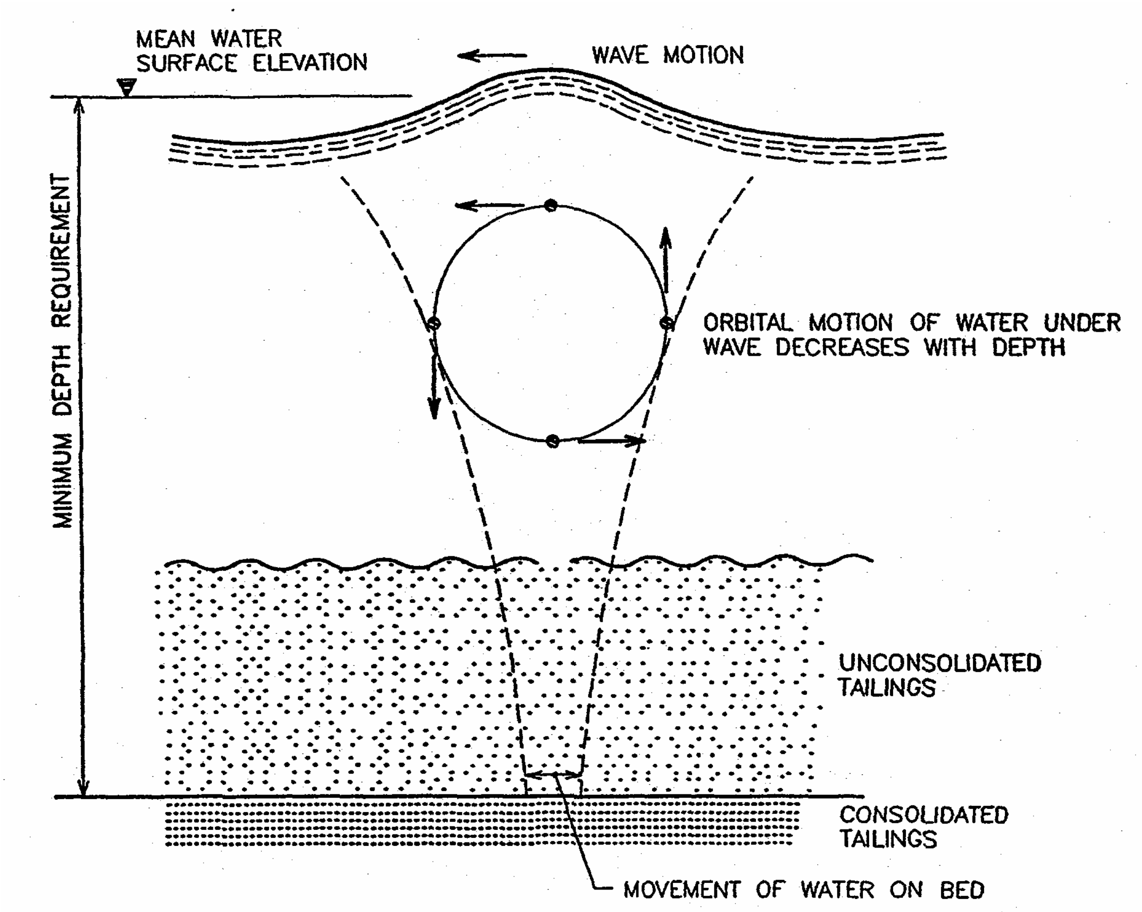
A general water cover design chart that indicates how shallow water cover design relates to the overall design of a tailings pond was developed by Atkins *et al.* (1997) and is reproduced in Figure 4.2-3. The design steps referenced to in the figure are described in Table 4.2-7.

A minimum depth versus windspeed curve needs to be developed based on site-specific parameters. A minimum predicted water depth curve was developed by Atkins *et al.* (1997) for a range of windspeeds for the Equity tailings impoundment. The key controllable variable at the Equity site is water depth. A minimum predicted water depth curve was developed for a range of windspeeds (Figure 4.2-5). For Equity, the observed disappearance of bed forms at depths of between 1.3 to 1.4 m, and the prediction of 1.4 m water cover depth for historic wind conditions provided confidence in the prediction equation (MEND 2.11.5ab). This study provided data to verify the prediction equations but more case histories are needed, under a wide range of climate, pond and wind conditions, to complete the calibration of the model.

<u>Design Guide Part V - Operational Monitoring of Tailings Ponds</u>

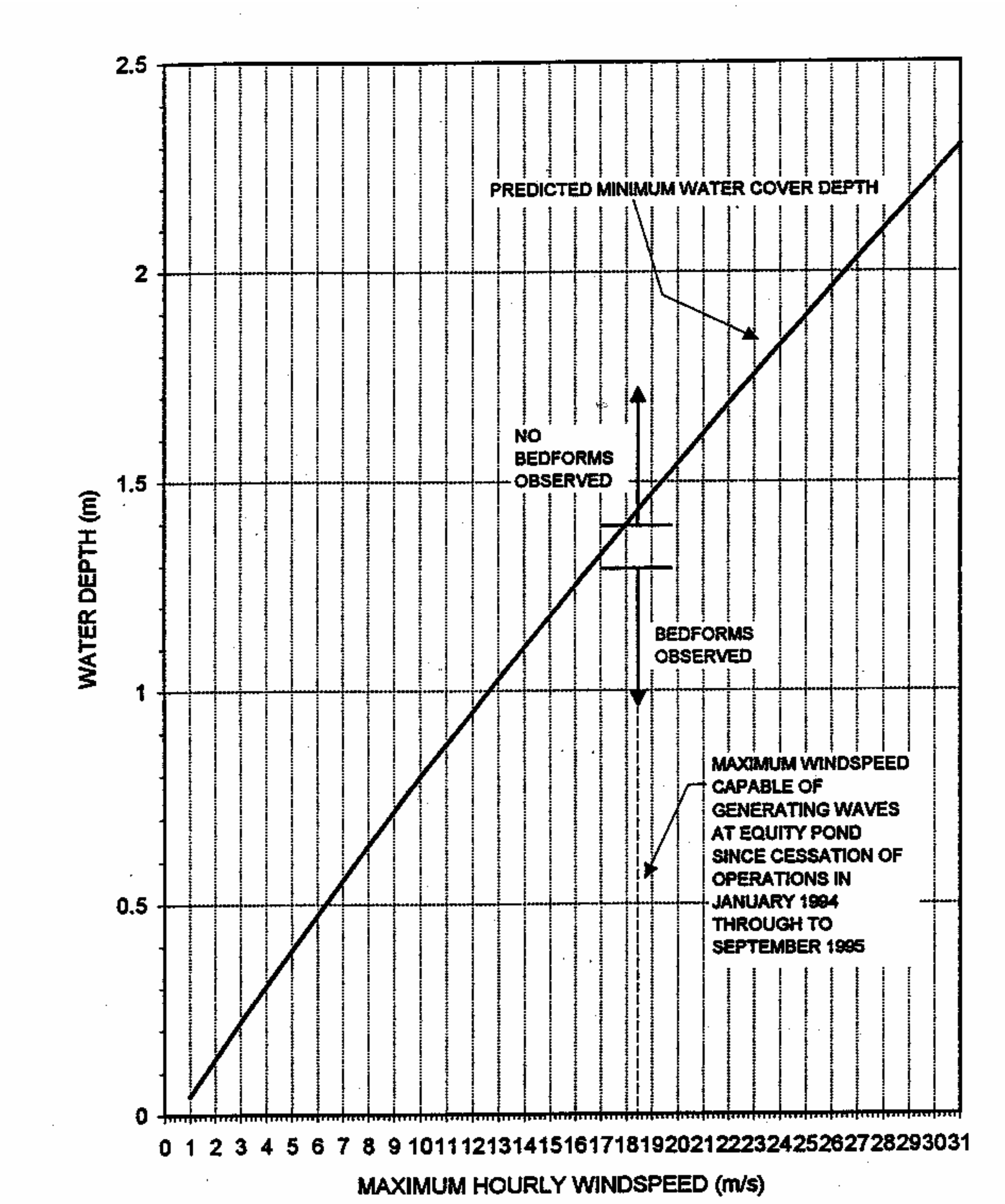
Part V defines a comprehensive set of operational monitoring requirements which are needed to obtain and maintain a continuous and updated working knowledge of the operational status of tailings ponds. This information is essential for updating the ICPD database which was used as the basis for the original design. As an essential management tool for mine operators, the database will have many applications including the validation of original design parameters, contributing to informed adjustments to operating systems and, most important, contributing to closure planning.

Figure 4.2-4 **Wave Interaction**



Source: MEND 2.11.9

Figure 4.2-5 **Predicted Minimum Depths versus Observations for Equity Silver Tailings Pond**



Source: Atkins *et al.* 1997

Table 4.2-7
Shallow Water Cover General Design Sequence

Step No.	Activity	Comment
1	Collect field data	• Collect data relevant to tailings bed stability and tailings pond design including climatic, hydrologic, and sedimentologic variables. • The two principle factors that affect the stability of sediments underwater are: 1. Entrainment forces on sediment surfaces by physical processes (e.g. moving water). 2. Inertial forces reflected in sediment properties (e.g. density) assist in resisting sediment transport. Sediment is transported when entrainment forces exceed inertial forces.
2	Collect topographic data	• Obtain site topography/survey data.
3	Review technical advances	• Review advances and technical improvements in scientific knowledge, construction methods and environmental considerations.
4	Determine regulatory requirements	• Review applicable regulations and objectives. • Regulations are likely to include water quality objectives for the discharge of excess water, and meteorological design event criteria.
5	Design and initiate data collection programs	• Fill in data gaps.
6	Obtain milling/ processing data	• Data includes tailings throughput (tpd), process water discharge to the tailings pond (m^3/s), tailings bulk density (t/m^3), tailings grains size distribution, and other flows to the tailings pond (m^3/s).
7	Assume pond dimensions	• To initiate planning, estimate initial pond dimensions (length and width).
8	Characterize design events and return periods	• Characterize design events and return periods for parameters such as winds, waves, runoff and precipitation.
9	Calculate minimum depth requirement for tailings bed stability	• Calculate the minimum depth of water cover for tailings bed stability. [Relevant equations are provided in the section following this table.]
10	Calculate the total impoundment volume	• Calculate the live storage volume of the water cover.
11	Determine water containment dyke height requirement	• Calculate the required height of containment structures including the appropriate freeboard.
12	Estimate capital and operating costs	• Prepare a benchmark estimate of capital and operating costs.
13	Vary design event/live storage	• Examine effects of various design events on the impoundment and costs.
14	Revise pond size and dimensions	• Examine sensitivity of pond design to changes in design variables based on 13.
15	Reassess mill discharge rate and finalize pond design	• Consider variations in milling rate, grind, etc. commencing at Step 6.
16	Operational and closure plans	• Continue to collect data during the operational phase and apply findings to refine the closure plan for the tailings area.

The generic minimum depth (d_{min}) of water cover required to maintain a physically stable submerged bed can be calculated as follows:

$$d_{\min} = 1.58 \; x \; 10^{-3} \; g \; (U_w F)^{\frac{2}{3}} \ln\left[5.14 \; x \; 10^{-2} \left(\frac{R(FU_w^4)}{V_v} \right) \right]$$

where: g = gravity (m/sec^2),

F = fetch length (m),

U_w = windspeed (m/sec),

R = wave height coefficient (for H_{sig}, R = 1). The significant wave height (H_{sig}) corresponds to the average height of the highest one third of all waves in the wave spectrum (MEND 2.11.9). The weight height expresses the energy carried by the wave form that impacts on the shore of an impoundment and influences bed stability.

V_b = critical bed velocity (m/sec).

The purpose of operational monitoring is to refine management strategies established during the design phase, and ultimately, to develop the final closure scenario. Monitoring entails examination of several components of the mining operation including: the flow and composition of various tailings pond inputs; pond water balance; pond environment (e.g., water quality, groundwater); and geotechnical stability. Supplementing the IPCD database with operational data should be an ongoing process whereby management strategies continually evolve. This process requires proactive management and should obviate the need for expensive remediation measures. Collectively, the operational data can be used to:

- Verify initial design assumptions (e.g., water balance, pond water quality, mass loadings);
- Update closure design;
- Fine tune operations and optimize operating costs;
- Trouble-shoot water quality issues which may arise; and
- Meet regulatory requirements.

<u>Design Guide Part VI - Closure Management and Other Closure Options:</u>

The goal of subaqueous disposal is to achieve a stable closure scenario - with an emphasis on the long-term physical and chemical stability of the tailings solids. Closure plans developed in the design stage should be continually refined throughout the operations stage so that a comprehensive plan is in place at shutdown. Closure planning aspects that need to be considered include:

- The post-closure water balance - it is likely to be different than during operations;

- Post-closure flows to the tailings pond and their effect on the pond water balance and water quality;
- Post-closure pond water quality and treatment requirements;
- Impacts to groundwater detected during operations should be monitored to verify closure predictions and validate mitigative measures; and
- Post-closure geotechnical monitoring programs need to continue to ensure that systems function as designed (e.g. containment structures). Qualified technical specialists should be engaged to assist in this area.

Geochemical Assessments of Subaqueous Tailings Disposal in B.C. and Manitoba

Pedersen *et al.* (1997a) and MEND 2.11 evaluated the post-depositional chemical behaviour of submerged mine tailings in two lakes and a tailings pond. Their results indicate that there are three modes of behaviour for submerged tailings.

- Mode 1, where tailings that do not release metals to the water cover. This mode was studied at Anderson Lake, Manitoba.
- Mode 2, where tailings that remobilize some metals to the water cover. This mode was studied at Buttle Lake, British Columbia.
- Mode 3, where tailings that release certain metals (e.g. As) apparently from soluble alteration productions formed during mineral processing. This mode was studied at the Equity Silver tailings pond in British Columbia.

Pedersen *et al.* (1997a) report that not all subaqueous disposal operations resulted in chemically benign behaviour. However, at operations where water cover contamination has been a problem, the cause has not been due to sulphide oxidation. As an example, Pb and Zn contamination of coastal waters near the Black Angel Mine in Agfaarlikavsaa Fjord, Greenland was traced to the dissolution of soluble Pb and Zn salts likely produced during mining and milling. Modifications to the milling procedures, the outfall design, and an increase in the extent of tailings flocculation markedly reduced the amounts of Pb and Zn released (Asmund *et al*. 1992; Pedersen *et al.* 1997a).

The following discussion of the research at Anderson Lake, Buttle Lake and Equity Silver is reiterated from Pedersen *et al.* (1997a).

Anderson Lake

Anderson Lake is located about two kilometres south of Snow Lake, Manitoba, and is a horseshoe-shaped 6 km long, shallow, mesotrophic to eutrophic typical Precambrian Shield lake with little outflow and a maximum depth of about 8 m. About 9 x 10^6 tonnes of mill tailings from the processing of copper-lead-zinc massive sulphide ores were deposited in the lake between 1979 and 1994, when mining temporarily ceased (MEND 2.11.4a). The pyrite-rich tailings consist of mainly silt-sized pyrite, quartz and feldspar, with accessory amounts of pyrrhotite, galena, chalcopyrite and sphalerite.

Water quality in the lake during recent years was poor as a result of an influx of acid rock drainage derived from an old roadway built of waste rock and tailings. Acidic drainage from a nearby abandoned mine was also pumped to the lake. During the decade prior to 1995, the outflow from the lake contained roughly 800 mg/L of SO_4^{2-} and 0.6 mg/L of dissolved Zn.

Tailings pore water and sediment samples were collected in Anderson Lake in 1993. The pore water samples were collected using peepers. Based on detailed measurements made on pore water samples, Pedersen *et al.* (1993) and MEND 2.11.3abc concluded that the submerged deposited tailings in Anderson Lake were not releasing metals to the overlying water column. Indeed, the opposite appeared to be true: sharp decreases with depth in the concentrations of dissolved Zn, Cu, Pb, and Cd in the upper few centimetres of the deposits were interpreted as representing uptake of metals from the contaminated overlying lake water, apparently via precipitation of authigenic sulphide phases at shallow depths.

Although the data collected in the 1990 study were consistent in indicating an influx of metals into the tailings, no samples had been collected from barely submerged deposits where wind-driven turbulence in summer and associated resuspension of the tailings could have promoted oxidation of sulphide particles and consequent metal release. This is the extreme case for subaqueous disposal. To study this specific setting, duplicate peepers were emplaced in August 1993 at a well-oxygenated, 1 to 1.5 m deep, highly turbid site where the surface sediments consisted of a lag deposit of pyrite-rich indurated clasts composed entirely of tailings particles. All pyrite particles, either discrete or within the clasts, appeared fresh and unaltered. A 2 to 3 mm thick rust-to orange-coloured ferruginous veneer, consisting of filamentous bacteria in a matrix of rust-to brown-coloured flocculant material, capped the sediments in a core collected at this location (Pedersen *et al*. 1997a).

Duplicate peeper profiles of dissolved Fe, Cu, Zn and Pb (Figure 4.2-6) illustrate that two principal diagenetic phenomena were active in these shallow, reworked deposits during summer, 1993.

1. The release of very high concentrations of iron to solution at shallow sub-bottom depths can be attributed to the dissolution of iron oxyhydroxides. This arises as a consequence of the well-known sequence of consumption of oxidants in sediments where organic matter is degraded by bacteria (Froelich *et al*. 1979). The bacterial use as electron acceptors, in general order, O_2, NO_3^-, manganese and iron oxyhydroxides and SO_4^{2-}. According to this

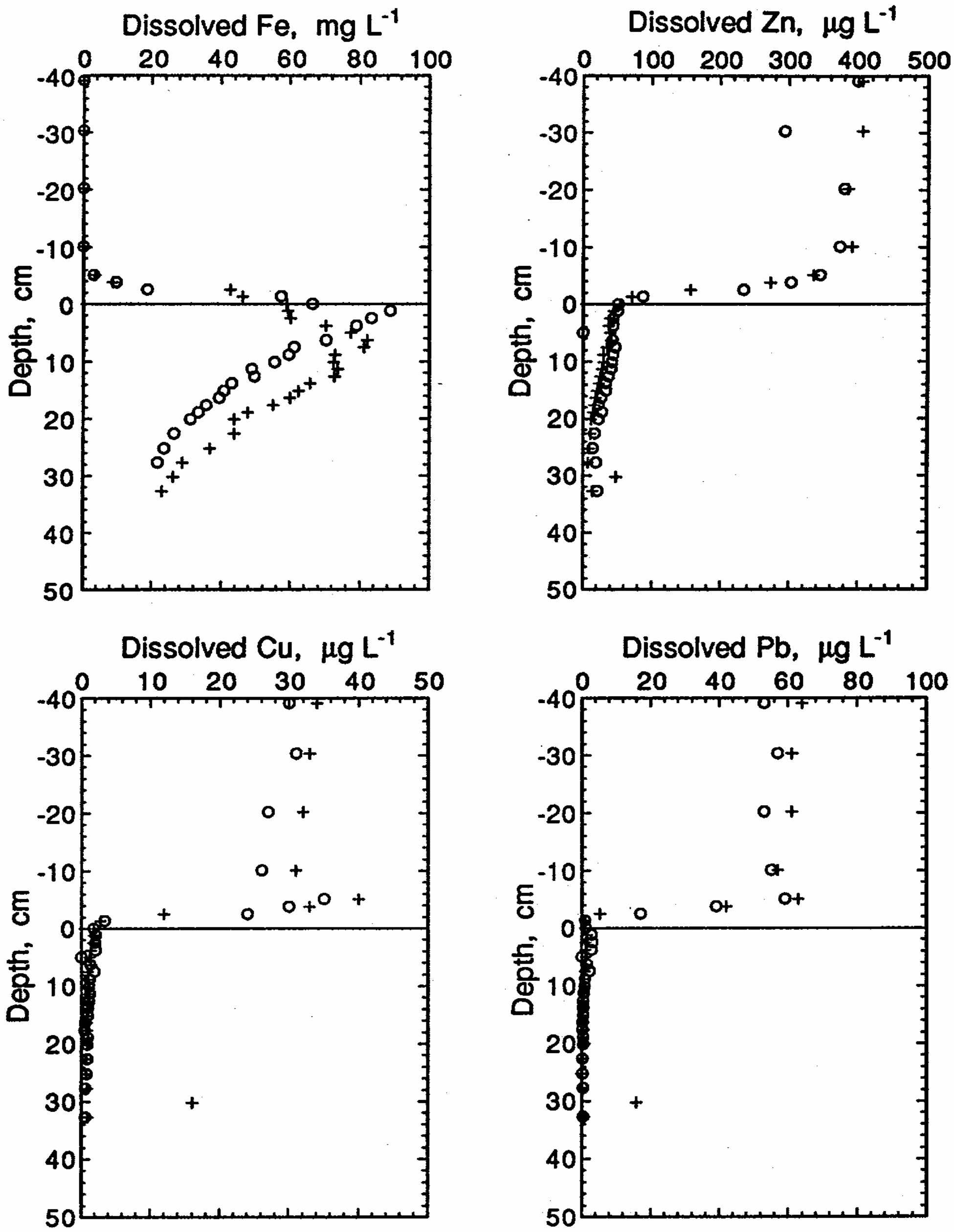
scheme iron oxyhydroxide phases are dissolved only after oxygen is depleted. Thus, the appearance of dissolved Fe in solution (Figure 4.2-6) strongly implies that the tailings were anoxic at very shallow depths at this site at the time of sampling, despite the water column being well oxygenated and frequently turbulent. The presence of anoxia is further supported by the near-zero concentrations of dissolved NO^{-}_{3} measured on shallow sub-surface pore water samples (MEND 2.11.3abc).

2. The rapid declines in dissolved Zn, Cu and Pb concentrations seen in duplicate pore water profiles (Figure 4.2-6) collected from the site illustrate the second diagenetic phenomenon. Despite the presence of a shallow, oxygenated and well-mixed water column, all three metals were diffusing into the tailings in the summer of 1993. This result is identical to that seen in studies at deeper sites in the lake in 1990 (Pedersen *et al*. 1993; 1997a,b). It has been previously suggested (Pedersen *et al*. 1993) that sulphate reduction and the associated production of dissolved sulphide in pore waters would promote the precipitation of Zn, Cu and Pb as authigenic sulphides in the Anderson Lake tailings. The common occurrence of pyrite framboids in surface sediments in the lake supports this view (MEND 2.11.1b And).

In most Canadian Shield lakes, sulphate inflow is minor and this normally limits sulphide precipitation during sedimentary diagenesis. In Anderson Lake, the presence of the acid (and sulphate) generating road along the north shore of the lake along with the sulphate-rich acidic drainage that is pumped to the lake from proximal abandoned mines, as well as ongoing dissolution of gypsum co-deposited with the tailings (MEND 2.11.1b And), guarantees abundant substrate to support bacteria sulphate reduction. All that is additionally required to deplete SO_4^{2-} is sufficient labile organic matter to foster the required oxidant demand, and the data collected to this point suggest that this requirement is met in the lake. Note that many candidate receiving basins for sulphide-rich tailings will not share the same characteristics as Anderson Lake with respect to sulphate inputs and organic matter settling fluxes (although these parameters could be manipulated by mine operators). Thus, the Anderson Lake experience should not be casually extrapolated to other deposits.

It was concluded that the pyrite- and pyrrhotite-rich tailings deposited in Anderson Lake show no evidence of oxidation even at a shallow turbulent site where the water column in summer is continuously well oxygenated. Indeed, multiple dissolved Zn, Cu and Pb profiles (Figure 4.2-6) across the sediment-water interface show that these dissolved metals are diffusing from the water column into the tailings, which are anoxic at a very shallow subsurface depth. Similar influxes have been previously observed in the lake, and can now be considered to be a well-established phenomenon there. The contamination of Anderson Lake waters by inputs of sulphate from external sources of acid drainage may contribute to metal uptake by the tailings by supporting the production by sulphate-reducing bacteria of metal-consuming sulphide.

Figure 4.2-6 **Duplicate Iron, Zinc, Copper and Lead Pore-Water Profiles Obtained in Tailings**



<u>Note:</u> The horizontal line at 0 cm depth in each panel represents the sediment-water interface, the location of which was established visually by examining the peeper surfaces for traces of sediments after retrieval of the dialysis arrays. Because the water column at the site was very well mixed, the metal analyses strongly imply that the interface in the figures has been drawn about 3 cm too low.

Source: Pedersen *et al.* 1997

Buttle Lake

Buttle Lake is a 35-km long by 1 km wide by maximum 87-m deep oligotrophic water body that occupies a U-shaped valley in an area of high relief on central Vancouver Island, British Columbia. Between 1967 and mid-1984, some 5.5 x 10^6 tonnes of pyrite-rich, Zn-, Cu-, and Pb-bearing tailings were discharged to the south basin of the lake via a raft-supported outfall that was submerged below the thermocline (MEND 2.11.1b). The tailings consisted of sand- and silt-sized silicate gangue minerals and residual copper, iron, lead and zinc sulphides. Base metal concentrations in the tailings solids ranged widely but averaged 7,000, 1,300 and 900 µg/kg for Zn, Cu and Pb respectively (Pedersen 1983). Lime (CaO) was used in significant quantity in the mill; approximately 1.0 kg of lime per tonne of ore was added to the tailings to raise the pH in the milling circuits and to enhance thickening.

Dissolved Zn, Cu, Pb and Cd concentrations in Buttle Lake water began to increase shortly after the mine commenced operation, and peaked in 1981, reaching levels in south basin surface waters as high as 370 µg/L Zn, 40 µg/L Cu, 25 µg/L Pb, and 3.6 µg/L Cd (Deniseger *et al.* 1990). The high metal loadings were derived not from the tailings (Pedersen 1983), but instead from ARD which percolated from a waste rock area into a stream beside the minesite and thence to the lake. In 1983, a surface and groundwater collection and treatment system was installed to capture the bulk of the acidic drainage and metal levels subsequently declined significantly.

The pH of Buttle Lake waters is near neutral or slightly alkaline (MEND 2.11.4a). Concentrations of dissolved metals measured in October 1993 as part of the study reported here were the lowest seen in the south basin since 1973. Zinc levels ranged from a minima of ~ 5 µg/L in surface or near-surface waters to a maxima of ~ 24 µg/L in the deepest samples at 35 m depth. Copper concentrations were < 2 µg/L throughout the water column.

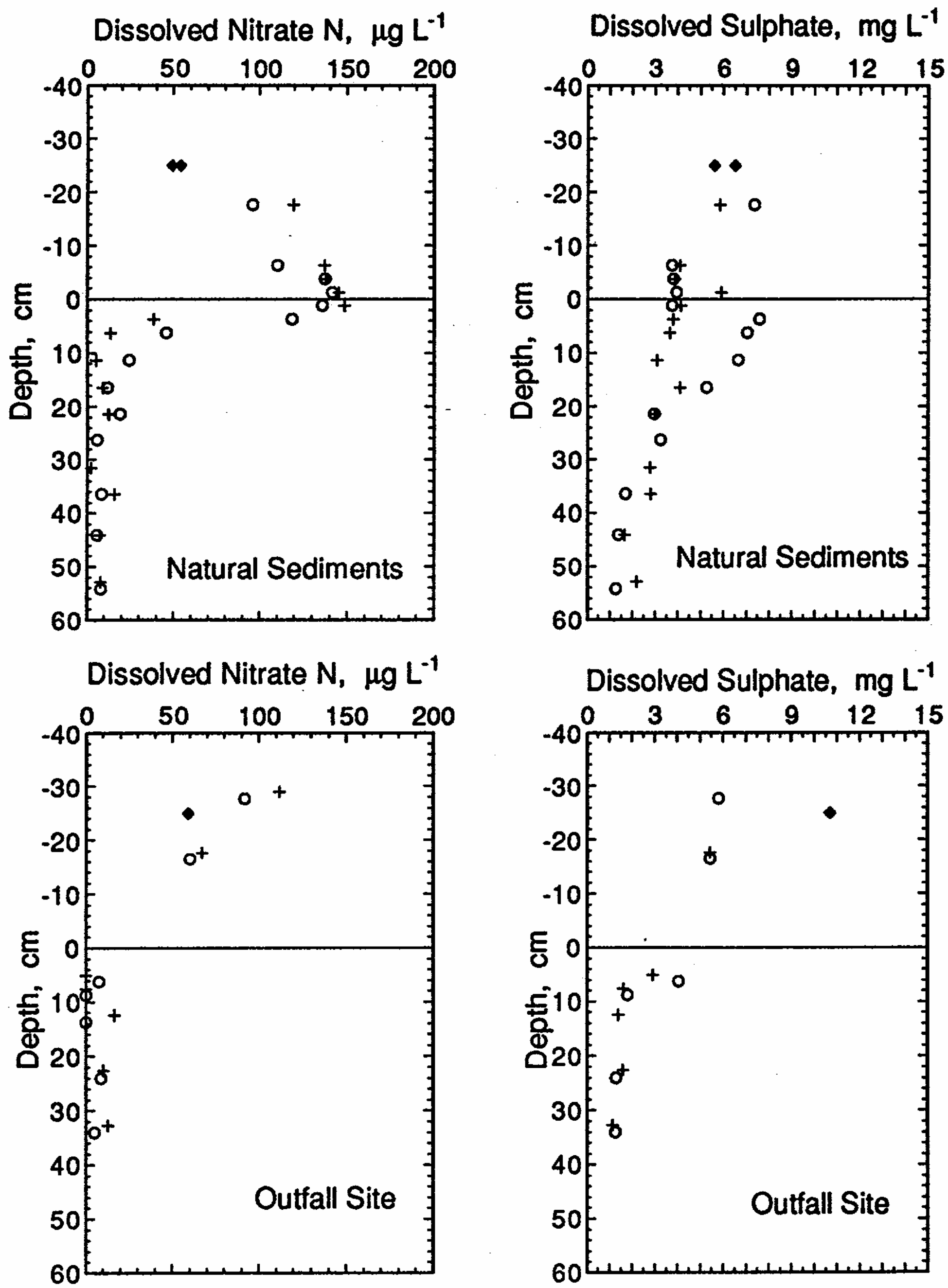
Previous MEND studies in Mandy Lake, Manitoba, and Benson and Buttle Lakes in British Columbia (MEND 2.11.1a, b and c) had shown that metal release from abandoned submerged tailings deposits was essentially non-existent. MEND 2.11.4a (also Pedersen *et al.* 1997a,b) investigated this aspect and applied higher resolution methodologies in their study of Buttle Lake. Sediment core and peeper sampling was carried out at two locations within Buttle Lake: 1) near the original tailings outfall, "the Outfall Site", and 2) a "Natural Sediments" site located 7 km northward in the tailings-free, central part of the lake. The lake bottom deposits at the Outfall Site comprised a ~ 4 cm thick layer of mostly natural, organic-rich sediments overlying homogeneous, pyrite-rich tailings.

Pore water samples from peepers emplaced at the two sites were analyzed for sulphate, nitrate and a suite of dissolved metals, including Fe, Mn and Zn. Duplicate dissolved nitrate and sulphate profiles (Figure 4.2-7) show rapid subsurface declines that indicate the onset of anoxia at depths of a few centimetres at the Outfall Site. At the Natural Sediments location, the nitrate and sulphate data imply a slightly lower relative oxidant demand and the penetration of oxygen to slightly greater depths. Dissolved iron and manganese distributions in the outfall deposits are consistent with the NO_3^- and SO_4^{2-} data: release of both metals to solution is seen below ~ 1-2 cm (Figure 4.2-8). The apparent paucity of manganese and iron oxide phases in the near-surface deposits at the Natural Sediments site is reflected by low concentrations of dissolved Mn and Fe in the upper two decimetres at this location (Figure 4.2-8). Given the rapid kinetics of iron oxidation at neutral pH, the low dissolved Fe concentrations in the upper 1 to 2 cm at both sites implies an aerobic surface layer that is at least 1 cm thick. Dissolved sulphide was undetectable at depth in either of the sites but this is not surprising given that sulphate is present as a minor constituent in Buttle Lake water, unlike the case in Anderson Lake. Any sulphide that is produced via sulphate reduction would almost certainly be precipitated from Buttle Lake pore waters immediately and quantitatively as metal sulphides.

The dissolved iron and manganese profiles indicate that oxide cycling - i.e. dissolution upon burial, upward diffusion of Fe^{2+} and Mn^{2+}, and reprecipitation of oxyhydroxides near the interface - is a significant early diagenetic process in the area of the abandoned tailings deposit in the south basin of Buttle Lake. Many dissolved metal cations (and some anions) have a significant adsorption affinity for Fe and Mn oxides. Thus, the oxide cycling in the near-surface deposits in the south basin could have an influence on the post-depositional mobility of other elements.

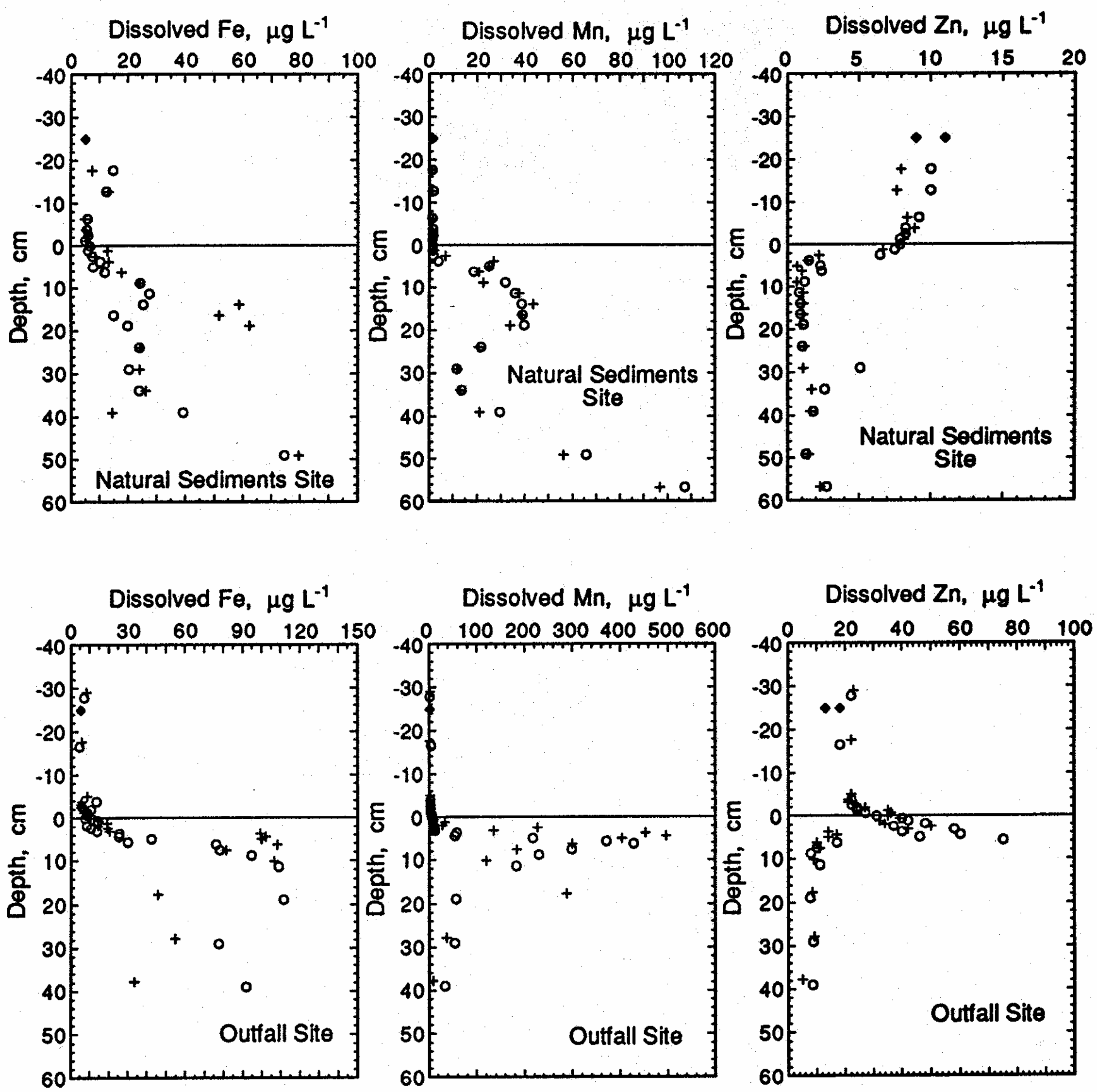
Zinc provides an example of the above. Dissolved Zn decreases sharply from ~ 10 µg/L to < 2 µg/L in the upper few centimetres at the Natural Sediments site (Figure 4.2-8), indicating that the metal is diffusing into these deposits and precipitating. This influx is similar in form (but not quantum) to that seen in Anderson Lake, and is also typical of many Shield lakes (Carignan and Tessier 1985). A different situation exists at the Outfall Site, however, where oxide phases are accumulating on top of the tailings. Zinc is released to solution between about 1 and 5 cm depth at this location (Figure 4.2-8) and consumed in the immediate underlying tailings. The maxima in the duplicate profiles are spatially indistinguishable from the six- to ten-fold higher manganese maxima, and lie slightly higher than the dissolved Fe peaks (compare the lower panels in Figure 4.2-8). The close correspondence between the zinc and manganese distributions suggests that a significant amount of zinc is adsorbed by Mn oxides in the near-surface deposits, and that the ongoing early diagenetic cycling of the oxides exerts the principal control on the post-depositional behaviour of zinc at this location. Precipitation of authigenic zinc sulphide in the tailings presumably accounts for the dissolved zinc depletion below ~ 7 cm depth, while the sink for manganese at depth is unknown.

Figure 4.2-7 **Duplicate Pore-Water Nitrate and Sulphate Profiles Obtained at Two Sites in Buttle Lake in October, 1993**



Source: Pedersen *et al.* 1997

Figure 4.2-8 **Duplicate Pore-Water Iron, Manganese and Zinc Profiles Obtained at Two Sites in Buttle Lake in October, 1993**



Note: The filled diamonds represent analyses of near-bottom water samples collected in Go-Flo bottles during the period the peepers were emplaced.

Source: Pedersen *et al*. 1997

The steep increase in the zinc concentration between the level observed in bottom water (~ 20 µg/L) and the peak value at ~ 5 cm depth (~ 70 µg/L) implies that zinc is diffusing from the sediments into deep waters in the south basin of Buttle Lake, which is opposite to the case for the natural sediments in the central basin. To put in perspective the potential impact of the indicated Zn efflux on local water quality, the dissolved zinc data were used to calculate the annual input of the metal to the lower 50 m of the water column in the south basin previously (MEND 2.11.4a). The resulting worst-case estimate was an annual addition of 0.5 µg/L. Thus, the ongoing diagenetic cycling of zinc is not posing a threat to Buttle Lake. Furthermore, the zinc cycling cannot be directly attributed to the subaqueous disposal of tailings in Buttle Lake. This conclusion draws on the original pore water work done in the lake during the period of active tailings discharge (Pedersen 1983). At that time, there was no zinc efflux from the tailings, and no indication that the oxidation of the sulphides was occurring on the lake floor. However, the lake was being contaminated by acidic drainage emanating from the nearby mine site. The cycling that is now occurring probably represents a legacy of this contamination from terrestrial sources. Had there not been an external source of dissolved zinc to the lake, there could not have been the significant uptake of zinc onto oxide phases and consequent recycling to the water column that has been occurring in recent years.

There is no evidence in either recent work (e.g. MEND 2.11.4a) or previous studies (e.g. MEND 2.11.1b) that pyrite-rich tailings are oxidizing on the floor of Buttle Lake despite the passage of more than a decade since discharge of tailings ceased. The tailings are now being progressively covered, albeit rather slowly, with a veneer of natural sediments that is anoxic at shallow sub-bottom depths. In the absence of future physical disturbance, the sulphides in the tailings should remain in a thermodynamically-stable anoxic state in perpetuity. However, there is ongoing diagenetic metal cycling in Buttle Lake sediments that is responsible for limited release of zinc to bottom waters. Rather than being blamed on post-depositional oxidation of tailings, however, this phenomenon can instead be attributed to the adsorption of zinc from the water column onto settling oxide phases or fresh precipitates that form continuously at or near the sediment-water interface. The source of the zinc is acidic drainage from the nearby mine site which is delivered to the lake by surface runoff. Here again, the geochemical and environmental villain is not the submerged tailings deposit, but instead the exposure of sulphides on land to oxygen.

<u>Equity Silver Tailings Pond</u>

The Equity Silver Mine operated near Houston in north-central British Columbia from 1980 until 1994. Ore minerals in the deposit consisted of chalcopyrite and the copper-silver-arsenic sulphosalts tennantite and tetrahedrite. Pyrite, arsenopyrite, sphalerite, galena, and pyrrhotite also occurred in the ore. Over the course of the mine life, production rates ranged from 1,500 t/d to 9,000 t/d. Flotation was used to process most of the ore, but in 1985 a cyanide leach plant was added to recover gold, and this ran until mine closure. Pressure oxidation leaching was also carried out during the early years of the operation, and the leaching residues were deposited in the pond. Tailings were discharged via a mobile floating platform to a constructed water-filled pond 1.2 km^2 in area and up to 5 m deep. The tailings hosted significant concentrations of pyrite, arsenopyrite and pyrrhotite. For several years prior to mine closure, the tailings pond also received neutralized sludge derived from a treatment plant that had been built to remove metals from drainage from waste rock piles. The sludge contained a variety of metal salts, and was unequally distributed in the pond. This complicated the interpretation of ongoing chemical diagenesis in the abandoned pond.

A case study at Equity Silver was also desirable because cyanide leaching was used in the mill, and significant quantities of cyano-metal complexes had been introduced to the tailings pond prior to the mine ceasing operations. As the Equity ore contained significant amounts of arsenopyrite, this afforded an opportunity to additionally assess arsenic chemistry under MEND.

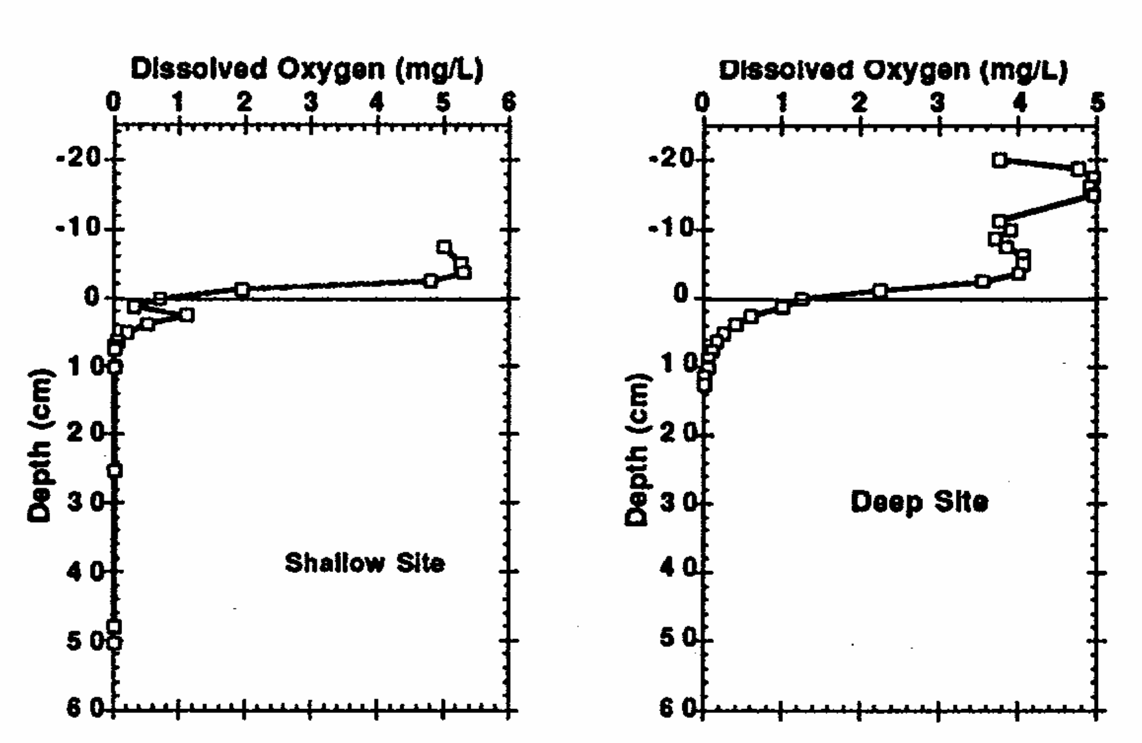
At Equity Silver, Pedersen *et al.* (1997a) addressed the issue that there is a possibility that tailings discharged under water in constructed tailings ponds could behave differently than submerged tailings in natural lakes. The Equity Silver tailings pond was chosen for study since it met two key criteria: tailings deposition to the pond had recently ceased, and fluxes of organic matter to the pond had been low during mine operation, unlike the case in productive Anderson Lake or Buttle Lake, which receives significant inputs of fluvial organic and lithic detritus.

Two sampling sites were used in the Equity tailings pond, one shallow (1.8 m) and the other deep (5.0 m). Analysis of samples from sediment cores showed that organic carbon is a minor constituent; concentrations ranged from < 0.05 to ~ 0.25 wt% (MEND 2.11.5c). Oxygen profiles measured on peeper samples at both locations showed a steep decline in O_2 content in the upper several centimetres of the tailings and complete oxygen depletion within the upper decimetre (Figure 4.2-9). Given the paucity of organic matter in the deposits, it is not obvious that aerobic bacteria can account for this depletion; abiogenic oxidation of sulphides could also play a role. To evaluate this, a modelling approach was used, as detailed in MEND 2.11.5c.

The lack of historical data on the compositions of the various discharges to the Equity pond during the lifetime of the mine compromises interpretation of the chemical distributions reported here and in MEND 2.11.5c. This constraint points out the importance in post-closure planning of knowing what goes into a pond during its lifetime. Without such information, it is difficult to predict in a meaningful way the future geochemical response.

Overall, the various MEND studies that have been carried out, particularly those in lakes, strongly support the original subaqueous disposal hypothesis: the storage of sulphide-rich tailings under water, if properly conducted, can be an environmentally sound and permanent disposal option.

Figure 4.2-9 **Dissolved Oxygen Profiles in the Near-Bottom and Pore Waters at each Site in the Equity Pond**



Note: A calibrated mini-electrode was used for the measurements, which were made inside a nitrogen-filled glove bag shortly after recovery of the peeper arrays.

Source: Pedersen *et al.* 1997

Assuming that all oxygen diffusing into the tailings is consumed by the oxidation of pyrrhotite, via the simplified reaction:

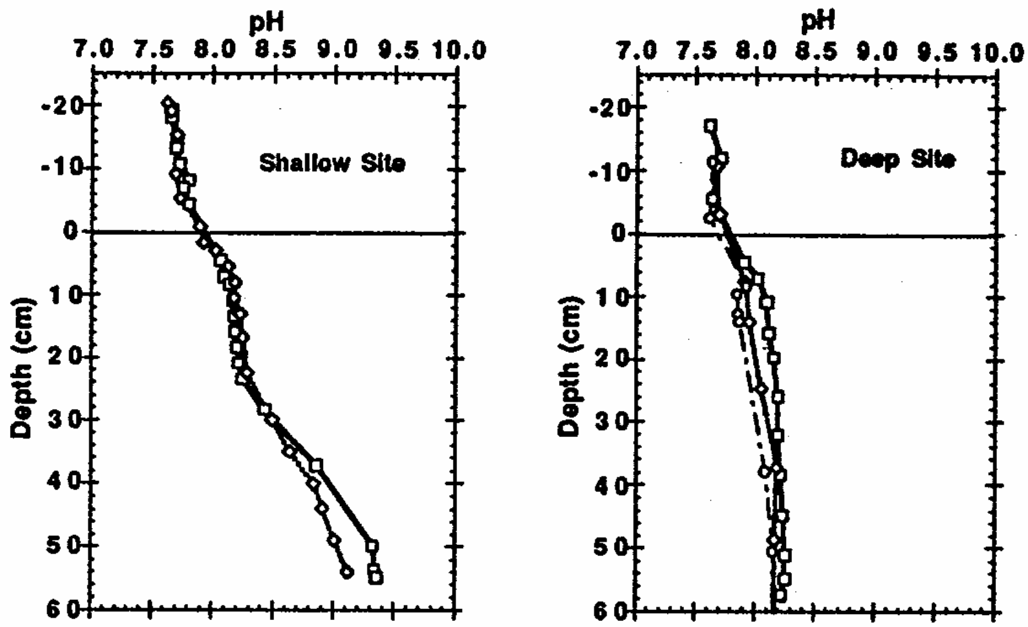
$$FeS + 9/4O_2 + 5/2\,H_2O \qquad\qquad SO_4^{2-} + Fe(OH)_3 + 2H^+$$

At steady state, a maximum of ~ 7 $mg/m^2/d$ of sulphate and ~ 3 µmol m^2/d of protons (H^+) would be released. The calculated maximum sulphate release would contribute < 0.03% of the water column sulphate inventory (MEND 2.11.5c). Thus, sulphide oxidation cannot account for the ~ 1,900 mg/L sulphate content of the pond water. Furthermore, although the impact of the proton release cannot be calculated in the absence of information on the buffering capacity of the pond water and sediments, pH measurements show that the pore waters are slightly alkaline, and there is no evidence of a decline in pH in the oxygen consumption zone (Figure 4.2-10). These observations argue rather firmly against the hypothesis that sulphide oxidation is occurring at the sediment/water interface in the Equity pond to the extent that there is a significant effect on pond water chemistry. Finally, there are internal sources of labile organic matter to the surface sediments in the pond, and these could account for the observed oxygen demand. Two sources were observed: (i) during the two-week deployment of the peeper arrays, the aluminum frames that held the peepers in the bottom became coated with a film of green algae; and (ii) periphyton was also observed growing near the water line along the shores of the pond.

Lack of oxidation of the submerged tailings in the pond implied by the modelling results and the pH data does not exclude release of at least one metal, however, arsenic is abundant in the solid phase of the Equity deposits. Concentrations at the shallow site, for example, reached as high as 18,000 ppm (1.8 wt%) (Figure 4.2-11). Triplicate profiles of dissolved As reflect the solid-phase distributions at the shallow site. Concentrations in solution reached 2.5 mg/L at 5 cm depth which is spatially very close to the solid-phase As maximum. A somewhat similar correspondence is observed at the deep site, where triplicate profiles reveal maximum release of arsenic to pore waters in the upper 5 cm. The solid-phase content is also highest in this interval.

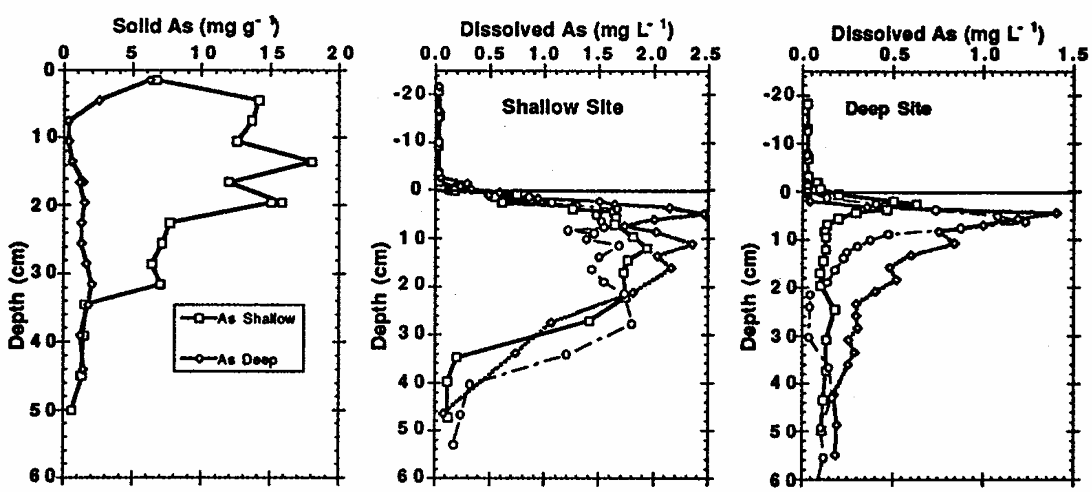
The dissolved antimony (Sb) distributions stand in marked contrast to those for arsenic. Triplicate peeper profiles show that the dissolved Sb concentration at both locations declines sharply across the sediment/water interface from about 35 µg/L in the pond water column to an average value two- to five-fold lower in pore waters below 3 cm depth (Figure 4.2-12). The steepness of the pore water concentration gradient across the interface indicates that precipitation of an authigenic Sb phase or phases must be occurring at very shallow depths at both sites. The composition of the implied sink for the metalloid is unknown. However, antimony in the reducing (and alkaline) pore waters should be present as the reduced trivalent species Sb^{3+}, and the pore waters contain very high concentrations of dissolved sulphate (typically 20-30 mmol/L, MEND 2.11.5c). Thus, a possible (but highly speculative) candidate salt that could provide the sink is $(Sb^3)_2\ (SO_4)^3$, which is "insoluble" under alkaline conditions (CRC 1994). However, since a solubility product is not available for this phase, the researchers could not verify whether or not this suggestion is plausible.

Figure 4.2-10 **Multiple pH Profiles Measured by Mini-Electrode in the Near-Bottom and Pore Waters at each Site**



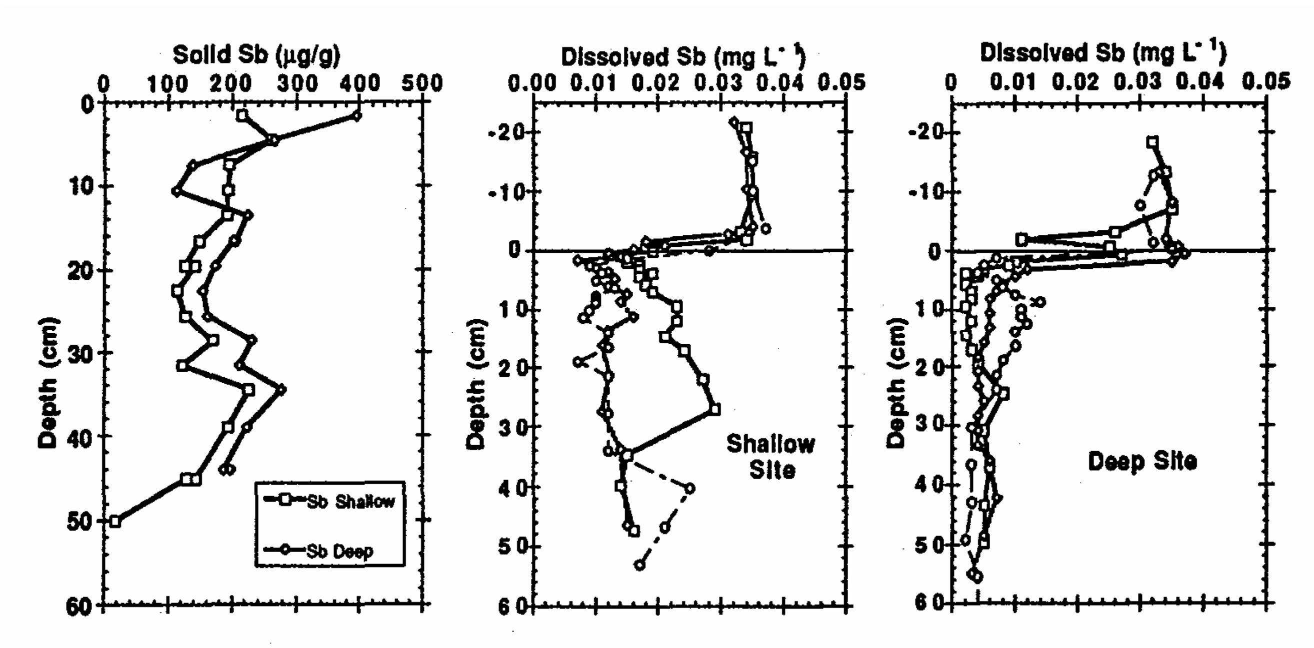
Note: Each profile was measured on a separate peeper array
Source: Pedersen *et al*. 1997

Figure 4.2-11 **Solid-Phase (from Sediment Cores) and Dissolved Arsenic Profiles at both Sites in the Equity Pond**



Note: Dissolved profiles were from separate peepers. The cores were collected within 3 m of the respective peeper arrays.
Source: Pedersen *et al*. 1997

Figure 4.2-12 **Solid-Phase (from Sediment Cores) and Dissolved Antimony Profiles at both Sites in the Equity Pond**



Note: Dissolved profiles were from separate peepers. The cores were collected within 3 m of the respective peeper arrays.

Source: Pedersen *et al*. 1997

<u>Geochemical Assessment of Subaqueous Disposal-Louvicourt Tailings, Val d'Or, Québec</u>

In 1995, MEND initiated a coordinated and important, multi-participant research project, referred to as the Louvicourt tailings project, to assess the reactivity of subaqueously disposed tailings in artificial or man-made impoundments with shallow water covers.

The Louvicourt tailings project encompassed both laboratory and field investigations related to the subaqueous disposal of fresh tailings at the Louvicourt copper-zinc mine located approximately 20 km east of Val d'Or, Québec. The Louvicourt mine tailings area is designed to be a flooded tailings repository. The tailings are potentially net acid generating with the predominant sulphide mineral, pyrite, representing about 30 to 45 wt % of the tailings mass (Li and St-Arnaud 1999). The tailings also contain minor trace amounts of pyrrhotite, sphalerite and chalcopyrite. The predominant carbonate minerals are ankerite and siderite, and the main silicate neutralizing mineral is clinochlore. The carbonate mineral content of tailings samples have ranged from near zero to 24 wt %.

The Louvicourt tailings project has five key reports. They are:

1. <u>Evaluation of Man-Made Subaqueous Disposal Option as a Method of Controlling Oxidation of Sulphide Minerals (Phase 1): Background and General Description</u> by Golder Associés (MEND 2.12.1b). This document presents an overview of the project and describes the mine site, the tailings pond, cell design, construction, filling and instrumentation.
2. <u>Reactivity Assessment and Subaqueous Oxidation Rate Modelling for Louvicourt Tailings</u> (MEND 2.12.1d). This document summarizes the results of investigations carried out by the Noranda Technology Center (NTC). In the studies, various Louvicourt tailings samples were characterized for grain size, quantitative mineralogy, geochemical whole-rock composition, and extended acid-base accounting. Flow-through cell leach tests were used to assess metal release from the tailings under various conditions in humidity cells. Modelling techniques were applied to examine site-specific factors and chemical and physical processes that influence submerged sulphide oxidation and ultimately the water cover quality (Figure 4.2-13). As such, the research directly addressed uncertainties related to the quantification of the effects of four mechanisms for the transport of dissolved oxygen to submerged sulphide tailings: diffusion, convection, circulating current, the resuspension of tailings solids, and wave action.
3. <u>Evaluation of Man-Made Subaqueous Disposal Option as a Method of Controlling Oxidation of Sulphide Minerals: Column Studies</u> (MEND 2.12.1e). This document presents the results of a two-phase column study carried out by CANMET under which Phase I of the study examined the effects of circulated and non-aerated water covers, and Phase II also included precipitation, runoff and drawdown at rates comparable to those observed at the Louvicourt site. The column testing involved the following four disposal scenarios:

- A 0.3 m deep water cover over the tailings;
- A 0.3 m water cover over a 0.3 m peat intermediate layer over tailings;
- A 0.3 m water cover over a 0.3 m sand intermediate layer over tailings; and
- A 1 m deep water cover over the tailings.

4. Subaqueous Disposal of Reactive Mine Tailings, Louvicourt Mine Test Cells: Geochemical Sampling and Analysis, Final Report by INRS-Eau (MEND 2.12.1c). This report describes the chemical behaviour and evolution over the course of the 2.5-year field test program for submerged tailings in two test cells constructed next to the Louvicourt tailings impoundment. Overlying water and pore water were characterized via direct microelectrode profiling and the deployment of dialysis arrays (peepers).
5. A Synopsis Report (MEND 2.12.1a), involved the in-depth review of the methodologies and results of the Louvicourt tailings project studies. The key conclusions of this document are reiterated later in this section.

Reactivity Assessment by NTC

Li *et al.* (1997) reported on work completed at NTC in assessing the degree of oxidation of sulphide tailings under a water cover for four idealized conditions (Table 4.2-8). For each case, NTC predicted the oxygen uptake due to sulphide oxidation, the metal release and the effect on water cover quality using mathematical modelling and site conditions and tailings properties representative of the Louvicourt tailings impoundment.

Table 4.2-8
Idealized Cases - Calculation of Sulphide Oxidation of Sulphide Tailings under a Shallow Water Cover

Case No.	Description	O_2 Concentration			Downward Infiltration	Tailings Resuspension
		Water Cover Depth	At Surface	At Water/Tailings Interface		
1	Stagnant water cover	0.3 m	11.3 mg/L	0 mg/L	None	None
2	Fully oxygenated cover with no downward infiltration	0.3 m	11.3 mg/L	11.3 mg/L	None	None
3	Fully oxygenated cover with downward infiltration	0.3 m	11.3 mg/L	11.3 mg/L	1 m/year	None
4	Case 3 with tailings resuspension	0.3 m	11.3 mg/L	11.3 mg/L	1 m /year	Yes

In Case 1 (Table 4.2-8), the water cover is oxygen saturated at the air/water interface and dissolved oxygen to the tailings is limited to molecular diffusion. This represents the minimum oxygen uptake scenario (Figure 4.2-14). The researchers observed that the effectiveness of a stagnant water cover would increase with depth, but the incremental gain in effectiveness rapidly decreases with increased water cover depth. Increasing the water depth to more than 1 m would not significantly increase the water cover effectiveness. The dissolved oxygen intakes for Cases 2 to 4 are also shown in Figure 4.2-14. In effect, the modelling results showed that for the Louvicourt site conditions and tailings characteristics and properties, the contributions of water cover aeration, downward infiltration, and tailings re-suspension are at low levels, and approximately one order of magnitude higher than oxidation under a hypothetical stagnant water cover.

MEND 2.12.1d reports that based on the mathematical modelling of the four cases shown in Table 4.2-8:

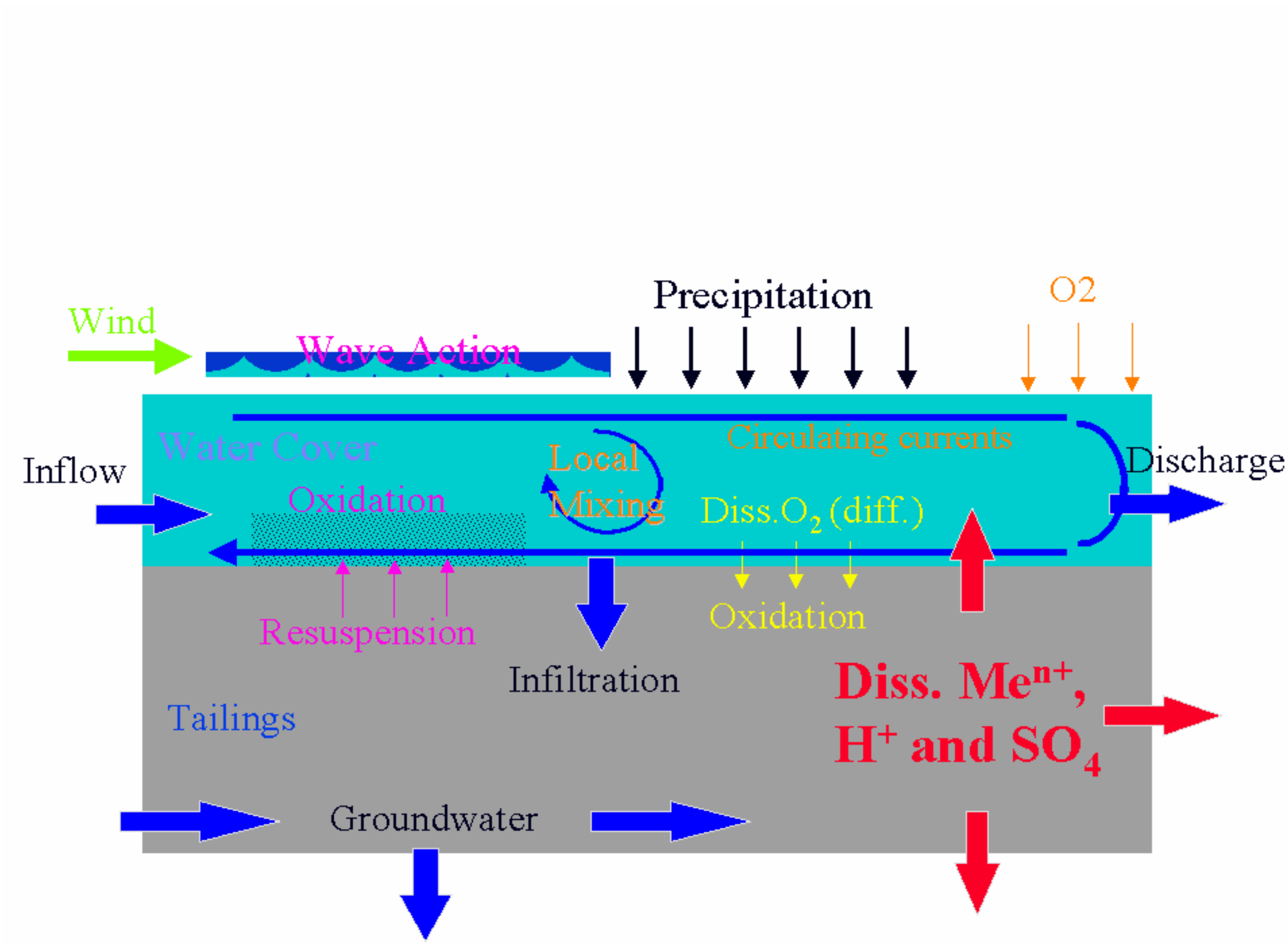
- The stagnant water cover transported the least amount of oxygen to the submerged tailings, with the flux being in the order of 3 g O_2/m^2 interface per year. This situation is unlikely to occur as winds are almost always present in the field and naturally cause mixing, circulation, wave action, and aeration in shallow water bodies. The three remaining cases are more realistic scenarios, which significantly increase oxygen flux into submerged tailings.
- Compared to the stagnant water case, mixing/oxidation of the water cover and tailings resuspension are each capable of increasing the oxygen flux by an order of magnitude, whereas downward infiltration of fully aerated water can enhance the oxygen flux by a factor of three.
- The range of oxygen fluxes predicted in the modelling suggests that for most sites, a simple well-maintained water cover, without additional measures, would be sufficient to suppress sulphide oxidation while maintaining the discharge from the water cover in compliance. For circumstances where this is not achievable, additional measures including physical, chemical and biological barriers/oxygen interceptors can be applied to enhance the effectiveness of the water cover.

<u>Column Testing by CANMET</u>

The following four subaqueous disposal scenarios were investigated using Louvicourt tailings in column tests (MEND 2.12.1e):

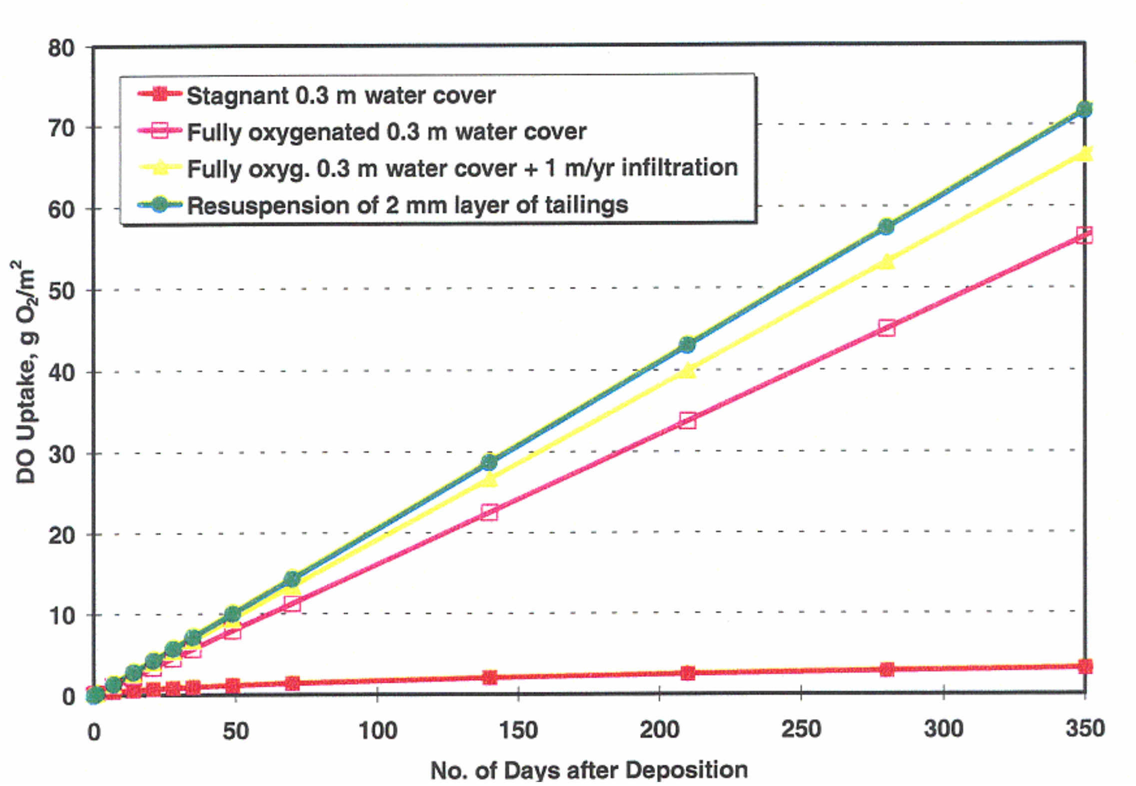
1. A 0.3 m deep water cover directly on tailings;
2. A 0.3 m deep intermediate peat layer over tailings under a 0.3 m water cover;
3. A 0.3 m deep sand layer over the tailings surface under a 0.3 m water cover; and
4. A 1.0 m deep water cover directly on tailings.

Figure 4.2-13 **Processes Affecting Subaqueous Sulphide Oxidation and Water Quality**



Source: Li *et al.* 1997

Figure 4.2-14 **Comparison of Dissolved Oxygen Consumption among Four Cases**



Source: Li and St-Arnaud 1999

Excerpts from the CANMET report (MEND 2.12.1e) follow.

Phase I of the column tests lasted for 200 days and focused on oxygen diffusion and ionic fluxes under conditions of a circulated water cover. In the first 100 days, the water cover in each column was circulated but not aerated. In the second 100 days, aeration of the water cover was also included. Phase II, which lasted for 13 months, incorporated precipitation, runoff and drawdown events at rates comparable to those observed in the field. The impact on the chemistry of the water cover and pore water in each series of columns was investigated. Both Phases I and II commenced with a new batch of natural water as water cover such that only the pore water in each column retained remnant effects of the previous stage of testing.

The test results indicate that, especially during Phase I, both sulphide oxidation and sulphate dissolution contributed to increasing sulphate concentrations in the water covers overlying the tailings. A slight pH depression that was observed in the overlying water in the peat and sand columns during Phase I was likely caused by the hydrolysis of Fe and Mn near the water/solids interface. After the initial flushing of stored weathering products, however, both peat and sand layers can serve as an effective diffusion barrier to suppress chemical weathering of the underlying tailings over the long run. Largely controlled by the alkalinity balance in the water covers, the 1.0 m deep simple water cover (without an intermediate barrier layer) appeared to perform better than the 0.3 m deep water cover in suppressing sulphide oxidation and metal leaching in the submerged tailings under laboratory test conditions. In any case, the precipitation of iron oxyhydroxides at the water/tailings interface and drawdown limited the efflux of undesirable metals to the overlying water column.

In agreement with the observations made during the field and modelling studies conducted by Li *et al.* (1997) and MEND 2.12.1d and INRS-Eau (MEND 2.12.1c), respectively, only minor dissolved zinc diffused from the tailings pore water to the overlying water column. The following conclusions were drawn based the observations and test results of the Phases I and II investigations of four subaqueous disposal options for the Louvicourt tailings (MEND 2.12.1e):

- While water covers serve to effectively limit the availability of oxygen (in terms of reduced concentrations), they do not totally eliminate sulphide oxidation in submerged tailings in the absence of a more effective oxygen scavenger than reactive sulphides. Largely depending on the alkalinity balance in a tailings disposal facility, a deeper water cover can be more effective in arresting metal leaching.
- Intermediate layers like peat and sand may act as source or sink of some dissolved constituents, depending on their chemical composition. While they generally serve to reduce access of dissolved oxygen to submerged tailings, such materials must be characterized in detail to assure their inert nature prior to incorporation as a diffusion barrier. This will avoid unwarranted impact to the water covers as noted in the sand- and peat-containing columns during Phase I of the investigation.

- Chemical processes occurring at the water/solid interface (e.g. co-precipitation or sorption with oxyhydroxides) and drawdown appear to have served to limit the efflux of undesirable parameters like dissolved copper and zinc to the overlying water. To decipher the detailed reactions involved and to more accurately delineate relevant chemical gradients, a sampling design with a much better resolution than that employed in the current study is required. Gel samplers may be preferable to sampling ports at fixed depths. Chemical reactions associated with drawdown may also have significant impact on pore water quality and cannot be ignored in studies using column set-ups.

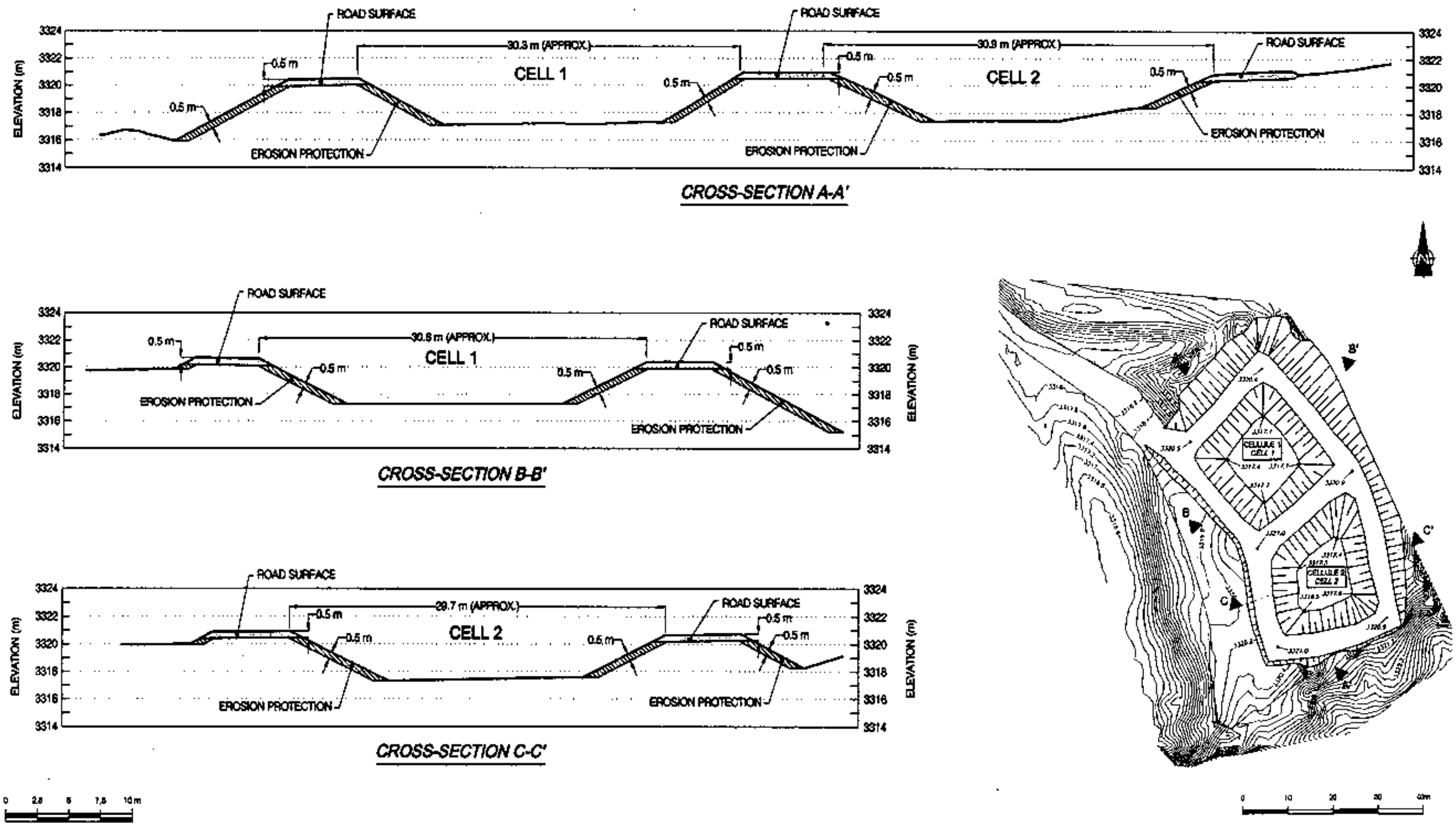
Louvicourt Field Cell Study by INRS-Eau

Two test-cells each measuring about 20 m x 20 m were constructed in 1996 in an area adjacent to the Louvicourt tailings pond (Figure 4.2-15). The cells were filled with 2 to 3 m of tailings and covered using a 0.3 m deep water layer. INRS-Eau (MEND 2.12.1c) undertook a geochemical sampling and analysis program from 1996 to 1998 to assess the chemical stability of the submerged tailings in the test cells. The work involved the following three major components:

1. The profiling of pH and dissolved oxygen at the water/tailings interface using a microelectrodes with 1 mm resolution.
2. The sampling of overlying water and the tailings pore water (at 1 cm resolution) using in-situ dialysis arrays (peepers) and subsequent analysis to determine the water chemistry.
3. Sequential extraction analysis of the superficial layer of the tailings to assess metal leachability.

The results showed that the shallow water cover (0.3 m depth) was effective in reducing the rate of sulphide oxidation in the tailings. Oxygen penetration into the tailings was less than 7 mm and oxygen consumption was about 4000 times less than that of tailings samples that were exposed to the atmosphere in humidity test cells. The progressive oxidation of tailings did however occur beneath the interface with the overlying water cover. Two years after tailings disposal under water in the field cells, there was clear evidence of Cd and Zn mobilization from the tailings to the overlying water with concomitant changes in the solid-phase partitioning of the two metals (e.g. from refractory to more labile fractions).

Figure 4.2-15 **Louvicourt Field Test Cells**



Source: Golder Associés 1999

Key Conclusions Drawn from the MEND Louvicourt Tailings Project

Pedersen (2000) critically reviewed the results of the Louvicourt Project and reported that the work has yielded a significant body of information, some of which is useful in supporting the hypothesis that the subaqueous disposal of sulphide-rich, reactive, freshly milled tailings strongly inhibits oxidation and consequent metals release. Selected conclusions reached by Pedersen (2000) are summarized in MEND 2.12.1a and reiterated below.

- The detailed field study carried out by INRS-Eau (MEND 2.12.1c) is the most directly applicable to proving the hypothesis. It shows with little argument that reactivity of unaltered pyrite and other metal sulphides *in situ* under 30 cm of water is at worst minor on time scales of a few years. This conclusion appeared robust despite the limited duplication imposed by the heterogeneity of the tailings placed in the cells.

- The body of evidence now available demonstrates compellingly that water covers are effective in preventing sulphide oxidation and acid generation (Pedersen 2000 and MEND 2.12.1a).

Geochemical Assessment of Subaqueous Disposal for the Voisey's Bay Project, Labrador

In 1997, Mugo *et al.* investigated the geochemical stability of pilot plant tailings from the Voisey's Bay Ni-Cu-Co project, Labrador in laboratory experiments which involved:

- The monitoring of temporal trends in metals concentrations/fluxes in fresh water and sea water covers; and
- The assessment of the submerged tailings pore water chemistry using peepers.

In the experiments, a 30 cm deep layer of homogenized sulphide tailings was overlain by a 300 L fresh or sea water cover. The water cover was circulated and directed laterally across the interface. The tests were carried out over a six-month time frame.

Samples of the tailings supernatant were analyzed for trace metals and physical and chemical parameters. The researchers estimated the fluxes of dissolved metals into the water cover through the normalization of whole-tank concentration changes for a known sediment surface area and elapsed time. Selected results of the tests as reported by Mugo *et al.* (1997) are reiterated below.

- The different behaviour of fresh water and sea water covers is evident as shown in Figure 4.2-16. The greater neutralization potential of sea water resulted in a relatively constant pH and sulphate levels;
- Dissolved metal fluxes for Ni, Co, Cu, Zn, Mn and Fe over time are shown in Figure 4.2-17. The data indicate that the initial metals release that occurred at the start of the experiment

was typically followed by decreased or near constant flux values. These trends are likely due to precipitation as dissolved metals are converted to less soluble species, or to sorption, co-precipitation and scavenging mechanisms. Metal fluxes over the long term are additionally controlled by oxygen flux to the tailings. Long-term metal concentrations and fluxes were lower in the sea water cover - likely due to the higher buffering capacity of the sea water; and

- The dissolved oxygen profiles in the fresh water and sea water cover columns have a pronounced gradient across the interface with penetration depths of ~ 1 cm and ~ 5 cm for sea water and fresh water covers respectively.

Mugo *et al.* (1997) report that tailings reactivity can be inferred directly from pH gradients across the tailings/water cover interface as both the biologically - catalyzed and abiotic oxidation of sulphide tailings generates acidity and therefore increases proton activity.

The flux of oxygen across the water cover/tailings interface depends on the oxygen concentration at the top of the diffusive boundary layer that separates the tailings from the water column and on the thickness of the boundary layer (Mugo *et al.* 1997). The researchers also report that oxygen flux can be estimated, assuming steady state conditions, using Fick's First Law;

$$F = - D_{eff} (dc/dz)$$

where:

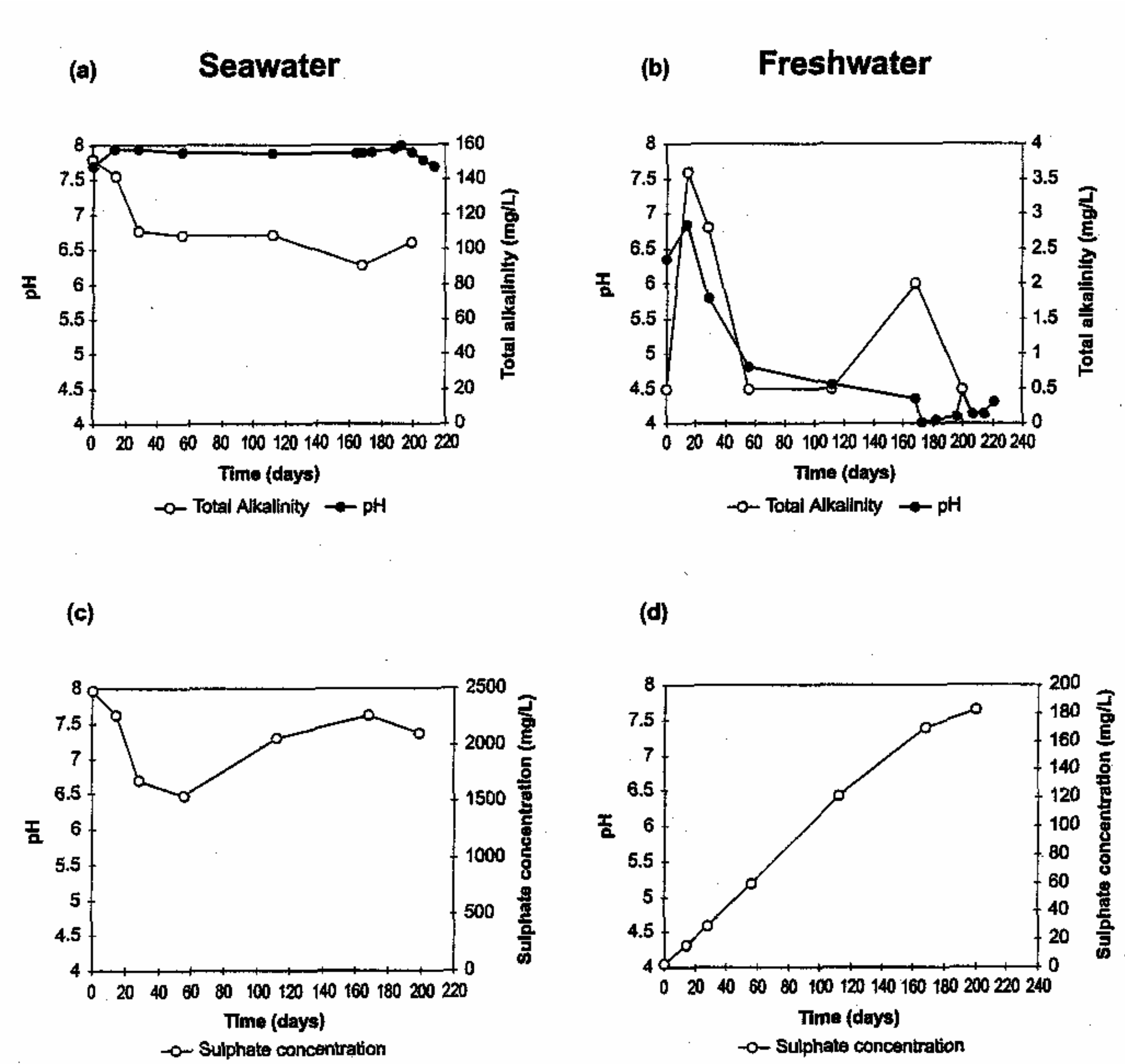
F	=	mass flux of oxygen (mass/unit area/time);
dc/dz	=	concentration gradient over distance z; and
D_{eff}	=	the effective diffusion coefficient (Eberling *et al.*1994).

The distribution of dissolved metals for fresh water and sea water covers, and peeper data revealed the following differences between the covers:

- The concentrations of dissolved Ni, Co and Zn in the tailings pore water peeper wells were lower in the sea water tank; and
- The metal concentrations profile peaks observed just below the interface in the sea water tank was not observed in the fresh water tank (Figure 4.2-18). The peaks in the sea water cover experiment suggests that dissolved metals sorption and/or co-precipitation is favoured because of the higher pH and buffering capacity of the sea water.

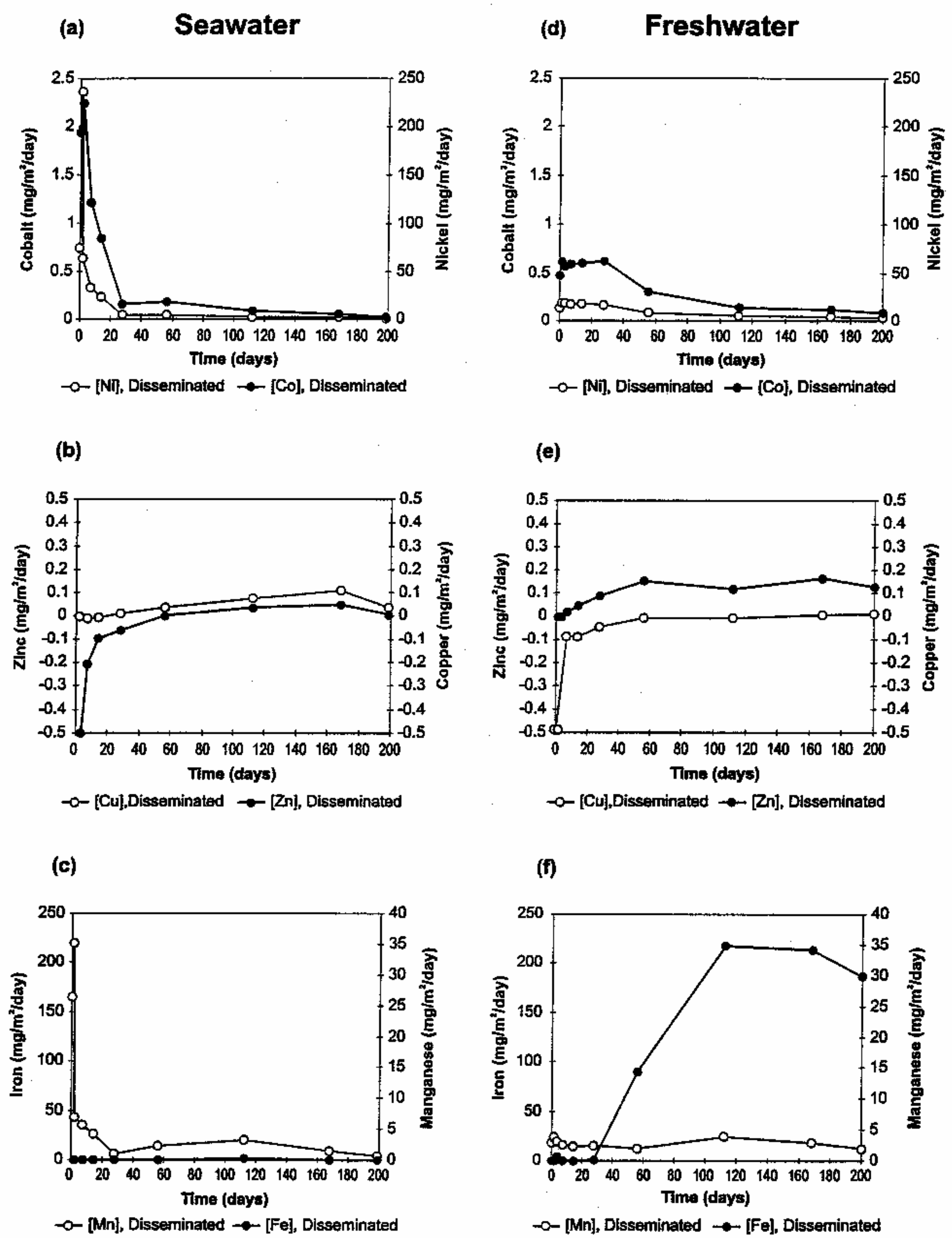
The results of the above testwork suggest that the subaqueous disposal of the Voisey's Bay sulphide tailings presents an effective method of minimizing both tailings reactivity over the long term and environmental impact.

Figure 4.2-16 **Distribution of pH, Alkalinity and Sulphate in Seawater (a,b) and Freshwater (c,d) Covers versus Time**



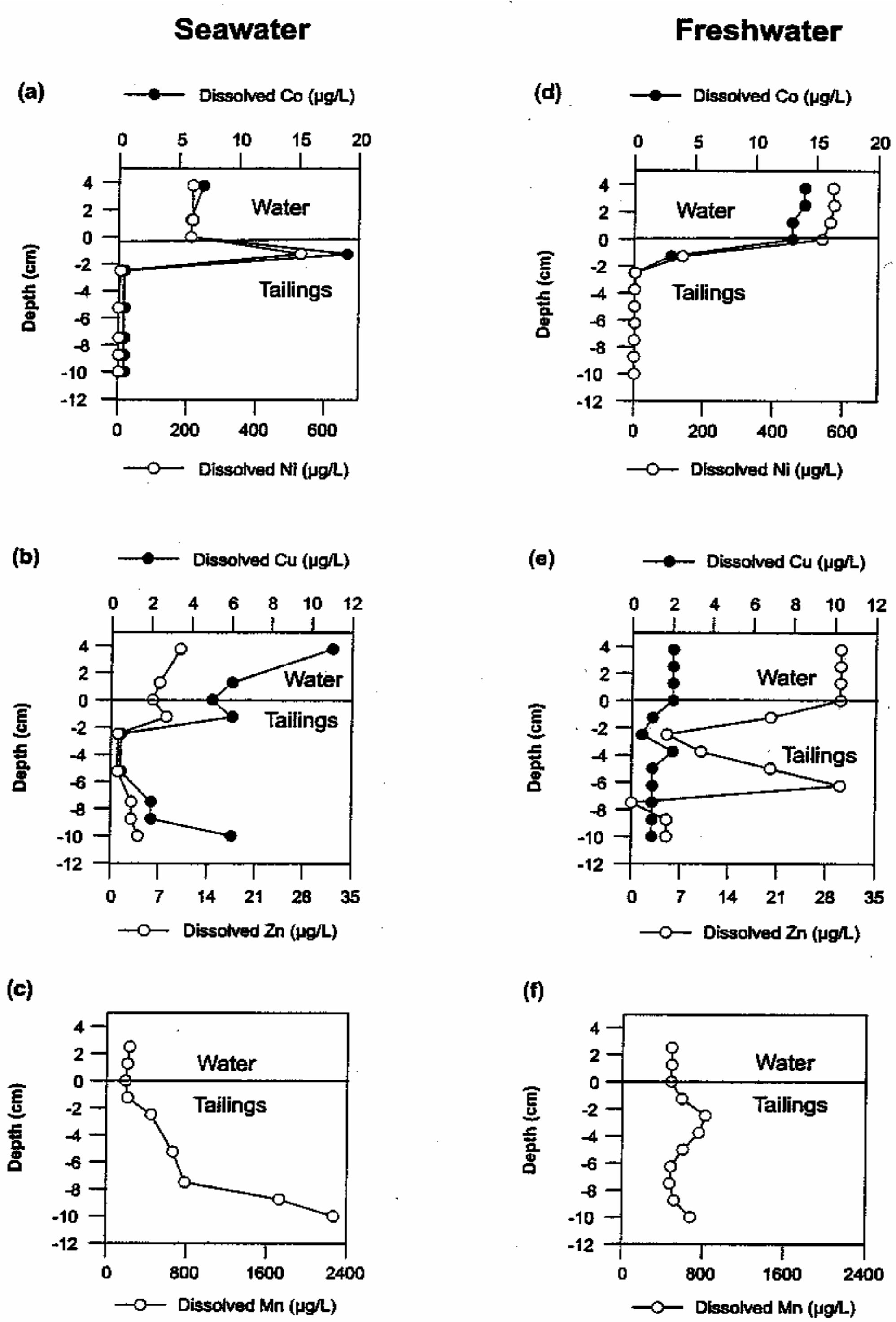
Source: Mugo *et al.* 1997

Figure 4.2-17 **Distribution of Dissolved Metals Fluxes in the Seawater (a,b,c) and Freshwater (d,e,f) Covers versus Time. Ni and Co (a,d); Cu and Zn (b,e); and, Mn and Fe (c,f)**



Source: Mugo *et al.* 1997

Figure 4.2-18 **Distribution of Dissolved Metals in the Pore Waters of Tailings Overlain by Seawater (a,b,c) and Freshwater (d,e,f) Covers. Ni and Co (a,d); Cu and Zn (b,e); and, Mn (c,f)**



Source: Mugo *et al.* 1997

Geochemical Assessment of Proposed Subaqueous Disposal in Panama

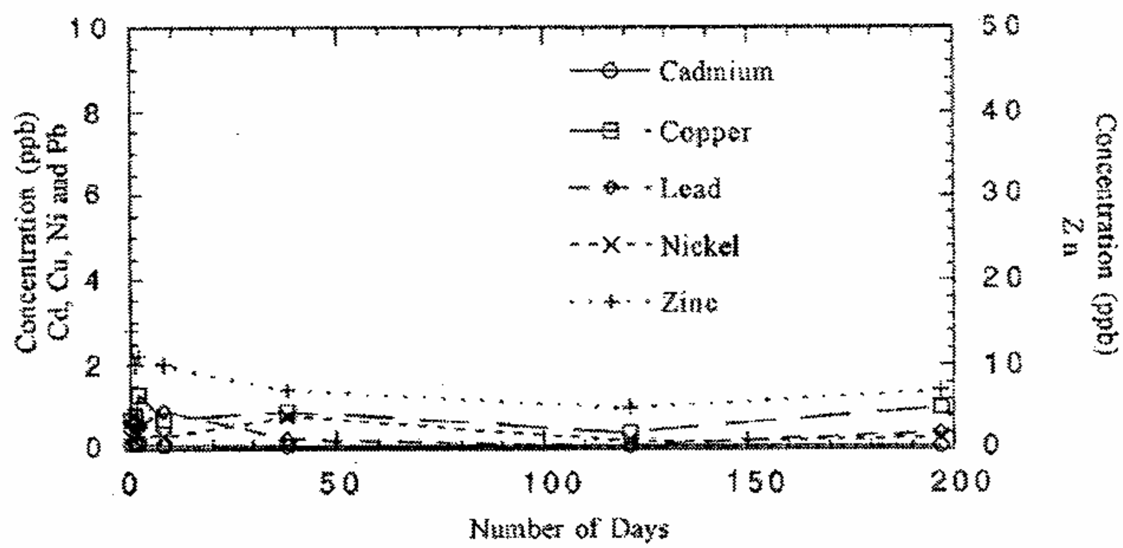
Sahami and Riehm (1999) assessed the subaqueous reactivity of sulphide-rich tailings from the Petaquilla Property, Panama in a laboratory study as part of a feasibility project. The 10-month long study was undertaken specifically to assess the short-term (days) and intermediate-term (months) chemical reactivity of tailings under a fresh water cover. The laboratory methodology was as follows:

- Pilot plant tailings were placed in a well at the base of a rectangular plexiglass tank and covered with 20 centimetres of distilled/deionized water. The water cover was circulated within the tanks to create laminar flow across the tailings/water cover interface; and
- Three independent techniques were applied to assess the submerged tailings reactivity:
  1. Time-series assessment of the supernatant chemistry.
  2. Tailings pore water chemical profiling.
  3. Direct measurement of oxygen intake.

The above techniques were applied ensemble to assess the geochemical fluxes across the interface and the processes responsible for the subaqueous reactivity. Selected results of the research by Sahami and Riehm (1999), over a 197-day monitoring time frame, are summarized below:

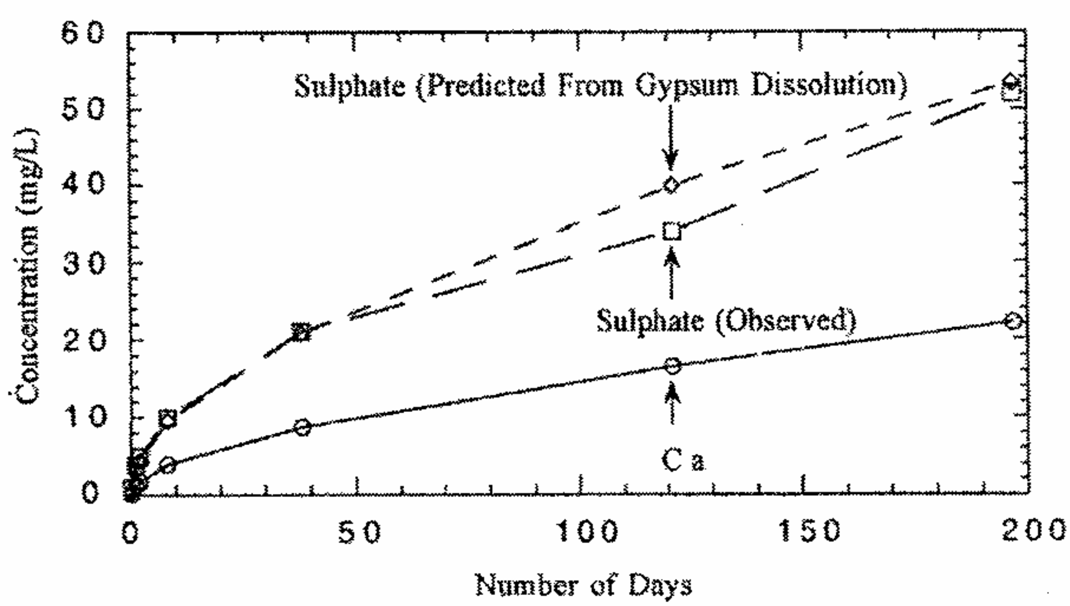
- The tailings cover pH rose from ~ 5.5 to ~ 7 in the first two weeks of the test and then stabilized at ~ 7 for the remainder. The rise in pH was considered to be due to the dissolution of calcite contained in the tailings. Concentrations of dissolved Cu, Cd, Ni, Pb and Zn over time are shown in Figure 4.2-19. Concentration of Al, Mn, Mg and Si increased over time due to the dissolution of aluminisiliates. Increases in calcium and sulphate with time (Figure 4.2-20) were controlled by gypsum solubility. There were no significant metal effluxes from the submerged tailings to the water column;
- Pore water data indicated that a fraction of ferrous iron and manganese (from reductive dissolution of iron and manganese oxides at depth) re-precipitated as oxides and oxyhydroxides in the topmost (oxic) layer of the tailings. Some iron and manganese diffused into the water cover; and
- Dissolved oxygen (DO) measurements made using oxygen microelectrodes were close to air saturation (6.7 mg/L) in the water cover to simulate field conditions. The DO profile decreased linearly in the top few millimetres of the tailings indicating oxygen diffusion into the tailings. DO was not present below a tailings depth of 2.5 cm due to oxygen consumption by organic degradation and sulphide oxidation.

Figure 4.2-19 **Water Cover Evolution of Priority Metals in the Petaquilla Tank**



Source: Sahami and Riehm 1999

Figure 4.2-20 **Water Cover Evolution of Calcium and Sulphate in the Petaquilla Tank**



Source: Sahami and Riehm 1999

Sahami and Riehm (1999) applied the results of the experiment to predict water quality impacts to a tailings pond water cover following tailings discharge. As an example, sulphate concentrations were predicted to increase by a maximum of 89 mg/L from background levels of 1 to 7 mg/L and remain below regulatory criteria for the protection of aquatic life. As such, untreated overflow from the envisaged tailings pond would likely not impact the local freshwater environment.

Subaqueous Disposal of Fresh Waste Rock – Eskay Creek

During exploration at the Eskay Creek Project from 1990 to 1991, 100,000 tonnes of potentially acid generating brecciated and sericitic waste rock was produced from an adit (Murphy *et al.* 1999). In less than a year, the pH of drainage from the pile began to decrease. A small oligotrophic alpine tarn (Albino Lake) was identified as a suitable subaqueous disposal site. The tarn averaged 7 m in depth to a maximum of 20 m depth. The total volume of the tarn was 1,080,000 m^3.

A plan was developed and implemented to add hydrated lime to the oxidized waste rock to stabilize soluble components (Higgs *et al.* 1997). Disposal of the waste rock began in 1994 and was completed in 1995. When the mine started in 1995, fresh mine fines and production waste were disposed of in Albino Lake.

In mid-1995, antimony concentrations increased from undetectable to nearly 1 mg/L. Antimony was found to be leaching from stibnite in the mine fines. The high pHs produced by lime addition increased the mobility of antimony. Ferric sulphate has been used to reduce antimony concentrations.

It was concluded that waste rock can be disposed of in natural water bodies with minimal impact.

4.2.2 FLOODING OF OXIDIZED MINE WASTES

4.2.2.1 Discussion of Theory

The use of a water cover over oxidized sulphidic tailings has been applied as a method for controlling sulphide oxidation. The potential benefits of a water cover over oxidized tailings include:

- The inhibition of further oxidation;
- Sufficient improvement of the water cover quality so as to enable the discharge of excess water without treatment after a transition period. The transition period will extend from the saturation/flooding of the tailings to some point in time when the quality of the water cover improves to allow excess water to be discharged to the receiving environment without treatment. As such, the duration of a transition period depends on site-specific conditions; and
- Reduction over time of seepage contaminant loads and treatment costs.

A strong word of caution is warranted here. Contaminants contained in the pore water and in soluble mineral phases of oxidized tailings can be released as a result of moisture saturation and the flooding of oxidized tailings. The concentrations and loads of the released contaminants can be excessive in some cases and require, or considerably add to, treatment of the water cover, the water cover effluent, and seepage over a transition period (e.g. chemical or passive treatment). The transition period may extend over a considerable length of time following flooding depending upon the tailings characteristics and other site-specific conditions. These issues and related receiving water quality concerns necessitate the site-specific scientific review of flooding proposals and regulatory input on a case by case basis. Regulatory requirements include the federal Fisheries Act, which states that no person shall carry on any work or undertaking that results in the harmful alteration, disruption or destruction of fish habitat and as such, the habitat provisions of the Fisheries Act and related policies need to be addressed in the early stages of project planning (Knapp 1999).

Predictive modelling techniques (Volume 3.0, Prediction) can be used to assess the level of effect of flooding oxidized wastes on water cover quality. The models can be applied to assess proposed flooding programs and factors such as:

- The potential benefits of adding alkalinity (e.g. lime) in the near-surface oxidized layer of tailings prior to flooding to assist in neutralizing acidity in pore water and from soluble mineral phases; and

- The potential benefits of adding a diffusion barrier over a submerged tailings surface. With or without a diffusion barrier, contaminant loads in seepage may remain elevated for some time after flooding due to increased infiltration and mobilization of contaminants.

4.2.2.2 Discussion of MEND and Relevant Research

MEND has completed a range of technical studies on the flooding of oxidized wastes at sites in Canada. These studies have encompassed comprehensive reviews of the theory related to underwater disposal, geochemical assessments, and extensive field investigations and demonstration projects. Together, the MEND reports for these studies serve as excellent references for readers seeking additional detailed information regarding the use of water covers for the decommissioning of oxidized wastes. These MEND reports are listed in Table 4.2-9 along with two relevant papers for work completed outside of MEND.

Table 4.2-9
Subject Covered under Application and Limitations

Classification	MEND Report	Application	Page No.
Flooding of Oxidized Tailings and Waste Rock	2.18.1	Review of water cover sites and research projects on flooding of oxidized mine wastes	4-76
	2.13.1a 2.13.1b	Geochemical assessment, and field demonstration of flooding using a shallow water cover at Quirke waste management area, Cell 14, near Elliot Lake, Ontario	4-76
	2.13.2a 2.13.2b 2.13.2c	Geochemical assessment and field demonstration of flooding tailings at the Solbec tailings area, Stratford, Québec	4-84
	2.13.3a 2.13.3b	Geochemical assessment of flooding tailings at the New Falconbridge Tailings Area at Falconbridge, Ontario	4-91
	2.15.1a	Flooding of pre-oxidized mine tailings: Mattabi Case Study	4-98
	-	Geochemical assessment of the flooding of tailings at the Stekenjokk Mine, Sweden	4-105
	-	Comparative geochemical assessment of the effect of cyclical submergence/exposure to the atmosphere of tailings in Lago Junin, Peru	4-106
	-	Underwater disposal of exposed sulphide tailings at the North Coldstream Mine, Northwestern Ontario	4-113
	2.36.2	Hydrogeochemistry of oxidized waste from the Stratmat Site, New Brunswick	4-113
	-	Subaqueous disposal of oxidized waste rock - Eskay Creek Mine, British Columbia	4-114
	-	Subaqueous disposal of oxidized waste rock - Skorovas Mine, Norway	4-117

MEND 2.18.1 Review of Water Cover Sites and Research Projects

MEND 2.18.1 reviewed water cover research and applications in Canada, Norway and Sweden. The design and performance of water covers applied in the field and from laboratory research projects were compiled.

MEND 2.18.1 indicates that state of the art predictive contaminant models ignore several important phenomena in water covers, including secondary mineral dissolution and metal release independent of oxidation and resuspension. Some of the work reviewed under MEND 2.18.1 hypothesized that subaqueous disposal will eventually establish a diagenetic environment for some tailings minerals with the accumulation of organic matter. However, there is uncertainty as to how long this would take to develop, and whether it would occur under all conditions.

Although water covers are a promising technology, several phenomena influencing their performance are not yet well understood. Additional research and field work was recommended in areas that include the nature and rate of accumulation of organic matter-iron hydroxide sediment at the tailings/water cover interface, and the contribution of solids resuspension to submerged tailings oxidation and water quality over the long term (completed in MEND 2.15.3).

MEND 2.13.1 Geochemical Assessment of Flooding, Quirke Waste Management Area

The Quirke Waste Management Area (WMA), shown in Figure 4.2-21, is located some 16 km north of the City of Elliot Lake, Ontario. It contains 46 million tonnes of mine tailings and waste rock and covers approximately 192 ha within a drainage basin of 275 ha (Rio Algom 1995).

Under MEND 2.13.1, a joint research program between CANMET and Rio Algom Limited in Elliot Lake, Ontario investigated the feasibility of establishing a shallow water cover on pyritic mine tailings as a close-out option for the control of acid generation and the release of metals.

The program included: laboratory lysimeter studies to investigate the oxidation and leaching characteristics of coarse and homogenized tailings with and without a water cover; a field evaluation of the partially submerged wetland/tailings basin at the Quirke Waste Management Area (WMA) with regards to its geochemical and biological controls on acid generation; and the establishment of a 65 ha flooded field demonstration project at the Quirke WMA and the evaluation of the hydrological and geochemical control parameters (Cell 14, Figure 4.2-21).

MEND 2.13.1a reports that column leaching studies were conducted to evaluate the oxidation and leaching characteristics of pyritic uranium tailings with and without limestone amendments and crushed waste rock, under unsaturated and submerged conditions. These experiments were specifically designed to determine the acid generation characteristics and subsequent metal and radionuclide releases of test samples subjected to submersion underwater and exposed conditions. The leaching experiments were conducted using PVC cylindrical column lysimeters filled with total mill tailings, coarse tailings, tailings amended with 7.5% (w/w) limestone of varying screen sizes, and crushed waste rock samples obtained from Quirke Mine. The experiments provided unsaturated conditions for leaching of test samples as well as a submerged condition for coarse tailings where a shallow water cover, 0.45 m in depth, was established using distilled water. The experiments were conducted at CANMET in Elliot Lake from April 1989 to June 1993. Selected results are reiterated below.

<u>For Unsaturated Conditions:</u>

- Unsaturated coarse tailings without limestone amendment oxidized readily and produced highly acidic drainage (pH ~ 1.0 - 2.0 ; acidity ~ 12,000 - 30,000 mg $CaCO_3$/L);
- Coarse tailings amended with coarse limestone (screen size -6.3 mm) also produced highly acidic drainage (pH ~ 2.5 - 3.0; acidity ~ 4,000 - 12,000 mg $CaCO_3$/L), but its onset was delayed by approximately one year in comparison to coarse tailings without limestone. Acidic drainage occurred with 90% of the available alkalinity still remaining in the sample; and
- Coarse tailings amended with pulverized wet ground limestone did not produce any acidic drainage during the entire four-year study period. Oxidation and acid generation processes were active in these tailings as well, but the fine grained limestone provided complete acid neutralization and acidic drainage was prevented.

<u>For Submerged Conditions:</u>

- Submerged coarse tailings did not produce any pore water acidic drainage for the first three years, but in the fourth year a weak acidic drainage (pH ~ 5.5 - 7.0; acidity ~ 10 - 20 mg $CaCO_3$/L) was observed;
- The submersion of tailings underwater decreased cumulative pore water acidity, sulphate and iron loadings by factors of 295, 200 and 950 respectively. The submersion of coarse tailings also resulted in increased mobility and release of Ra-226 in the pore water when tailings developed acidic conditions at the surface after they were depleted of available alkalinity and gypsum. There was a 10 fold increase in the cumulative loading of Ra-226 in the pore water over that of unsaturated coarse tailings;
- The increased drainage of Ra-226 in the pore water was also accompanied by that of iron, both occurring near the end of the fourth year, where as gypsum was completely removed from the tailings during the first three months of leaching; and
- Surface water above submerged tailings contained low concentrations of dissolved metals, when the water cover above the tailings was continuously maintained in a well oxygenated

condition by fresh water inflow and where both surface and pore water flows were present in equal proportions. The dissolved Ra-226 concentration in the surface water was also low, ranging between 30 and 140 mBq/L.

MEND 2.13.1b reports on diffusion lysimeter studies carried out to determine the surface oxidation and mass release characteristics of underwater deposited pyritic uranium tailings. The studies examined two cases:

- Unoxidized tailings that had been submerged for more than 12 years; and
- Partly oxidized tailings.

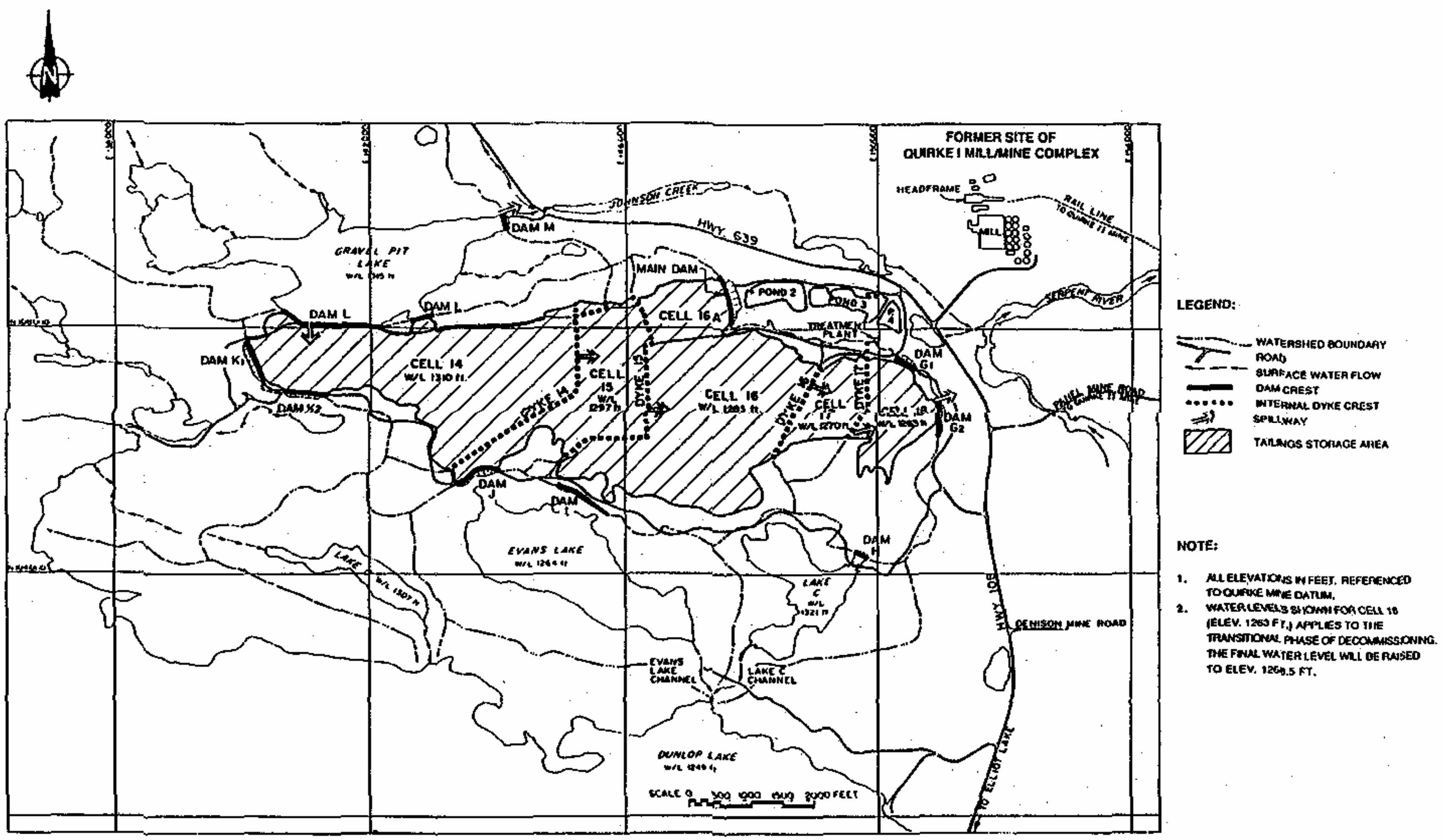
The studies were conducted using aquarium type Plexiglas™ lysimeters for unoxidized tailings obtained from the Quirke mill and kept underwater in the laboratory since 1982, and weathered tailings obtained from the Quirke waste management area and deposited underwater in the laboratory in 1990. A water cover, approximately 0.2 m in depth, was provided using distilled water.

Selected results of the study are reiterated below.

For Unoxidized Tailings:

- Unoxidized tailings, deposited underwater in 1982, oxidized very slowly and a narrow oxidized and iron hydroxide precipitate zone, 2 - 3 cm in thickness, was formed at the surface of the tailings at the water-tailings interface. Mobilization and release of iron to the surface water, under oxidizing conditions, resulted in its precipitation and covering of tailings with a layer of ferric hydroxide;
- The oxidized zone at the tailings surface released low acidity (5 - 20 mg $CaCO_3$/L), low concentrations of Mn and Pb, and high concentrations of Ca, Mg and Ra-226 to the surface water. Gypsum dissolution contributed to the increased release of Ca and, to a certain extent, Mg;
- The dissolution of radium (Ra) from the tailings surface, and most significantly under acidic conditions, was a dominant factor in increased Ra mobilization and its release to the surface water. Solubility of gypsum, when present, and hence the sulphate ion concentration controlled the release of Ra and its surface water concentration that decreased with increasing sulphate concentration. Diffusion of Ra from the tailings pore water was low (less than 10%) compared to mass dissolution from the surface of the tailings. The concentration of other metals in the surface water were low as a result of iron hydrolysis and precipitation; and

Figure 4.2-21 **General Arrangement Plan for Quirke Mine Waste Management Area Rio Algom Limited**



Source: Rio Algom 1995

- Tailings pore water contained mainly dissolved gypsum, Mg, and Ra, and low concentrations of Fe, Mn and Pb. Long-term exposure of the tailings to natural and fluorescent room light in the lysimeter resulted in the formation of an algae layer on the surface of the tailings, which slowly contributed to oxygenation of the tailings substrate and decrease in the previously established anoxic conditions as well as sulphate reduction.

<u>For Partly Oxidized Tailings:</u>

- Similar to unoxidized tailings, further oxidation of weathered and partially oxidized tailings underwater was very slow and limited to the near surface zone of the tailings;
- The weathered tailings released high acidity (~ 1700 mg $CaCO_3$/L) and high metal concentrations (e.g. Fe ~ 550 mg/L; Al ~ 110 mg/L) to the pore water but their diffusion to the surface water was moderate to low (acidity ~ 10 - 20 mg $CaCO_3$/L; Fe ~ 0.5 mg/L; Al ~ 2.0 mg/L);
- Iron was dissolved from weathered tailings but its transfer to the surface water was also low, as it hydrolyzed and precipitated forming an iron hydroxide sink layer at the tailings surface. Mass dissolution of gypsum and Ra from the tailings surface resulted in increased concentrations of Ca, Mg and Ra (10,000 - 32,000 mBq/L) in the surface water. Transfer of other metals was low; and
- For weathered tailings under high acidic conditions, the transfer flux of Ra was significantly higher (approximately 50 times) and those of Ca, SO_4 and Mg were significantly lower, (e.g. decrease in transfer coefficients by 2 to 3 orders of magnitude), than those for unoxidized tailings.

In both cases, the slow oxidation of underwater deposited tailings at the surface as well as the dissolution and release of metals and radionuclides from the oxidized surface have been important factors in determining the surface water quality. The data suggest a need to examine the use of diffusion or oxygen barriers above the tailings surface for further controlling the surface oxidation as well as the release of metals and radionuclides to the surface water column.

The decommissioning and tailings flooding concept was tested in Cell 14 of the Quirke WMA. The cell has an area of approximately 65 ha and provided a suitable opportunity to study the effect of tailings flooding. A water pond could be maintained through the use of containment dykes and an ample water supply was available from adjacent Gravel Pit Lake to flood the tailings surface and maintain the water cover. The full scale flooding of Cell 14 commenced in October 1992. The overall results of the extensive program as reported by Davé and Vivyurka (1994) are as follows:

- The laboratory lysimeter study results indicated that without a water cover, the pyritic tailings oxidized rapidly and generated acidic drainage. The oxidation rate for well drained coarse tailings was about two orders of magnitude higher than that for homogenized tailings which had a higher degree of moisture saturation;

- The results of the Quirke WMA field program demonstrated that acid generation in the pyritic tailings could be controlled using a shallow (0.5 to 1 m depth) water cover; and
- The Quirke WMA wetland/water cover system was found to be effective in controlling acid generation. The drainage from this site had a pH > 7, and dissolved iron concentration of ~0.1 mg/L.

Kam *et al.* (1997) undertook an interim performance assessment of the Quirke Mine Waste Management Area (WMA). The Quirke Mine uranium tailings contain about 5% pyrite and have low uranium concentrations. The approach to decommission the WMA was selected after the comprehensive review of decommissioning options. The selected option, which was subsequently implemented, was to submerge the tailings by constructing a series of internal dykes within the WMA to create cells which were then flooded to submerge the tailings (Figure 4.2-21).

The WMA covers an area of 192 ha and consists of five cells separated by internal dykes with an average difference in the operating water level between cells of 3.5 m. The difference in elevation between the topmost cell (Cell 14) and the lowest cell (Cell 18) is about 40 metres. Spillway on the internal dykes allow excess water to cascade to the clarification pond located at the east end of the WMA. The effluent is treated using lime and barium chloride and settling ponds prior to release to the receiving environment. Additional fresh water to recharge the water covers in the cells is available from Gravel Pit Lake located north of Cell 14 (Figure 4.2-21). The decommissioning profile of the terraced WMA is shown in Figure 4.2-22.

As part of the Environmental Assessment Review Process, which governed all federally regulated facilities until 1995, and at the request of the Atomic Energy Control Board, the proposal to decommission the Quirke Mine WMA along with three other tailings management facilities in the Elliot Lake area, was referred to an independent Environmental Assessment Panel for public review (Kam *et al.* 1997). The Panel's report concurred that in the Elliot Lake environment, flooding of tailings is the most effective way to minimize long-term environmental impacts (CEAA 1996).

Kam *et al.* (1997) reviewed the geotechnical and hydrogeological aspects of Cell 14 as well as the water quality in the cell. Cell 14 covers approximately 65 ha and is contained by 3 engineered perimeter dams and Dyke 14. Prior to the construction of the cell, the tailings surface was exposed. Its surface required regrading to provide a minimum 0.6 m deep water cover. Limestone was applied to the regraded tailings surface. The cell was flooded in 1992 and has since been maintained at, or near, the operating water cover level with fresh water input from Gravel Lake. Kam *et al.* (1997) reviewed the extensive monitoring database for Cell 14. Their findings with respect to water quality and degree of success attained in inhibiting acidic drainage generation in the flooded tailings are summarized as follows:

- Cell 14 has a dominant downward flow with very little or no surface water flow. As a result, the surface water quality has remained excellent with many parameters meeting Provincial Water Quality Objectives. The pH has remained in the 6 to 7 range with the alkalinity level holding steady at around 7 mg/L;
- The sulphate concentration has continuously declined since flooding. Sulphate due to acid generation is very low. Water quality data indicate that there is little diffusion and upward leaching of metals into the water cover; and
- Cell 14 now supports a variety of wildlife including benthic life, fish and birds.

Kam *et al.* (1997) also assessed changes in the Cell 14 tailings pore water. Table 4.2-10 presents data for shallow and deep piezometer monitoring stations located upstream and downstream of Dyke 14 (Figure 4.2-21).

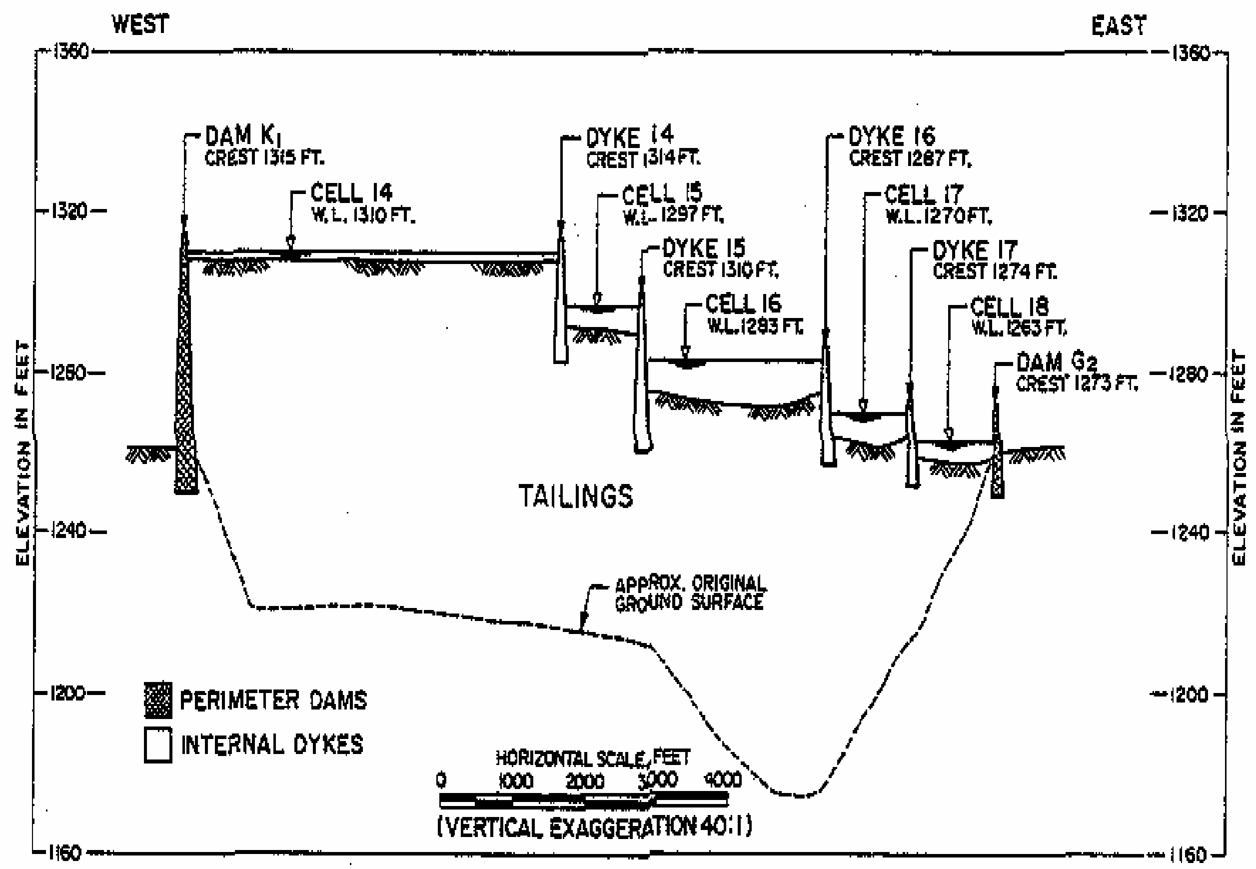
The data demonstrate that acidity has declined significantly from peak measurements prior to flooding of 12,500 mg/L. It was predicted that essentially all acid pore water present in the shallow topmost layer of the tailings will be flushed from the cell over the short term. The rapid flushing of historic (pre-flooding) acidic pore water in Cell 14 will increase short-term effluent treatment requirements. However, after the historic acidic pore water is flushed from the WMA, seepages from the cells should have no impact on the operation of the decommissioned WMA.

Table 4.2-10
Cell 14 Tailings Pore Water Quality Data 1992 to 1996

Parameter	Year	Piezometer Number and Location			
		Upstream Dyke 14		Downstream Dyke 14	
		6C Shallow	6A Deep	7D Shallow	7A Deep
pH (Units)	1992	3.2	6.9	3.7	7.5
	1993	3.6	6.4	3.6	7.2
	1994	3.3	6.7	3.6	6.9
	1995	3.1	7.4	3.5	6.1
	1996	4.2	7.0	-	-
Acidity (mg/L)	1992	3,500	24	3,180	13
	1993	685	13	830	18
	1994	350	8	105	10
	1995	220	6	61	95
	1996 (To Oct)	140	35	-	-

Source: Kam *et al.*1997

Figure 4.2-22 **Schematic Decommissioning Profile**



Source: Kam *et al*. 1997

In the Elliott Lake area, water covers have been applied at the Quirke, Panel, Spanish American and Stanleigh WMA's, and the Denison tailings management area. In all cases, the tailings were originally deposited in rock-rimmed basins where engineered, zoned embankment dams were constructed to close-off topographic low points along the perimeters. Berthelot *et al.* (1999) report that the water quality limits used to evaluate effluent quality from the flooded tailings basins include: Atomic Energy Control Board Decommissioning License criteria, Federal Metal Mining Liquid Effluent Regulations and Guidelines, operational loading criteria, and Provincial Surface Water Quality Objectives.

MEND 2.13.2 Geochemical Assessment of Flooding, Solbec Tailings Area

Amyot and Vezina (1999) reviewed the flooding of the historic Solbec Cu-Pb-Zn mine tailings basin located near the village of Stratford, about 200 km east of Montréal, Québec. The mine operated from 1962 to 1970. The mill operated from 1962 to 1977 to process 1.9 M tonnes of Solbec mine ore as well as more than 2.9 M tonnes of ores from other mines in the area. In total, 4.8 M tonnes of massive sulphides consisting of chalcopyrite, sphalerite, galena, pyrrhotite and pyrite were processed, with tailings disposed in a 66 ha tailings basin. The total tailings volume is estimated to be 4.2 M tonnes. When the mill closed in 1977, about 20 ha of tailings were left submerged in the north end of the basin.

In 1983, a panel from the Québec Ministry of Environment (GERLED 1985) reviewed the water quality data for the Solbec tailings pond – the data indicated that sulphide oxidation was occurring. The Solbec tailings basin and other historic acid generating sites in Québec, were classified as posing a potential high risk to the environment. In 1986, Québec's then Ministry of Energy and Resources sponsored a study on reclamation options for the Solbec tailings basin. The study report, dated June 1987, presented flooding as the optimal solution from economic and environmental perspectives. The following research was then undertaken to assess the viability of flooding the tailings:

- The submergence and monitoring of oxidized tailings in 10 cm diameter columns to assess the effect of flooding, and the use of 1.22 m diameter cylinders sunk into the tailings down to the water table as part of an experiment to simulate *in situ* flooding;
- Monitoring piezometer platforms were installed in the tailings basin to monitor water quality in the oxidized mine tailings on surface, and in the underlying sequence of unoxidized tailings, peat, till and bedrock;
- A study to ensure that the water cover would be maintained over the long term. The study reviewed the water balance for the basin and provided the geotechnical information required for the design of the tailings water cover containment dams; and

- Another study demonstrated that the leaching of the oxidation by-products from the oxidized tailings would be offset if limestone was added to the tailings. In December 1993, tailings flooding after the addition of lime, was recommended as the reclamation method.

Other studies that were conducted from 1990 to 1993 to refine the closure plan included a study (MEND 2.13.2b) to assess the depth of water required to inhibit acid generation due to wave action and the resuspension of particles. The study used a longer fetch and higher wind speed than those at the Solbec site, and provided guideline recommendations for the water cover depth. Another study assessed the required liming rate and application method (MEND 2.13.2c) and recommended an application rate of 118 tonnes/ha and that the lime be mixed into the tailings to an average depth of 15 cm. A laboratory study which examined the effect of tailings flooding on microbiological activity (MEND 2.13.2c) found that maintaining non-oxidized tailings underwater prevents colonization by *Thiobacillus ferrooxidans*-type bacteria, and the artificial flooding of unoxidized and oxidized tailings combined with lime inhibits microbial activity.

The flooding of the tailings with a minimum 1-metre depth water cover was made possible by constructing two dams (Figure 4.2-23). The water level is controlled by a 4 m wide concrete spillway designed to handle the 1 in 100-year flood. To minimize the height of the water containment dams, about 5 ha of higher elevation tailings located at the south end of the pond were relocated within the impoundment.

The tailings surface was limed prior to flooding. Hydrated lime ($Ca(OH)_2$) was added to the tailings pond in areas where the tailings were already submerged. In areas where the tailings were exposed, calcium carbonate ($CaCO_3$) dust and granules were mixed into the tailings surface by plowing to a depth of 0.3 m - this operation was difficult and resulted in an irregular distribution of lime and a lime application rate greater than that recommended in MEND 2.13.2c - on average, 230 tonnes/ha were applied. The water level was allowed to rise after the dams were constructed. The water pond covered the tailings at their highest elevation (329 m) in September 1995. The pond level continued to rise, fed by natural inputs and reached the spillway invert elevation (330 m) in February 1996.

The environmental monitoring program for the tailings basin was developed under MEND. The monitoring program assessed the water cover, tailings, effluent quality, water balance, effects of waves, ice and erosion on the tailings, and microbial populations. The program was initiated in the fall of 1994 with additional sampling in the spring, summer and fall of 1995, 1996, and 1997. Sampling was also conducted in 1998 (MEND 2.13.2d).

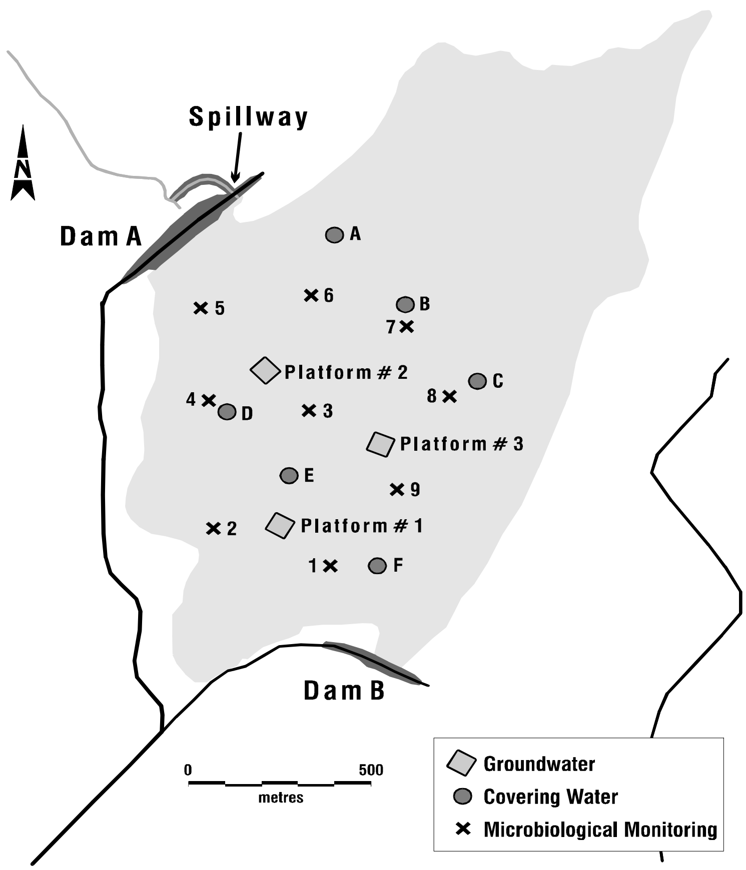
Six surface water sampling stations (A-B-C-D-E-F) oriented roughly perpendicularly to the axis of Dam A (Figure 4.2-23) were used to collect water cover samples at surface, at mid-depth, and ± 150 mm above the tailings surface. The pore waters present in each stratigraphic unit (oxidized tailings at surface to bedrock) were collected at three groundwater monitoring stations located within the tailings pond (Figure 4.2-23) - each platform had five monitoring wells. Screened sampling tubes allowed sampling of pore water from each stratigraphic unit including the bedrock (Figure 4.2-24). The groundwater data collection program was reduced when Platform 2 collapsed during the winter of 1997-98. Tailings samples were obtained by coring, and used to provide culture media that were seeded to permit the isolation, identification and enumeration of bacteria.

Selected monitoring data for the time period from Fall 1994 to Fall 1998 are reproduced in Table 4.2-11. With regards to the data, Amyot and Vézina (1999) report that:

- The water cover pH rose from a slightly acid pH in the fall of 1994 to very alkaline levels in the spring of 1995 following a thaw and lime dissolution. The pH has subsequently hovered around neutral. Following the complete flooding of the tailings in the Fall of 1995, the pH of the pore water in the layer of oxidized tailings decreased slightly, and then returned to neutral range;
- The dissolved iron concentration of the water cover has decreased and seems to have reached a stable low range. The dissolved iron concentration in the oxidized tailings layer initially increased as the tailings and oxidation by-products were progressively saturated – the iron concentration peaked at 373 mg/L in the fall of 1995 and has since returned to < 100 mg/L;
- Dissolved zinc concentrations in the water cover have decreased to low values (e.g. < 0.01 mg/L). Zinc concentrations in the oxidized tailings pore water rose to a maximum of 23.6 mg/L (spring 1996) and has decreased to about 2 mg/L. Concentrations in the unoxidized tailings and peat layers have reduced to about 0.2 and 0.1 mg/L Zn, respectively; and
- The concentration of dissolved copper in the water cover has declined to low values (e.g. < 0.01 mg/L). The dissolved copper concentration in the pore water in the oxidized tailings layer has also decreased from about 1 mg/L to < 0.01 mg/L. The decline in dissolved copper concentrations was expected as a result of the lime addition and a pH rise.

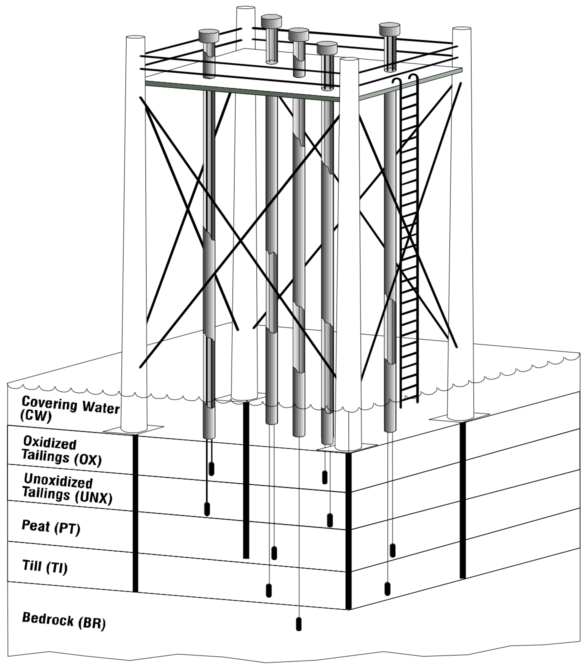
Amyot and Vézina (1999) report that the gradual saturation of the tailings caused metals in solution to migrate from the oxidized tailings layer to the underlying unoxidized tailings layer and, in the case of iron and zinc, even to the peat layer unit below the tailings. No other stratigraphic zone was affected and the phenomenon is now regressing. This seems to indicate that oxidation was inhibited by flooding the tailings. The water cover effluent quality meets the effluent requirements of Québec Directive 019 (Ministry of Environment and Wildlife) and those for drinking water.

Figure 4.2-23 **Solbec Tailings Pond and Sampling Stations**



Source: Amyot and Vézina 1999

Figure 4.2-24 **Piezometer Monitoring Platform**



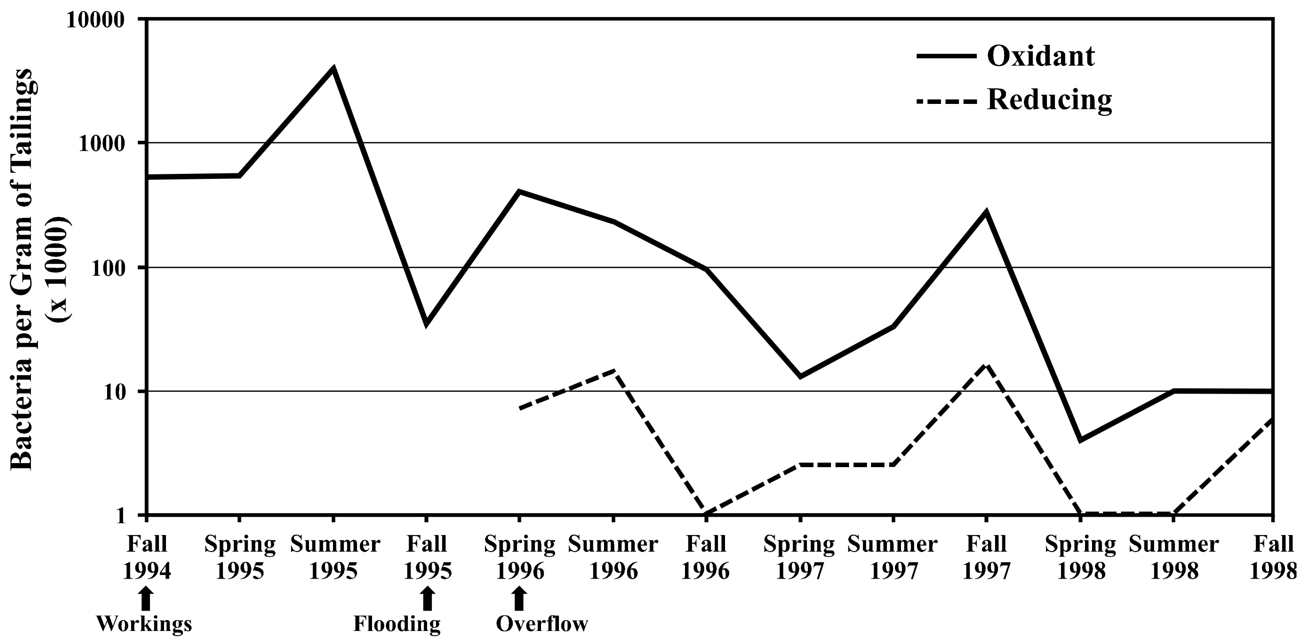
Source: Amyot and Vézina 1999

As part of the research to confirm the suitability of flooding, microbial species dominant in the acid generation process were identified in tailings samples, and species of *Thiobacillus* and other heterotrophic and acidophilic bacteria were observed. Figure 4.2-25 indicates changes in the microbial populations of oxidant species. Amyot and Vézina (1999) report that the oxidant microbial populations that were initially high have progressively decreased.

The populations, which stood at just over 500,000 individuals per gram of tailings in 1994, reduced to 100,000 individuals per gram by the summer 1998. Their population is highly sensitive to temperature, a phenomenon that was observed in the summer of 1995, when numbers rose to nearly four million individuals per gram of oxidized tailings.

One explanation for the decreasing trend in the oxidant microbial populations as well as the regression in concentrations of metals such as iron, zinc and copper is the appearance of sulphate-reducing bacteria - these bacteria were first observed in samples in spring 1996.

Figure 4.2-25 **Oxidant Microbial Populations in Tailings**



Source: Amyot and Vézina 1999

Table 4.2-11
Surface Water and Pore Water Quality Data (1994-1998) – Solbec Flooded Tailings

Horizon	Parameter	Fall 1994	Spring 1995	Summer 1995	Fall 1995	Spring 1996	Summer 1996	Fall 1996	Spring 1997	Summer 1997	Fall 1997	Summer 1998	Fall 1998
Water Cover	pH	5.0	10.7	6.7	7.6	7.8	7.2	7.3	7.3	7.7	7.8	7.6	7.2
	Fe	142	2.9	1.80	0.35	0.29	0.17	0.30	0.41	0.05	1.4	0.10	0.05
	Zn	0.83	0.11	0.02	0.04	0.01	0.02	0.19	0.03	<0.01	0.02	0.03	<0.01
	Cu	0.49	<0.01	0.01	0.01	<0.01	0.02	0.03	<0.01	<0.01	0.07	0.03	<0.01
Pore Water in Oxidized Tailings	pH	6.8	7.0	6.4	6.5	6.1	6.4	6.1	6.2	7.0	7.5	6.6	6.7
	Fe	555	99.8	183	373	343	130	150	132	18	38	70	29
	Zn	2.4	17.0	17.6	20.2	23.6	10.0	9.3	7.8	1.8	2.0	1.4	1.4
	Cu	0.18	1.4	1.0	0.23	0.59	0.54	0.50	0.05	<0.01	0.02	<0.01	<0.01
Pore Water in Unoxidized Tailings	pH	7.2	8.6	8.1	8.4	8.3	7.7	8.2	7.4	8.6	8.0	8.3	8.7
	Fe	0.15	0.01	0.04	0.01	49.4	0.04	0.06	0.05	0.24	0.26	0.43	0.50
	Zn	2.0	0.54	0.31	0.07	0.83	0.25	0.12	0.28	0.12	0.17	0.20	0.10
	Cu	0.08	0.19	0.09	0.05	0.04	0.06	0.05	0.04	<0.01	0.05	0.10	0.10
Pore Water in Peat Layer	pH	6.6	8.1	6.9	6.8	6.8	6.5	6.8	6.6	7.3	7.3	6.6	6.6
	Fe	0.01	0.01	0.10	0.85	7.8	29.9	34.9	10.4	0.91	0.54	0.09	2.2
	Zn	0.56	0.09	0.31	0.06	0.10	0.24	0.08	0.09	0.03	0.07	0.10	0.09
	Cu	0.23	0.05	0.07	0.02	<0.01	0.04	0.04	0.03	<0.01	0.01	<0.10	<0.01
Pore Water in Till	pH	7.4	7.6	7.6	7.8	7.7	7.2	7.7	7.4	7.7	7.4	7.4	7.2
	Fe	<0.01	0.12	0.15	0.06	0.03	0.02	0.15	0.19	0.31	0.98	0.27	1.9
	Zn	0.16	0.45	0.02	<0.01	0.63	0.03	0.18	0.03	0.02	0.03	0.01	0.03
	Cu	0.20	0.12	0.04	0.01	0.01	0.03	0.02	0.02	<0.01	0.02	<0.01	<0.01
Groundwater in Bedrock	pH	7.7	8.5	7.6	7.5	8.2	7.4	8.2	7.4	7.9	7.4	6.8	7.6
	Fe	<0.01	0.01	0.35	0.08	<0.01	<0.01	0.03	0.03	<0.01	<0.01	0.09	<0.01
	Zn	0.01	<0.01	0.03	0.01	<0.01	0.03	0.02	0.03	0.02	0.06	0.03	<0.01
	Cu	<0.01	0.01	0.03	<0.01	<0.01	0.03	0.02	0.01	<0.01	<0.01	<0.01	<0.01

Note: Metal concentration in mg/L
Source: Adapted from Amyot and Vézina 1999

MEND 2.13.3 Geochemical Assessment of Flooding, Falconbridge New Tailings Area

DeVos *et al.* (2000) and Hall (1999) reviewed the development and implementation of the closure plan for Falconbridge Limited's "New" Tailings Area (NTA) at Falconbridge, Ontario. The closure plan involves the flooding of tailings through the construction of dams to maintain ponds, and the relocation of exposed tailings from elevated beaches to the ponds. The tailings were relocated and have remained flooded since 1996.

The New Tailings Area (NTA) was constructed approximately 2 km from the Falconbridge Mill. Prior to the closure works program, the 90 ha basin was contained by the surrounding topography and Dams 1, 6, 7 and 7A (Figure 4.2-26). The tailings were deposited in the southwest corner of the NTA with the tailings beach extending to the edge of a pond. About 3.2 million tonnes of tailings (averaging 7% S) were deposited in the NTA between 1978 and 1984.

Hall (1999) reports that in the mid-1980's, an effort was made to minimize the potential for acidic drainage generation from the NTA. This included the construction of Dam 12 (Figure 4.2-26) over tailings to contain tailings solids in the Upper Terrace area and allow water to drain to the Lower Terrace. Milling practice was modified to produce a pyrrhotite concentrate which was then disposed in a separate tailings area, and a low sulphur (1% S) tailings. From 1985 to 1988, the low sulphur tailings were placed upstream of Dam 12 in the Upper Terrace area to provide a cover over the sulphide tailings. It is estimated that 1.1 million tonnes of low sulphur tailings were produced during this time period, with an unspecified tonnage of the coarse fraction removed for mine backfill. Low sulphur tailings deposition ceased in 1988 when the Falconbridge Mill closed.

The closure plan for the NTA was developed in the 1990's. The plan was based on excavating exposed, unsaturated tailings in the Upper Terrace and then placing them under a water cover within the same terrace. The tailings in the Lower Terrace would also be flooded. The closure plan was developed based on numerous studies which included additional tailings characterization, geochemical modelling to predict sulphide oxidation rates in the tailings, particularly in the vadose zone, and effects on the quality of the water cover. The feasibility of flooding the NTA and maintaining a suitable water balance was also assessed. It was determined that a suitable water pond could be maintained in the Upper Terrace through the construction of dams (Dams 1, 1B, and a new Dam 12) and a spillway at Dam 1B (Figure 4.2-27).

Tailings relocation was necessary as the natural topography prevented the flooding of all exposed tailings in the Upper Terrace. At the normal water level of 299.5 m, approximately 12 ha of beached tailings would remain exposed - these exposed tailings were relocated underwater by truck and excavator in the fall of 1996, and by dredging in September and October of 1997. The dredged tailings were re-deposited immediately upstream of Dam 12. Approximately 850 tonnes of lime were spread on the surface of the tailings prior to final phase of dredging (Hall 1999).

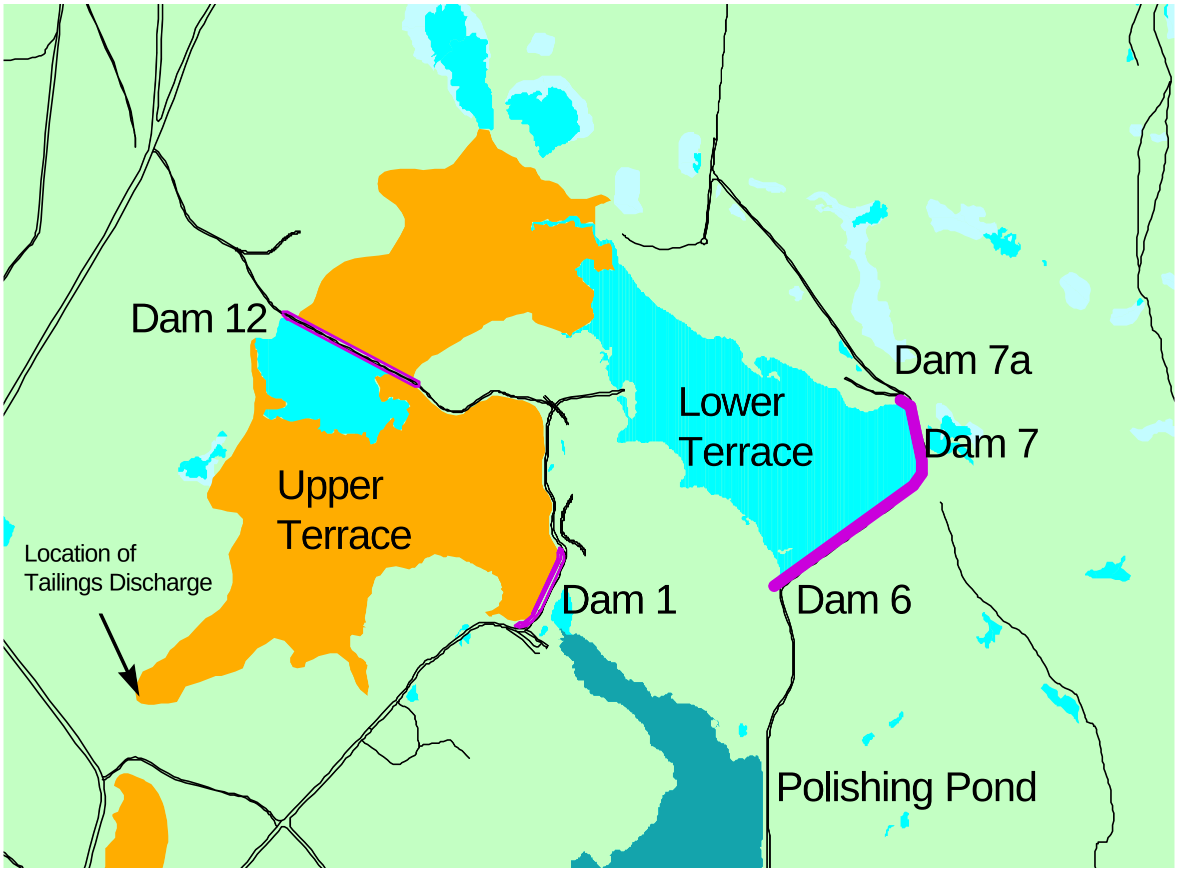
The difference between the maximum tailings elevation and the normal water level for the Upper Terrace provides a minimum water cover depth of 0.5 m. Hydrologic modelling has indicated that this is sufficient to ensure that tailings exposure episodes will be prevented or at least very infrequent and limited in duration. Maximum wave heights on the pond were predicted for a 1 in 100-year wind hourly velocity of 29 m/s. For an estimated fetch of 0.82 km and a fetch reduction factor of 0.73, the maximum wave height was predicted to be 0.55 m. As such, the extremely unlikely coincidence of maximum water level (derived from a PMP storm event) and a 1 in 100-year return wind event from the critical direction would result in a wave top elevation of 300.9 m which is below the dam crest elevation.

The closure plan included the flooding of the sulphide tailings in the Lower Terrace. To maintain the tailings in a flooded state, the tailings located above the maximum tailings elevation of 292.3 m were dredged and relocated to areas having lower elevations. It was determined that the use of a 1 m nominal deep water cover over the tailings in the Lower Terrace would be practicable. The use of a 1 m deep water cover reduces the likelihood of tailings exposure during extended drought periods to a negligible value (Hall 1999). The depth of the water cover was selected based on the ease of implementation and a minimal chance of tailings exposure.

It was proposed that at closure the existing decant tower at Dam 6 would be modified to passively control the pond water level. Flood routing has indicated that a daily storm event with a 50-year return period would be contained without a release through the spillway. The maximum wave height in the Lower Terrace reservoir was predicted for a 1 in 100-year wind velocity of 29 m/s, as used for the Upper Terrace. For a maximum fetch of approximately 1.2 km, the predicted maximum wave height was 0.70 m. Should the maximum wave height occur while the water level is at the peak (following the PMP), the top of wave elevation would match the elevation of the rip-rap.

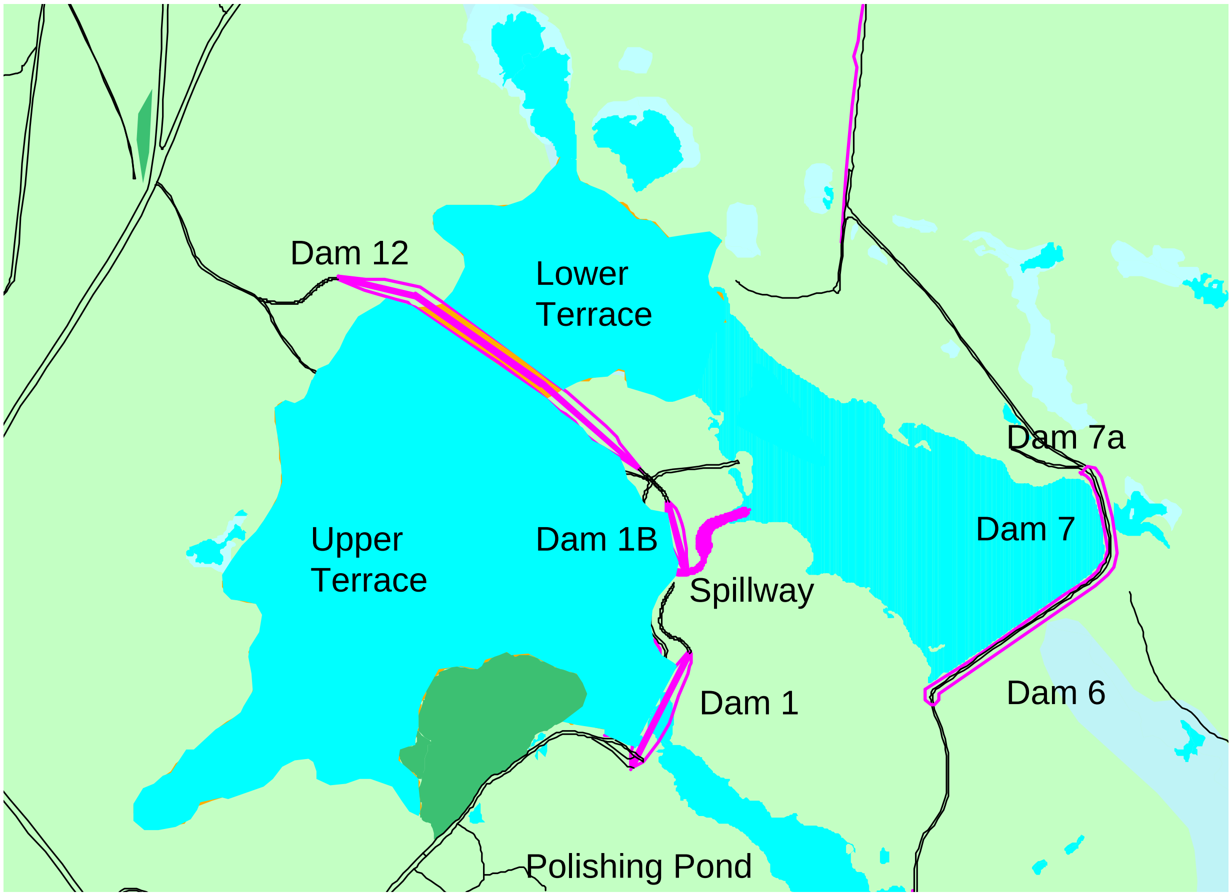
Hall (1999) states that a key hydrological design consideration for the closure of the NTA was the minimum depth of water to permanently submerge the tailings. This aspect was addressed in a drought analysis that included the development of a statistical model to simulate a continuous 200-year record of weather conditions on a monthly basis. The simulated record was input into a water budget, and applied to estimate the draw-down in the Upper and Lower Terraces. The drought analysis concluded that draw-down would not exceed 0.4 m in 200 years in either the Upper and Lower Terraces.

Figure 2.4-26 **Falconbridge New Tailings Area Prior to Flooding**



Source: Hall 1999

Figure 4.2-27 **Falconbridge New Tailings Area (1999)**



Source: Hall 1999

The chemical composition of the water cover in the Upper Terrace was predicted using the PHREEQC geochemical speciation and mass transfer model based on a scenario involving the mixing of oxidized and partially-oxidized tailings, pond water, and added lime. In the model, the water cover was allowed to equilibrate with respect to atmospheric carbon dioxide, oxygen and ferrihydrite. Amorphous aluminum hydroxide, nickel hydroxide, and calcite were allowed to precipitate upon saturation in the solution. The maximum pond pH of 8 corresponds to the dissolution of approximately 61 tonnes of lime in a 545,000 m^3 pond. The upward diffusion of metals from the upper 1 m of tailings to the pond was modelled using the POLLUTE program which accounts for:

- Diffusion of contaminants from the tailings into the pond water;
- Desorption of contaminants from the tailings; and
- Dilution with the pond surface water.

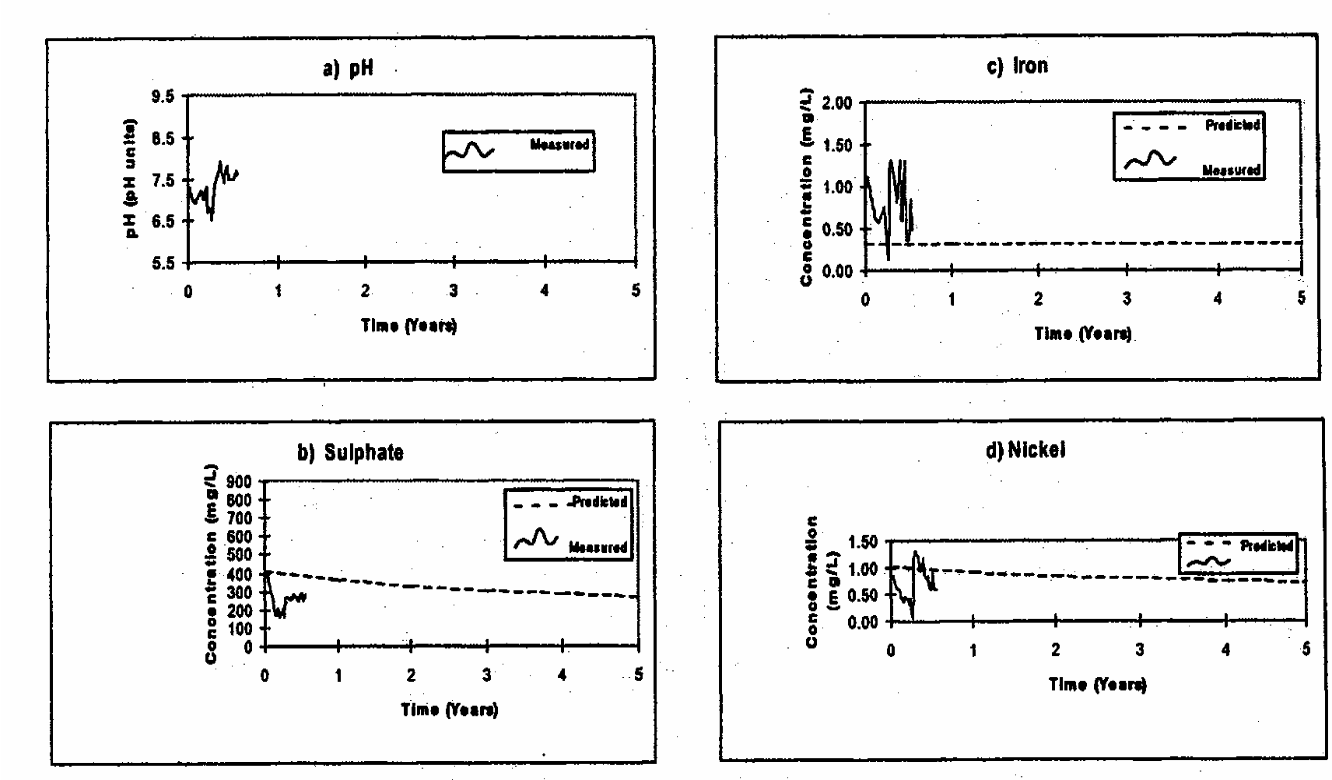
The chemical composition of the Lower Terrace water cover results from the mixing of inputs from: groundwater from a spring at the north end of the Lower Terrace, precipitation, natural runoff, surface water discharged over the Dam 1B spillway, and groundwater seepage from the Upper Terrace. It is estimated that it will take about 4 to 5 years for surface water originating in the Upper Terrace to report to the Lower Terrace as seepage, based on the estimated seepage rate of 28,000 m^3/a (Hall 1999).

DeVos *et al.* (2000) report that the predicted post-dredging water quality for the Upper Terrace pond was used to represent the initial pond water quality in the diffusion modelling. Metal loadings to the pond from precipitation and natural runoff were added to the loadings from the diffusion modelling and used to predict sulphate, iron, copper, nickel and zinc concentrations in the pond water. Predicted and measured values for pH, sulphate, iron and nickel concentrations in the Upper Terrace pond water are shown in Figure 4.2-28.

The Upper Terrace was flooded to the point of flow through the spillway in the spring of 1996. Figure 4.2-29 shows the nickel concentration trend in the Upper Terrace (as measured at the spillway) and in the Upper Terrace (as measured at the decant tower). Figure 4.2-30 shows the pH trend in the ponds.

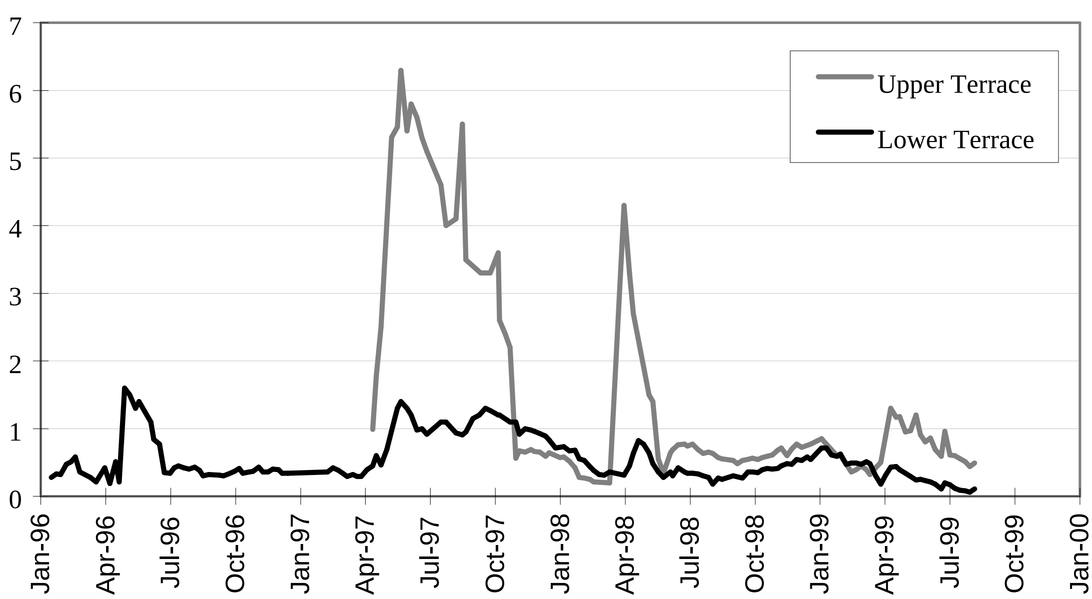
Metal concentration peaks have occurred each spring. The magnitude of the peaks, however, has declined from 6 mg/L Ni in 1996, to slightly more than 1 mg/L in 1999. Since February 1999, the nickel concentration in the Lower Terrace has been below provincial effluent quality limits. Predicted and measured nickel concentrations at the Dam 6 decant from the Lower Terrace pond (Figure 4.2-31) shows significantly lower actual nickel concentrations suggesting there may be attenuation (e.g. through co-precipitation with iron (DeVos *et al.* 2000)).

Figure 4.2-28 **Measured versus Predicted Water Quality, Upper Terrace**



Source: De Vos *et al.* 2000

Figure 4.2-29 **Nickel Concentrations**

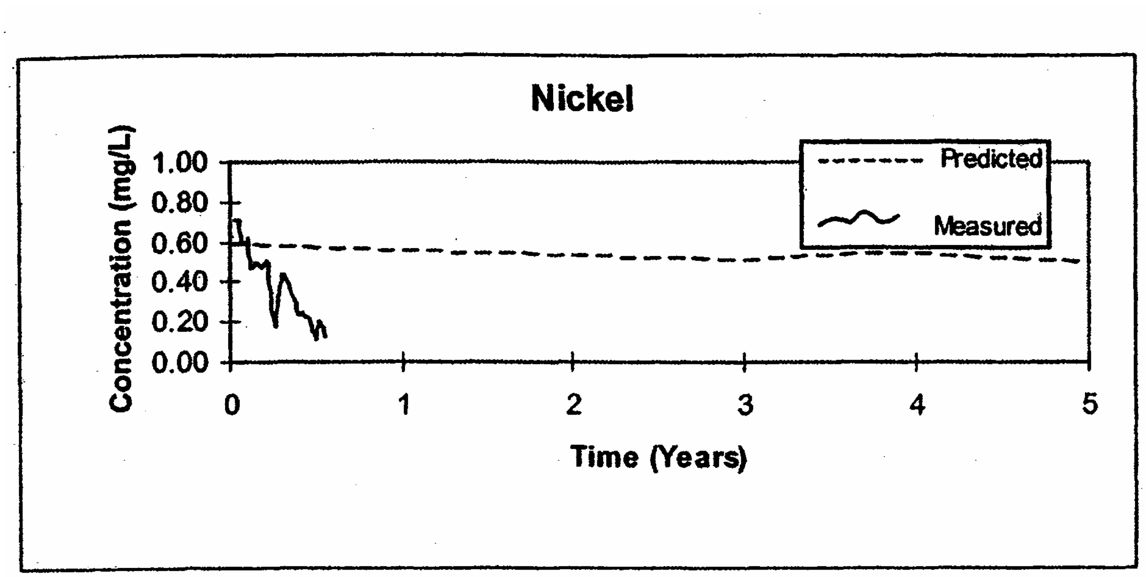


Note: Nickel concentration in mg/L
Source: Hall 1999

Figure 4.2-30 **pH**

Source: Hall 1999

Figure 4.2-31 **Measured Versus Predicted Nickel Concentrations at the Lower-Terrace Discharge**



Source: De Vos *et al.* 2000

It is estimated that a total of 75,000 tonnes of tailings were relocated by excavator and truck and 328,500 tonnes were moved by dredging. The total cost of these activities, including mobilization and demobilization of the contractor's equipment was $1.04 million Canadian. The costs of dam construction, and preliminary engineering, research and design are estimated to be $1.39 M and $0.18 M respectively. The estimated total project cost is $2.61 M or $29,000/ ha.

In 1998, Falconbridge engaged the Noranda Technology Centre (NTC) to determine a means to accelerate the establishment of aquatic vegetation in the water covers for two main reasons (Hall 1999). Firstly, vegetation assists in stabilizing tailings and preventing resuspension. Secondly, an objective in closing the area was to establish a viable ecosystem and return it to a productive use. In a preliminary trial, four species of submergent vegetation were identified and transplanted into the Lower Terrace. Follow-up inspections of the test plots have indicated that the plants are healthy and viable. Plans are being made to sponsor a second trial in the Upper Terrace, where turbidity may pose a challenge to plant growth. Hall (1999) indicates that on a qualitative basis, the fact that aquatic vegetation has begun to thrive and the presence of schools of minnows lead to the conclusion that the flooding of the NTA was the appropriate closure option.

DeVos *et al*. (2000) concluded that although a significant portion of the tailings in the New Tailings Area had been exposed for up to nine years, oxidation had not proceeded to the extent that precluded flooding. It is too early to evaluate the effectiveness of the numerical water quality predictions. The establishment of the organic substrate is expected to further improve the water quality by preventing the resuspension of tailings solids, and limiting the transport of oxygen to the sulphide solids.

MEND 2.15.1a Flooding of Pre-Oxidized Tailings, Mattabi Case Study

MEND 2.15.1a commenced in 1992 to directly address uncertainties in the design of water covers for use in decommissioning oxidized tailings. Laboratory column experiments and a field cell test were completed. Some of the laboratory experiments involved the use of sand or peat layers over the flooded tailings/water cover interface. The tailings used in the experiments were not alkali-amended. The research involved oxidized and unoxidized tailings from the Mattabi mine tailings impoundment located 73 km northwest of Ignace, Ontario.

MEND 2.15.1a reports that the Mattabi tailings impoundment was active from 1972 to 1991 over which time some 15.5 million tonnes of tailings were disposed over a 125 ha area (Figure 4.2-32). The tailings were deposited to a maximum depth of about 10 m in an oblong bowl-shaped depression that had previously held a shallow lake. Additional tailings storage was obtained constructing waste rock "starter dams" and then raising them using cycloned tailings to form structures to physically support the tailings mass. The structures were permeable and allowed the tailings to drain and resulted in a zone of unsaturated tailings. Selected data for samples of oxidized and unoxidized tailings samples collected in 1992 and 1993 from trenches in the north portion of the tailings impoundment are presented in Table 4.2-12.

Table 4.2-12
Selected Data for Mattabi Tailings Solids

Item	Unoxidized Tailings 1992	Oxidized Tailings 1992	Unoxidized Tailings 1993	Oxidized Tailings 1993
Pyrite Content	33.04[(1)]	36.20[(1)]	51.68[(1)]	34.64[(1)]
Chalcopyrite Content	0.14	0.26	0.32	0.26
Sphalerite Content	0.81	0.90	0.30	0.90
Galena Content	0.15	0.20	0.24	0.23
Arsenopyrite Content	0.42	0.39	0.21	0.60
Paste pH	2.95	4.29 $\pm$ 0.19	5.41	4.44 $\pm$ 0.92
Total S (%)	20.3	19.6 $\pm$ 1.5	18.4	26.4 $\pm$ 1.7
AP (kg $CaCO_3$/t)	634	613 $\pm$ 47	575	825 $\pm$ 53
NP (kg $CaCO_3$/t	-0.2	-0.15 $\pm$ 0.21	6	-0.45 $\pm$ 0.07
NP/AP	0	0	0.01	0

Note: [(1)] Mineral content expressed as mass%.
Source: MEND 2.15.1a

Column tests were completed using eight, plexiglas columns (Figure 4.2-33). The loading of the columns is summarized in Table 4.2-13.

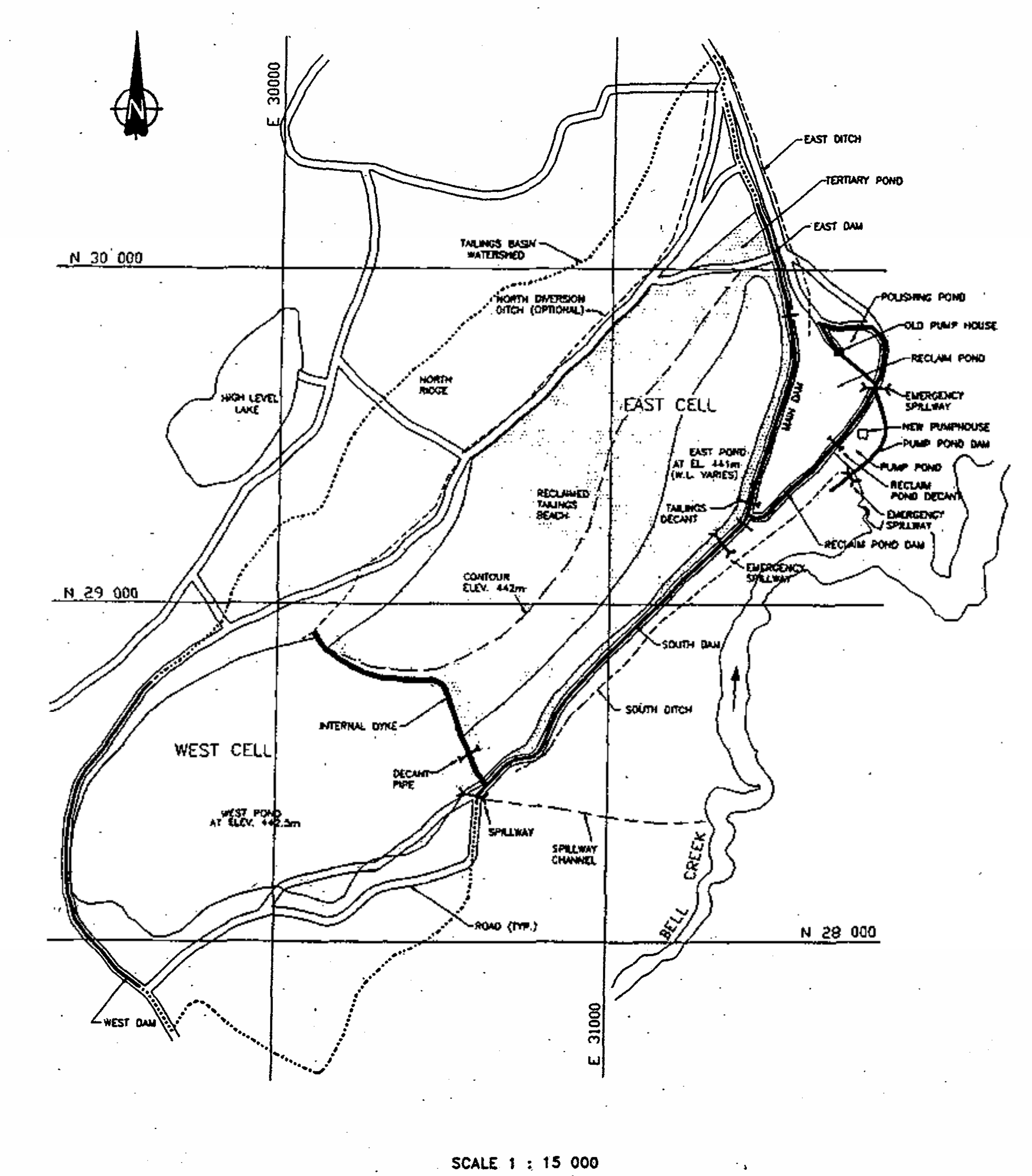
Table 4.2-13
Test Column Configurations

Column No.	Water Cover Depth	Attenuation Layer
MTWC-1 and MTWC-2	1 m	None
MTWC-3 and MTWC-4	1 m	10 cm sand layer
MTWC-5 and MTWC-6	1 m	10 cm peat layer
MTWC-7 and MTWC-8	No cover, saturated tailings	None

Source: MEND 2.15.1a

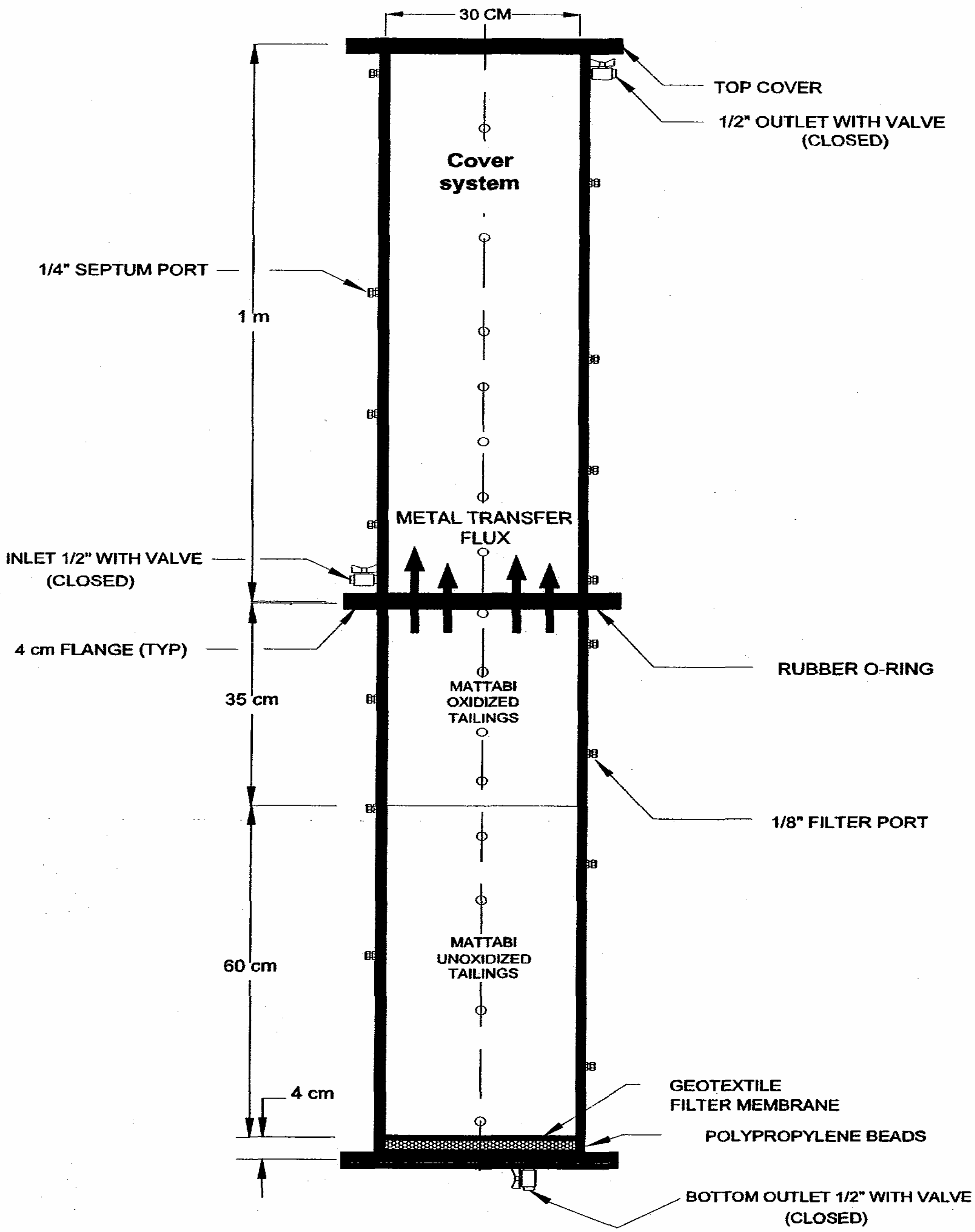
A field test cell was constructed over oxidized tailings in the Mattabi tailings impoundment area (Figure 4.2-34). The oxidized tailings surface within the cell was then flooded using lake water. The test cell parameters were subsequently monitored.

Figure 4.2-32 **Current Plan for Tailings Basin Closure**



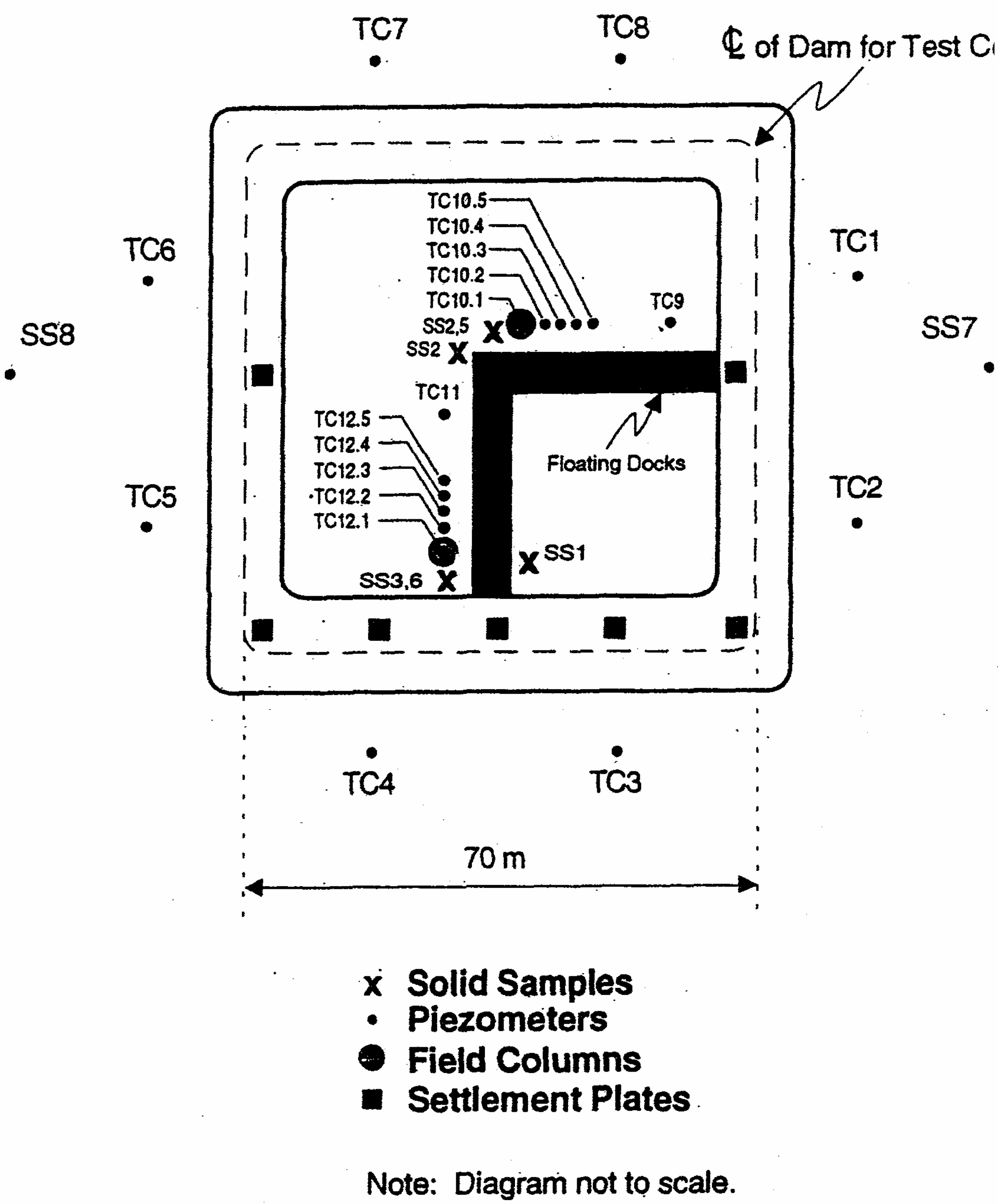
Source: MEND 2.15.1a

Figure 4.2-33 **Schematic Diagram of Laboratory Columns**



Source: MEND 2.15.1a

Figure 4.2-34 **Plan of Test Cell, and Instrumentation and Sampling Locations**



Source: MEND 2.15.1a

The results of the study, as reiterated from MEND 2.15.1a, are listed below:

- The laboratory column and field cell tests showed that directly flooding oxidized Mattabi tailings without installing an attenuation layer causes the release of metals and sulphate from pore water solution and soluble mineral phases to the water cover. In the columns, average total iron, sulphate and zinc concentrations in the water cover reached 257 mg/L, 927 mg/L and 20.2 mg/L, respectively, approximately one year after flooding. Corresponding concentrations in the field cell water cover one year after the flooding of July 1992 were 466 mg/L, 2850 mg/L and 3.8 mg/L, respectively. Differences between concentrations measured in the laboratory and in the field tests were likely due to differences in initial pore water compositions and soluble mineral contents between the columns and the test cell;

- Because the pore water of Mattabi tailings was rich in ferrous iron, the establishment of a water cover was accompanied by the precipitation of a thin layer of hydrous ferric oxide precipitate on the surface of the tailings. This was due to the oxidation of ferrous iron by dissolved oxygen and subsequently hydrolysis of ferric iron. The precipitate contributed to the removal of zinc and other metals from the cover by sorption reactions both in the columns and in the field test cell;

- Dilution of the water cover by addition of deionized water (laboratory columns) or rain and snowmelt (field test cell), flushing of solutes by water infiltration from the cover to the tailings, and removal of some metals by precipitation and sorption on the hydrous ferric oxide precipitate progressively reduced solute concentrations in the water cover. In July 1994, i.e. two years after the first flooding of the field test cell, metal concentrations in the test cell water cover met regulatory discharge limits: As < 0.5 mg/L, Cu < 0.3 mg/L, Fe < 3 mg/L, Pb < 0.2 mg/L and Zn < 0.5 g/L. However, the water cover pH remained lower than the minimum of 6.0. It is estimated that about 1855 mm of water infiltrated during this period. In the laboratory column water covers, the zinc concentration decreased much more slowly than in the field test cell and was still ~ 6 mg/L after 620 days of simulated infiltration, which translates into ~ 955 equivalent field days given average field infiltration rates; and

- The persistence of elevated zinc concentrations in the laboratory column water covers is likely explained by geochemical equilibrium of the overlying water with a Zn-containing solid phase in the hydrous ferric oxide layer at the tailings/water interface. Hence, the hydrous ferric oxide precipitate, although contributing to the decrease of zinc concentrations in the column water cover during the first year of testing, later acted as a source of soluble zinc as metal concentrations in the water cover and pore water decreased. In the field cell, the zinc content in the precipitate layer (0.22%) was less than in the laboratory columns (1.28%), presumably because of lower initial pore water concentrations. As a result, zinc did not leach as much, and discharge limits were met in the water cover two years after first flooding the cell, as pointed out above. Hence, the geochemical characteristics of the oxidized tailings (pore water composition, soluble minerals) have a large influence on the time required to achieve discharge limits in the water cover.

The effects of the attenuation layers made of sand or peat were only tested in the laboratory columns. Fluxes of metals from the tailings to the water cover were greatly reduced by placing an attenuation layer at the tailings/water interface. This was primarily because diffusion of metals from the oxidized tailings through the attenuation layer and to the water cover is a very slow process. Moreover, the attenuation layer also imposes a diffusion control on the availability of dissolved oxygen to the tailings, and therefore limits further tailings oxidation more effectively than a simple water cover (MEND 2.15.1a).

When a sand layer was used, the water cover met discharge limits during the entire duration of the tests. The average total iron, sulphate, and zinc concentrations in the water cover were below 0.3 mg/L, 50 mg/L, and 0.03 mg/L approximately half a year after flooding. The peat layer was less effective because of its own leachable zinc content: after 163 days of test, the zinc concentration was still 1.5 g/L in one of the columns. Hence, it is important to ensure that material used to build the attenuation layer has low contaminant levels so that it does not contribute to the contamination of the water cover. A careful and detailed characterization of this material is therefore very important. In general, peat obtained around the vicinity of mining sites does not appear to be a good candidate for building attenuation layers as it is often a source of iron and other metals in water cover applications (MEND 2.13.1).

Further, MEND 2.15.1a reports that seepage water quality was only measured in the laboratory column tests. Seepage concentrations were initially very high (similar to pore water concentrations) and remained stable for all columns until the displacement front reached the bottom of the columns (at about 0.7 pore volume), after which they decreased over time as infiltrating water flushed the contaminated pore water. The presence of a water cover accelerated the decrease in seepage concentrations when compared with tailings left exposed to the atmosphere, probably as a result of reducing further tailings oxidation.

The time or pore volumes of infiltration required to meet discharge limits in the seepage depends on the distribution of the contaminants in the tailings. When contaminants are mostly contained in the pore water, discharge limits may be achievable after flushing the tailings with ~ 3 pore volumes. However, when a significant portion of the contaminants is contained in soluble solid phases, concentrations in the seepage stabilized at elevated values and long-term seepage treatment will be required. The presence of an attenuation layer at the water/tailings interface seemed to have relatively little influence on the seepage water quality.

MEND 2.15.1a also stated:

- Long-term maintenance of water covers generally requires that seepage losses from the facility be minimal. In this case, the contaminated pore water will remain in place for many years, and seepage treatment, if part of the closure plan, can be expected to be long lasting; and

- If the seepage rate is high, volumes of seepage will need to be intercepted and treated. Economic considerations suggest that at sites with high seepage rates, a water cover may be suitable only if it can be supplemented with gravity-fed fresh water from a nearby source to the cover, and thus compensate for seepage losses. In this situation, treatment costs for the seepage will increase after flooding due to higher infiltration rates. Although the quality of seepage will increase over time, treatment may still be required in the very long term since solute concentrations may reach near-asymptotic values that are still above discharge limits, as was the case in some of NTC's tests. Hence, the establishment of a water cover over oxidized tailings is not recommended at sites with significant seepage losses.

Geochemical Assessment of Flooding - Stekenjokk Mine, Sweden

In 1997, Ljungberg *et al.* completed a geochemical field study to assess the efficiency of flooding sulphide tailings at the Stekenjokk Zn-Cu Mine in Northern Sweden. The decommissioning of the tailings in 1991 involved raising the water level in the tailings pond and the flooding of both oxidized and unoxidized tailings.

The Stekenjokk tailings pond covers an area of 1.1 km^2. Water depth ranges from 0.6 m to greater than 7 m, with an average depth of ~ 2 m. The total pond volume is 2 Mm^3. The sampling program used by the researchers is constructive and their technical paper would serve as a useful reference to readers interested in sampling a tailings pond. In particular:

- A portable Masterflux® peristaltic pump and silicone tubing was used to collect water samples from selected depths. Samples were field-filtered using a 0.45 µm pore size filter. The water column was sampled seasonally at numerous locations to avoid bias due to dimictic conditions; and

- A modified gravity corer was used to collect tailings samples which were then conserved using argon gas. Pore water samples were extracted in an argon-filled glove box within eight hours after sampling.

Their key conclusions with respect to the effect of tailings flooding, based on data obtained, are as follows:

- The most important source of Zn in the water column is probably diffusion from pore water in those areas of the tailings that were oxidized before flooding. The tailings pond is oxic year round and is well mixed; and
- The pore water in the tailings that were oxidized prior to flooding contain higher metal concentrations than the pore water in the unoxidized tailings (Table 4.2-14). The dissolution of gypsum in the oxidized tailings is the most probable explanation for the high levels of Ca and S in the oxidized tailings pore water.

Table 4.2-14
Comparison of Selected Water Cover and Tailings Pore Water Quality Data

Parameter	Water Cover	Pore Water in Oxidized Flooded Tailings (Station P3)		Pore Water in Unoxidized Flooded Tailings (Station P1)	
		0-4 cm Depth	5-10 cm Depth	0-6 cm Depth	7-12 cm Depth
Zn	139	1,470	3,640	257	152
Pb	0.17	16.5	29.4	31.8	29.3
Ni	1.3	16.0	34.6	11.5	10.0
Cu	1.8	10.1	11.6	5.4	7.1
Cd	0.69	4.34	6.11	2.08	0.56
Fe	0.03	0.06	1.31	0.04	0.73
Ca	20	406	622	142	287
S	11	289	472	115	254

Note: All units in mg/L; data obtained after filtering
Source: Adapted from Ljungberg *et al.* 1997

Sulphide oxidation and carbonate buffering probably occurs, at a low rate, in the upper, exposed layer of unoxidized tailings. The researchers indicate that, based on Humidity Cell data, with the high pH conditions prevalent at the tailings area, sulphide reactivity may decrease with time as the sulphide surfaces become hydroxide coated.

Ljungberg *et al.* (1997) additionally assessed the possible effect of the oxidation of resuspended sulphides on the quality of the water cover. It was determined that resuspension is not an important metal source of this pond as the pond includes a breakwater system to prevent tailings resuspension.

Geochemical Assessment of Cyclical Submergence/Exposure - Lago Junin, Peru

Pedersen *et al.* (1999) report that recent research carried out on submerged sulphide-rich tailings deposited permanently under water shows that the rates of acid generation and metals release are either negligible or so slow as to have no measurable effect on water quality. This conclusion is fully supported by repeated sampling and analyses programs in Anderson Lake, Manitoba and Buttle Lake, British Columbia - both lakes contain submerged tailings that sequester metals as opposed to releasing metals to the lake bottom water.

Pedersen *et al.* (1999) assessed the contrasting geochemical behaviours of permanently submerged, and alternatively submerged-exposed sediments in Lago Junin, located in the Peruvian altiplano. Their findings emphasize the importance of permanently maintaining water covers over sulphide tailings and waste rock. The methodology used at Lago Junin was similar to that used at Anderson Lake and Buttle Lake. At Lago Junin, high-resolution (peeper) sampling of lake bottom water and tailings pore water was undertaken at the following two stations:

1. Station J2 located in the centre of the basin, at 8.5 m depth where the sediments contain ~20 wt% organic carbon and ~2.5 wt% sulphur and are permanently submerged; and
2. Station U2 located in a shallow basin where the sediments become exposed and unsaturated due to an annual lake drop. The sediments at this location are predominantly mining-derived deltaic deposits containing ~4 wt% organic carbon and 2 wt% sulphur.

Pedersen *et al.* (1999) report:

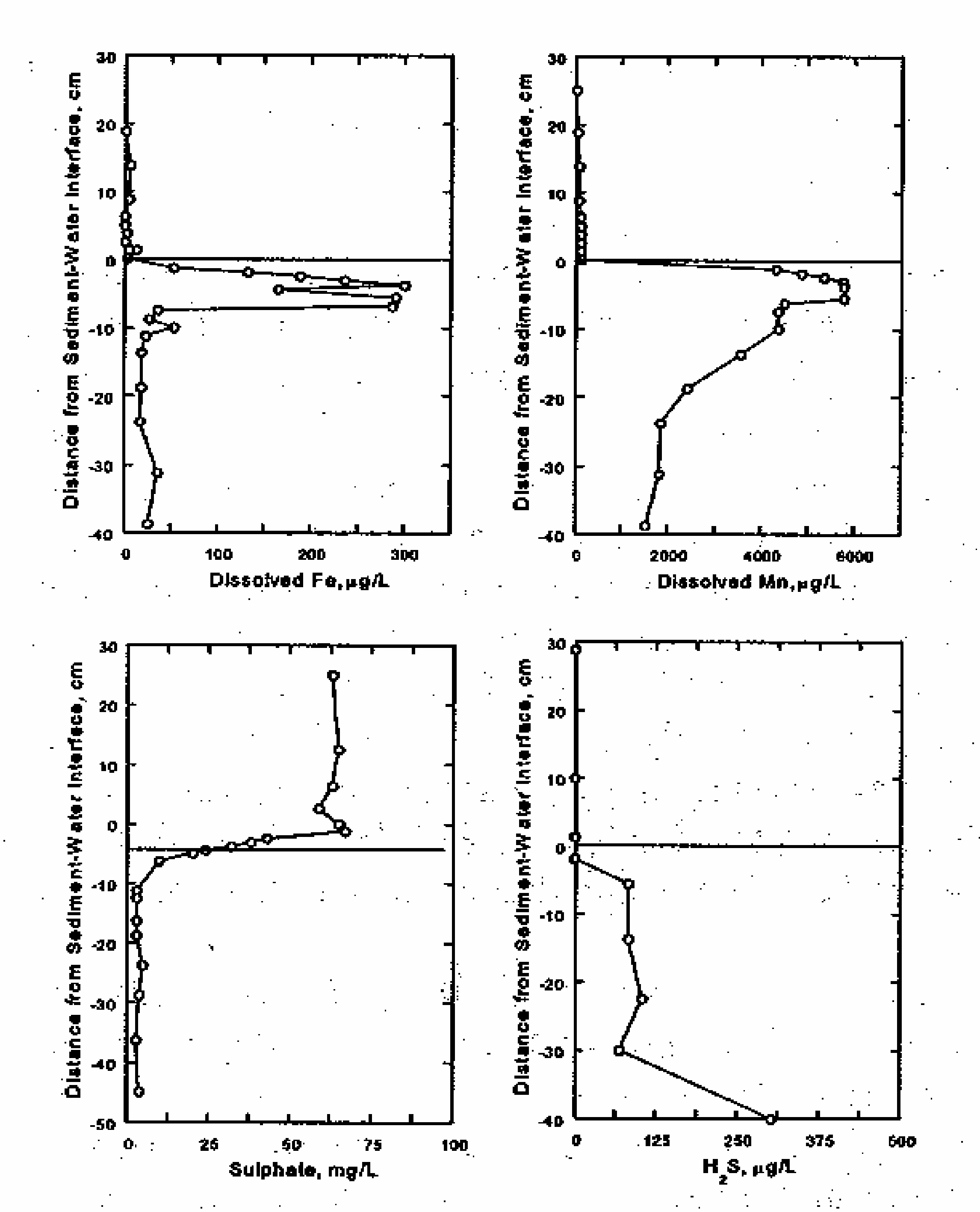
- At Station J2 (permanently submerged sediments), profiles of dissolved Fe, Mn, SO_4^{2-} and H_2S indicate the sediments become anoxic at very shallow depths. The reductive dissolution of manganese- and iron-bearing solid phases liberates their dissolved components to the pore water in the upper few centimetres of the sediment column as shown in Figure 4.2-35. Sub-oxic conditions are indicated by the abundance of ferrous iron in the upper horizons. Sulphate reduction produces a precipitous decline in SO_4^{2-} concentrations from in excess of 60 mg/L at the sediment-water interface to less than 5 mg/L at a depth of 10 cm. H_2S, an end product of sulphate reduction, is present in pore water in the top two centimetres. Collectively, the data imply that fully anoxic conditions exist in close proximity to the sediment/water interface.
- Trace metal profiles (Figure 4.2-36) demonstrate that chemistry changes occur across the interfacial sediment layers. The maxima of dissolved Zn and Ni at the interface indicate these elements are diffusing to deeper sediment depths and into the overlying water column. The remobilization depths of Zn and Ni occur 2 to 4 cm above the predicted horizon of Mn (IV) and Fe (III) reduction suggesting that these pore water enrichments are not the result of reductive processes. Rather, the interfacial maxima likely reflect the oxidative dissolution of labile organic-particulates at the sediment-water interface.

At station J2, profiles of trace metals respond to changes in the redox environment with depth. Steep decreases in the concentrations of Zn and Cu reflect the removal of these metals from pore solution (Figure 4.2-36). Copper concentrations decrease from ~9 µg/L in the water column to below detection limits (0.1 µg/L) at approximately 2 cm below the sediment-water interface. Similarly, Zn values decrease from ~275 µg/L at the interface to ~2 µg/L below 2 to 3 cm depth. The consumption zones, which occur in the upper few centimetres of the sediment column, are consistent with sulphide production horizons, and suggest that Zn and Cu are precipitated as authigenic sulphides. Sulphide precipitation has previously been proposed to limit the concentrations of these elements in the pore waters of both pristine lakes and mining-impacted systems. Minor removal can also be observed for Ni and Pb; however, the magnitudes of diffusive influxes for these elements are considerably lower. Collectively, the data imply that the sediments in the main basin of Lago Junin are serving as a permanent sink for Zn and Cu, and to a minor extent, Pb and Ni. Diffusive influx calculations for Zn, Cu, Ni and Pb indicate that the precipitation of authigenic phases contributes insignificantly to the accumulation rates of these elements.

The diagenetic state of the periodically unsaturated deposits in Upamayo (Station U2) differs greatly from that observed in the main basin. The reductive dissolution of Mn and Fe oxides results in increases in the concentrations of dissolved Mn(II) and Fe(II) in the shallow pore waters (Figure 4.2-37). The abundance of dissolved iron in the pore waters indicates suboxic conditions. Pore water sulphate concentrations also increase in the top 5 cm in the sediment column. The pore water maximum of over 1,500 mg/l represents a two order of magnitude increase over bottom water values. The magnitude of sulphate remobilization does not permit the detection of sulphate removal via reactions, and as a result, such profiles offer little with respect to the diagenetic state of the deposits. However, the hydrogen sulphide data demonstrate the presence of fully anoxic conditions at a depth of 30 cm (Figure 4.2-37).

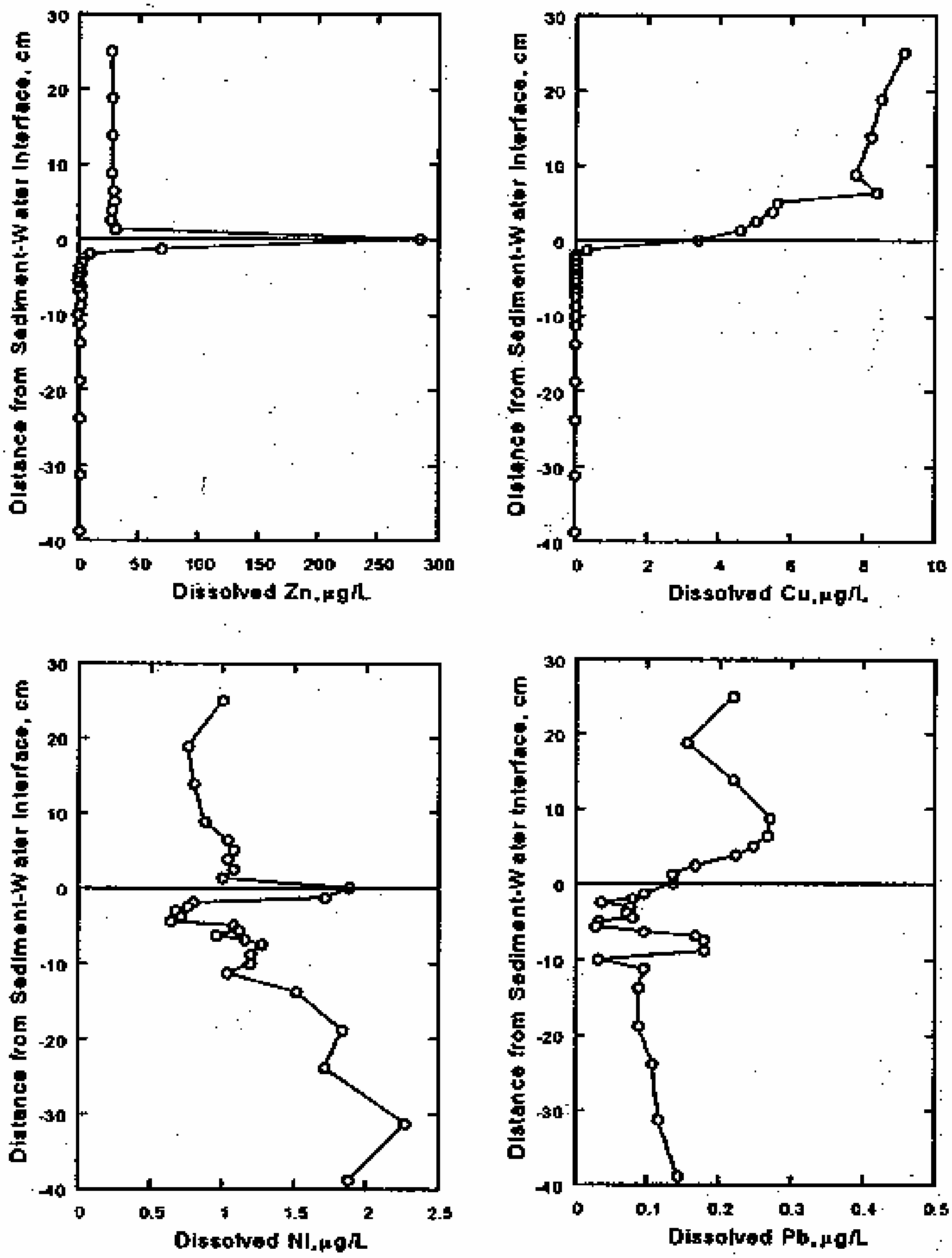
Pedersen *et al.* (1999) also report that the pore waters at Station U2 are characterized by a pronounced decrease in pH which has a significant effect on metal remobilization. Values drop across the sediment-water interface to a minimum of ~pH 3.2 at a depth of 40 cm (Figure 4.2-38). The decrease in pH in the pore waters likely reflects the result of acid-generating reactions accelerated by periodic wetting and drying associated with lake level fluctuations. From the period 1988 to 1997, annual changes in lake level averaged over 1.7 m and ranged from 0.85 to 2.5 m. During the wet season survey, the lake level was near its seasonal maximum. Since the water depth at Station U2 was 80 cm, it is likely that the sediments at U2 would be exposed to the atmosphere for a considerable period of time each year. In an average year, unsaturated conditions would likely extend to a sediment depth of approximately 1 m. This annual periodicity of unsaturated and saturated conditions likely fosters conditions which encourage sulphide oxidation in the sulphur-rich (2-4 wt% S) deposits at this site. In contrast, pore water pH at Station J2 remains invariant with depth (Figure 4.2-38).

Figure 4.2-35 **Dissolved Iron, Manganese, Sulphate amd Hydrogen Sulphide Distributions in Peeper Pore Waters at Station J2, Lago Junin, Peru**



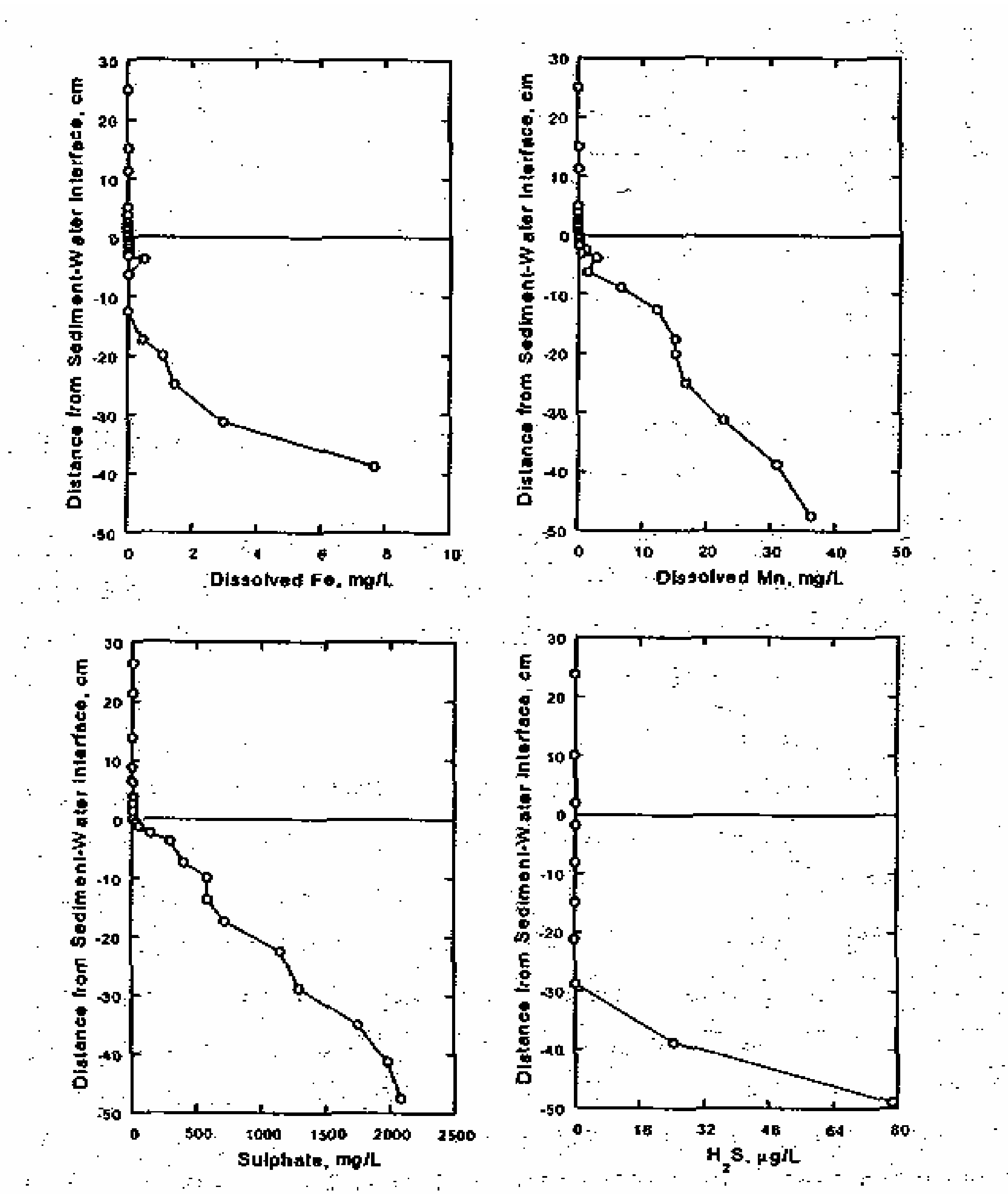
Source: Pedersen *et al.* 1999

Figure 4.2-36 **Dissolved Zinc, Copper, Nickel and Lead Distributions in Peeper Pore Waters at Station J2, Lago Junin, Peru**



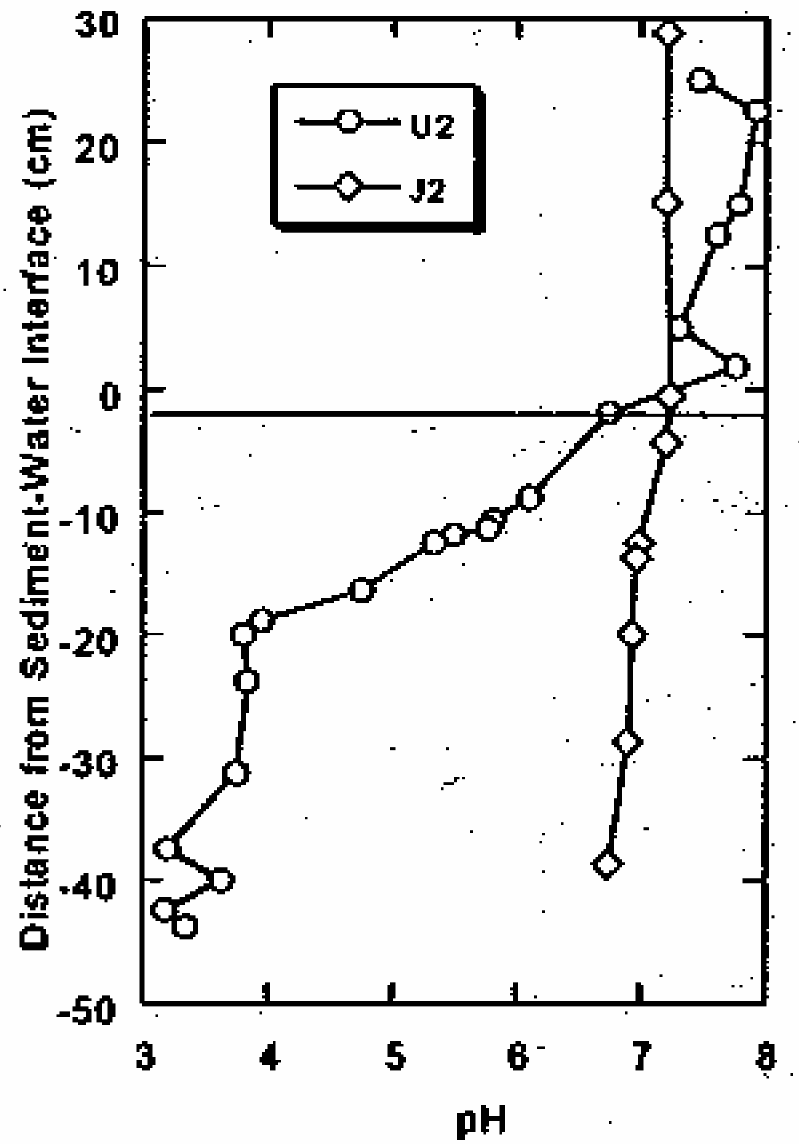
Source: Pedersen *et al.* 1999

Figure 4.2-37 **Dissolved Iron, Manganese, Sulphate amd Hydrogen Sulphide Distributions in Peeper Pore Waters at Station U2, Lago Junin, Peru**



Source: Pedersen *et al.*1999

Figure 4.2-38 **Comparison of pH across the Sediment-Water Interface at Stations J2 and U2, Lago Junin, Peru**



Source: Pedersen *et al.* 1999

The water level fluctuations at Station U2 (representative of the shallow Upamayo region of Lago Junin) and the resulting periodic exposure of the sulphur-rich deposits, will result in continued acid-generation and metal remobilization. Although fluxes to the overlying water column were minor at the time of sampling (late wet season), release rates may be considerably higher at the onset of flooding in the early wet season. A pulse of dissolved metals to the water column, for example, may result from the dissolution of oxidation products generated over the unsaturated period. Studies which have examined the reactivity of consistently submerged unoxidized, sulphide-rich tailings deposits have demonstrated that rates of sulphide oxidation are highly reduced in subaqueous environments. Specifically, rates of acid generation and metal remobilization resulting from sulphide oxidation have been shown to be so minor as to impact negligibly the chemistry of overlying waters. From a geochemical perspective, therefore, permanent flooding of the deposits in Upamayo presents a suitable remediation strategy to minimize sediment reactivity.

The contrast between the results from Anderson and Buttle Lakes and Lago Junin (Pedersen *et al.* 1999) emphasize a critical factor regarding the subaqueous disposal of sulphide-bearing waste rock and tailings: that such materials must be discharged under water immediately after milling, and they must remain submerged in perpetuity if chemically benign behaviour is to prevail.

North Coldstream Tailings - Sulphide Tailings Relocation

The North Coldstream Copper Mine, located near the hamlet of Kashabowie in northwestern Ontario ceased operations in 1967. In the last five years of the mine life, an estimated one-half million tonnes of sulphide tailings were deposited into a licensed tailings management area (TMA-2). Burns *et al.* (1999) describe a two-year field investigation and technical program to develop the closure plan for the exposed tailings beach. The closure plan involved the relocation of about 80,000 m^3 of oxidized tailings, by dredging, to a metre below the surface of the TMA-2 tailings pond. The exposed shoreline was then restored and revegetated.

MEND 2.36.2 Hydrogeochemistry of Oxidized Waste from the Stratmat Site, New Brunswick

The Stratmat Mine extracted lead-zinc-copper-silver from massive sulphide orebodies located 50 km northwest of Miramachi, New Brunswick. Open pit and underground mining was started in 1989, and completed in 1991. A 1.84 million t waste rock pile was constructed. The rock contained variable sulphur concentrations (<0.01 to 23%) averaging about 3%. Neutralization potentials were low (generally less than 10 kg $CaCO_3$/t) with some exceptions. The closure plan included the provision for blending of limestone with the rock during construction and return to the mined-out pits at closure. In 1991, acidic seepage was observed indicating that the limestone provided very little delay in onset of acid discharge.

This project evaluated the effect of placement of oxidized rock in water using laboratory leach columns. Three stages were analyzed:

- Initial stage – Inundation of rock with water;
- Second stage – Circulation of water through the waste rock; and
- Third stage – Static water with 60 cm overlying water.

Details of the analytical procedure are shown in Figure 4.2-39. A total of 72 pore volumes were extracted by the tests.

During the initial stage, significant flushing of soluble minerals occurred as pH fell to about 4.0. Concentrations of sulphate, iron, and lead were 1500 mg/L, 5 mg/L and 1.5 mg/L, respectively. Concentrations of Ca+Mg+Mn were expressed as 360 mg eqCa/L. In the first rinsing cycle, concentrations of most parameters increased, except for Pb and Fe. Lead concentration was probably controlled by dissolution of anglesite, and therefore decreased as sulphate concentration increased. Iron concentration was probably limited by the solubility of ferric oxyhydroxides. As rinsing continued (4 pore volumes per cycle), concentrations typically decreased. Lead concentrations increased from 2 mg/L to greater than 6 mg/L. During the static stage, the overlying water was sampled. Concentrations initially decreased and then recovered to levels comparable to the circulation stage. At the same time pH increased to 4.5 and then fell to 3.5.

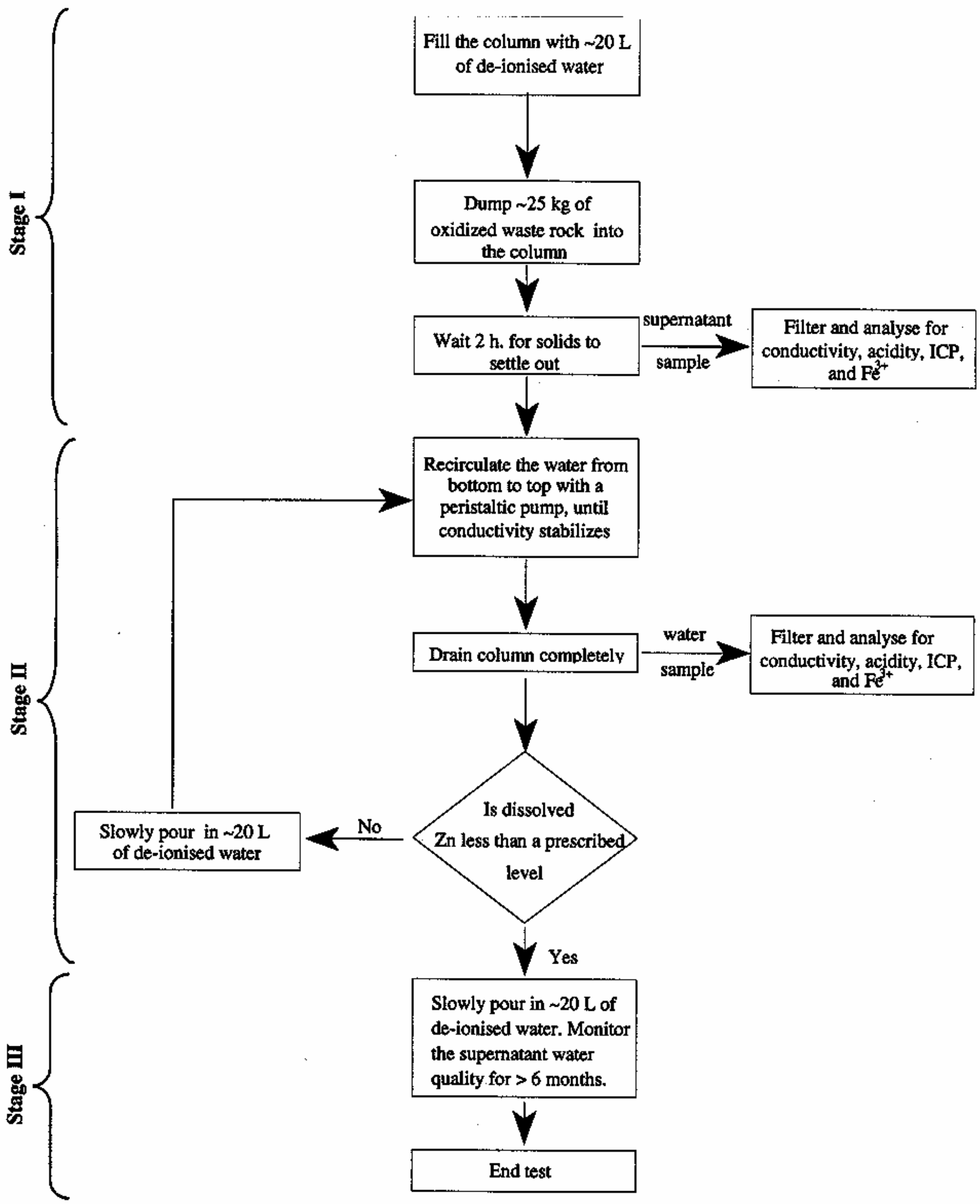
After 72 pore volumes, concentrations of Zn and Pb exceeded effluent criteria in the Metal Mine Liquid Effluent Regulation (MMLER) of the Fisheries Act.

Based on the results, the sequence of leaching was observed to be (in order of first element to last element): Al, Cu, Mg, Zn, Mn, Ca, Pb. This sequence indicates that Al is depleted first, then Cu and so on. The implication of the sequence is that concentrations of elements such as Pb would be difficult to reduce by repeated rinsing. The concentrations of Cu and Zn in rinse waters were not limited by mineral solubility at these pHs, and therefore could be flushed rapidly. Iron and lead concentrations were believed to be controlled by the solubility of ferric oxyhydroxides and anglesite, respectively, and were therefore flushed at relatively constant concentrations.

<u>Subaqueous Disposal of Oxidized Waste Rock - Eskay Creek Mine, British Columbia</u>

During exploration at the Eskay Creek Project from 1990 to 1991, 100,000 tonnes of potentially acid generating brecciated and sericitic waste rock were produced from an adit (Murphy *et al.* 1999). In less than a year, the pH of drainage from the waste rock pile began to decrease and reached pH 3. A small oligotrophic alpine tarn (Albino Lake) was identified as a suitable subaqueous disposal site. The tarn averaged 7 m in depth to a maximum of 20 m depth. The total volume of the tarn was 1,080,000 m^3.

Figure 4.2-39 **Column Dissolution Experiment Procedure**



Source: MEND 2.36.2a

A plan was developed to add hydrated lime to the waste rock to stabilize soluble components during re-handling and subaqueous disposal. A test program was developed to determine the appropriate quantity of lime to add during disposal (Higgs *et al*. 1997):

- Seven samples were collected from different locations to characterize the overall dump materials in terms of rock type, degree of oxidation and proportion of the waste rock represented by the sample;
- Paste pH and neutralization potential were determined on the minus 20 mesh fraction of the seven samples to derive a correlation between paste pH and lime demand; and
- A 300 kg sample of the dominant (81%) and reactive rock type (brecciated rhyolite) was collected from one location. The sample was subsequently used for bulk stabilization tests. The testwork was conducted on 6 to 7 kg whole splits mixed to a slurry with 66.9% and 57.5% solids (control and treated cases, respectively). Lime was added at a rate of 3.08 kg for treatment. The samples were leached twice to determine if the neutralization products were stable.

Results of the treatment are shown in Table 4.2-15. These results demonstrated effective neutralization and stability of the neutralization products.

Table 4.2-15
Oxidized Waste Rock Stabilization Test Results

Parameter	**Oxidized Waste Rock**		**Treated Waste Rock**	
Cycle	Cycle 1	Cycle 2	Cycle 1	Cycle 2
pH	2.89	2.97	7.81	8.11
Sulphate SO_4	4444	2604	1827	1647
Acidity $CaCO_3$	1458	888		
Alkalinity $CaCO_3$			95.5	88.5
Dissolved Metals				
Arsenic	0.41	0.20	< 0.02	< 0.02
Cadmium	0.243	0.135	< 0.002	< 0.002
Copper	5.90	3.47	0.018	0.015
Iron	374.1	207.9	0.43	0.37
Manganese	33.0	17.7	5.84	4.4
Lead	1.71	1.26	< 0.02	< 0.02
Antimony	0.21	0.12	< 0.02	< 0.02
Zinc	46.76	25.82	< 0.01	< 0.01

Note: All concentrations in mg/L

Disposal of the waste rock began in 1994 and was completed in 1995. A causeway was constructed into the lake for deposition of waste, which was spread using a loader. Actual lime addition rates were determined on a truck-by-truck basis using the correlation between paste pH and lime requirement. Monitoring of the lake water during disposal indicated acceptable water quality.

<u>Subaqueous Disposal of Oxidized Waste Rock - Skorovas Mine, Norway</u>

MEND 2.33.1 reported on remedial action at the Skorovas Mine designed to address high metals and acidity loadings entering natural lakes. The underground mine extracted massive sulphide ore from 1952 to 1984, initially producing pyrite concentrate (until 1976) for acid production, then copper and zinc concentrates. A 160,000 m^3 rock pile was developed adjacent to the mine entrance. Monitoring indicated that 82% of copper load and 84% of zinc load from the site originated from this rock pile. A further 10% of the load originated from minor rock piles. The balance of the load was from natural sources, such as pyritic outcrops.

The remediation plan involved re-location of the acid generating waste rock to Dausjøen ("Dead Sea"), a naturally acidic lake approximately two miles from the mine site. A test dump using 100 m^3 of rock in 2300 m^3 of water showed that the pH of the isolated lake water fell from 3.8 to 2.4, and copper concentrations increased from 0.58 mg/L to 3.8 mg/L. Initially, it was planned to add lime or limestone directly to the waste rock during re-location to neutralize the acidic products *in situ*.

Due to concerns about the stability of the bottom sediments (primarily iron-rich precipitates), a facility was constructed to contain the waste rock behind a toe berm constructed of 2 m^3 boulders. The lake was drawn down about 5 m to allow the rock to be placed in a dry condition. Instead of adding a neutralizing agent directly to the waste, milk of lime was discharged though a submerged pipe during construction activities. The rate of discharge was 0.12 m^3/s. The rock pile was re-located in three months, using a backhoe, and six to eight dump trucks. The rock surface was cleaned manually to ensure that all acid generating rock was removed.

The rock was placed in 2 to 3 m lifts, building from shore to the retention berm. The final surface was covered with 0.3 m of compacted glacial till. The till cover on the front face was not compacted, and it was not continuous over the whole face. Sampling points were added to the rock pile. Finally, a new dam wall was constructed at the outflow of the lake to raise the lake level 2 m above the original level.

In 1991, following construction, the desired reduction in load from the site was achieved. The pH of Dausjøen was about 4.0 (comparable to pre-remediation). Monitoring of pore water in the rock mass indicated pH of 2.4. MEND 2.33.1 predicted that this stored pore water would equilibrate with the lake water, possibly resulting in chemical degradation of the lake water, and that this effect could have been avoided by adding a neutralizing agent to the rock as originally planned.

Costs for remediation work are shown in Table 4.2-16 (MEND 2.33.1).

Table 4.2-16
Costs for Remediation Work – Skorovas Mine

Item	**Cost (1991 Canadian $)**
Planning, design, call for tenders, pollution monitoring	595,000
Safety, liming systems, water displacement, dam construction	425,000
Construction, re-location of mine rock	2,550,000
Cleaning up, removal of buildings, roads and railroads	850,000
Total	4,420,000

4.2.3. VARIATIONS

<u>Subaqueous Disposal of Tailings in the Strathcona Tailings Treatment System</u>

Stogran *et al.* (1997) and MEND Associate Project APC-1 undertook a subaqueous tailings pilot column test program to evaluate the effects of the subaqueous deposition of pyrrhotite and fine tailings from the Strathcona Mill on water cover quality, and to determine if subaqueous tailings deposition is a viable means of controlling acid generation. Selected aspects of the thirteen-month test program including their key conclusions are presented below.

Five Plexiglass columns measuring 1.83 m high and 0.3 m in diameter were filled with combinations of unoxidized tailings and a 1 m deep acidic water cover (Figure 4.2-40). The tailings combinations were:

- Strathcona Mill (SM) tailings consisting of a 3:5 volume ratio of tailings slimes and pyrrhotite tailings;
- Thickened SM tailings;
- Pyrrhotite tailings;
- Thickened pyrrhotite tailings; and
- Pyrrhotite tailings with a 0.1 m thick sediment substrate top layer.

The columns were monitored to assess physical and chemical changes in the water cover and the tailings pore water over time. During the initial placement of the tailings in the columns, the

Figure 4.2-40 **Schematic Representation of the Five Pilot Columns**

Height (mm)	**Column 1**	**Column 2**	**Column 3**	**Column 4**	**Column 5**
1800	Acidic • Water •	Acidic • Water •	Acidic • Water •	Acidic • Water •	Acidic • Water •
800			Substrate		
700 60	Pyrrhotite • Tailings	Thickened • SM Tailings	Pyrrhotite • Tailings	Thickened • Pyrrhotite Tailings	SM • Tailings
	Filter Bed	Filter Bed	Filter Bed	Filter Bed	Filter Bed

- denotes sampling ports

Source: Stogran *et al*. 1997

acidic water cover was displaced by the neutral water in the tailings slurry. To provide an acidic water cover at the onset of the test program, the water cover was replaced using fresh acidic water from the Strathcona Tailings Treatment System.

Sampling ports in the columns were used to collect samples of the water cover, water at the water cover/tailings interface, and of the tailings pore water on a bimonthly basis. The tailings were well-characterized through analyses that included: an ICP-ES multi-elemental scan; mineralogical examinations using X-Ray Diffraction and polished section investigations; flow through leach testing; sieve, cyclosizer and hygrometric analyses; and the determination of other properties (i.e. slurry viscosity, specific gravity, settling density, hydraulic conductivity, and tailings porosity).

The effect of the water column in each of the columns was assessed through monitoring of conductivity, Eh, pH, SO_4^{2-}, and metal contractions. A bacteriological analysis determined that the fungal growth occurred within four of the five columns - fungal hyphae was not present in the pyrrhotite tailings water cover. Identification of the bacteria and fungi indicated that there were no facultative or sulphate reducing bacteria in the water cover samples. However, significant populations of *Thiobacillus* were identified in water samples from the water column/ tailings interface in the pyrrhotite and thickened tailings columns, and in the pyrrhotite tailings water cover. Minor populations of *Thiobacillus* were identified in the SM tailings column interface and water cover. A stagnant 1 m deep water cover was found to be capable of maintaining the submerged tailings in a reducing condition.

Field Studies of Biologically Supported Tailings

St. Germain *et al.* (1997) assessed the feasibility of improving the effectiveness of water covers by including a biological oxygen-consuming sediment layer. The concept of maintaining biological activity with the sediment layer using a living plant cover was assessed by the researchers at the Brenda Mine tailings pond in British Columbia, and the Heath Steele tailings pond in New Brunswick.

The decomposition of organic matter by aerobic bacterial activity within a sediment layer serves as an oxygen-consuming layer. In addition, dissolved metals contained in the water column and in the tailings pore water that mitigate into the organic layer can be retained within the sediment through organic metal-complexation, and through sulphate-reducing bacterial activity leading to sulphide precipitation. St. Germain *et al.* (1997) considered a biologically supported water cover as a water cover that includes an active biological system where organic layer generation and biological activity are sustained. A biologically supported water cover may provide additional benefits by:

- Preventing tailings resuspension, and

- Removal of metals from the water column through adsorption and ion-exchange.

The investigations and selected findings of St. Germain *et al.* (1997) with respect to their research at the Brenda and Heath Steele tailings pond are summarized below.

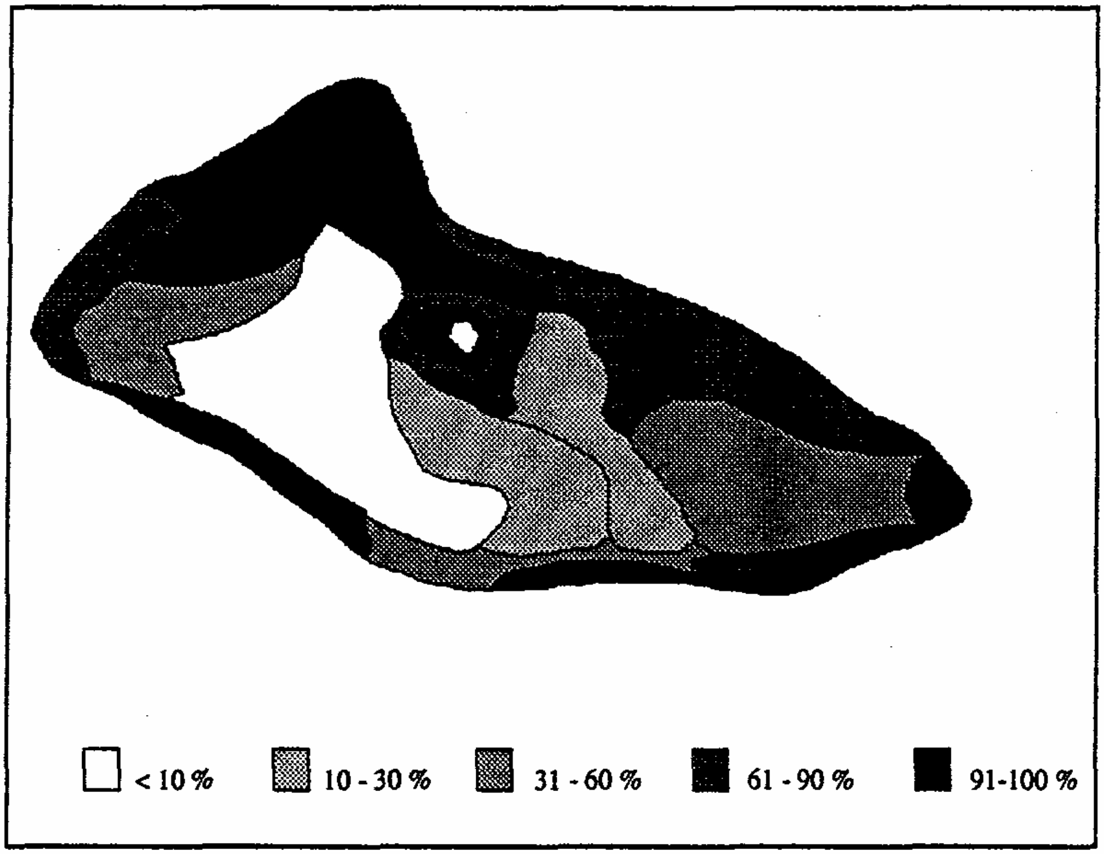
Brenda Mine Tailings Pond:

The Brenda Mine tailings pond (non-acid-generating tailings) located near Peachland in south central British Columbia, has a basic pH (8.4) and Mo and Cu concentrations of approximately 2.2 and 0.01 mg/L respectively. The sediment pore water was sampled *in situ* for five weeks using a peeper. Field measurements of pH, dissolved oxygen, oxidation-reduction potential and temperature were made for the extracted pore water samples. Analytical tests were subsequently completed for metal content (30 elements), total organic carbon, NO_2-N, NO_3-N, PO_4-P, and SO_4. Sediment samples were obtained at 20 cm depths using coring tubes, and samples separated when district layers were present. Aliquots of 20 to 30 g samples were dried at 90ºC to a constant dry weight and then analyzed for total organic carbon. The enumeration of sulphate reducing bacteria (SRB) was completed using diluted (IOX) sediment samples and an immunoassay test kit - Conoco Inc's Rapidchek II SRB detection system. The Rapidchek II test uses purified antibodies to detect the adenosine-5'- phosphosulfate (APS) reductase enzyme that is common to all SRB strains.

The biological production of the Brenda Mine tailings pond following the closure of the mine in 1990 was restricted by a nutrient in excess of 10^3 N:P. In 1993, this ratio had been modified to 8N to 1P by aerial application of granular fertilizer (11%N, 52%P and 0%K). This led to the development of an 8 cm thick filamentous green algae layer and a diverse plankton community. The researchers reported that mixed stands of rapid growing, native macrophytes were introduced to produce organic carbon, decrease sediment movement, bioaccumulate metals and provide habitat for microflora. The sandwiching of a 5 cm thick layer of plants between two layers of wire was found to be the best planting technique. Eight hundred and fifty sandwiches consisting of 75% *Elodea canadensis* (Canadian waterweed), 15% *Potamogeton crispus* (pondweed), 5% other species were distributed in shallow water in September 1992. Weedbeds varying from 60 to 100% cover were observed at depths of 1 to 4 m in 1995 (Figure 4.2-41). Since that time, the percentages of the different species have varied. Plant survey data for 1996 are presented in Table 4.2-17. In 1996, in excess of 90% of periphyton (plantleaves) consisted of phytoplankton.

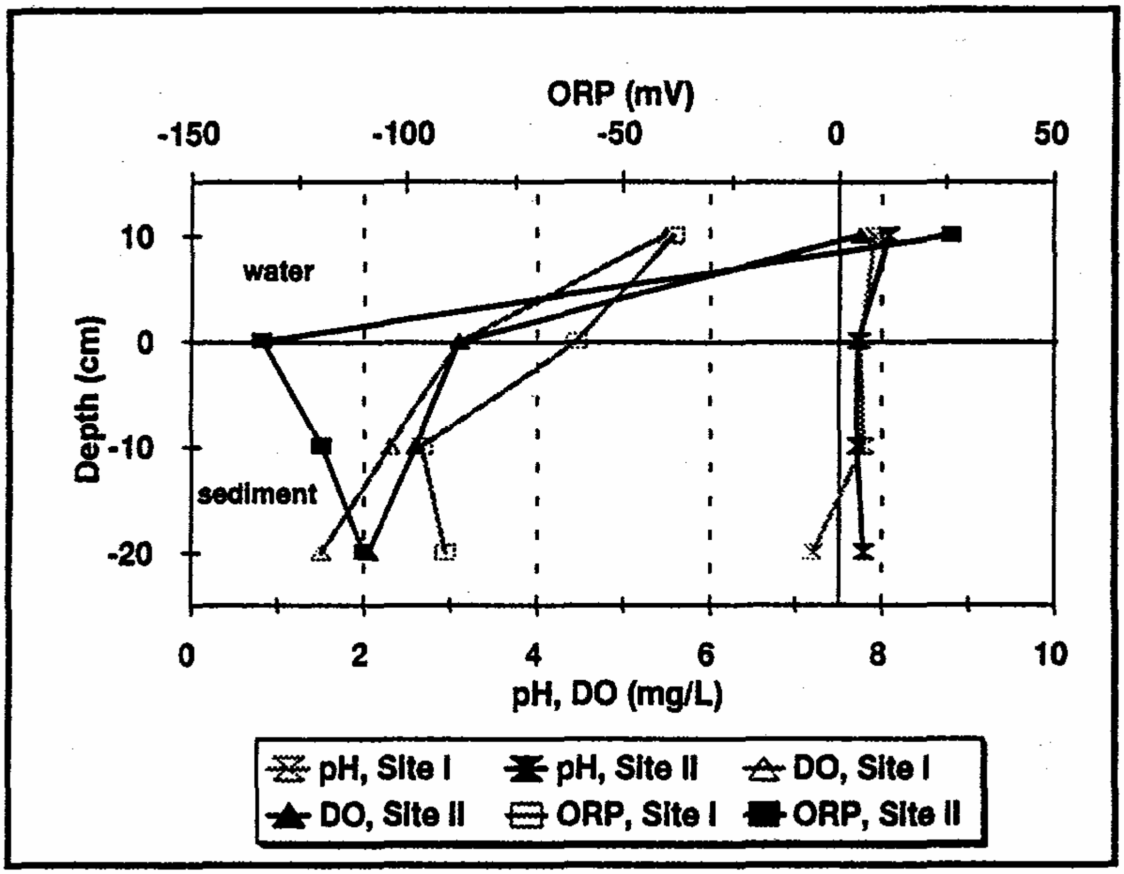
The sediment pore water profile demonstrates that reducing conditions are present in the sediment layer (Figure 4.2-42). Establishing a plant cover over non-acid-generating tailings, such as those found in the Brenda Mine tailings pond, was demonstrated to be feasible. While the Brenda Mine tailings are not acid producing, SRB activity is beneficial as it can lead to molybdenum precipitation in a biologically unavailable form.

Figure 4.2-41 **Aquatic Macrophyte Cover in Brenda's Tailings Pond in 1995**



Source: St. Germain *et al.* 1997

Figure 4.2-42 **pH, DO and ORP Profiles at Sites I and II in Brenda Tailings Pond**



Source: St. Germain *et al*. 1997

Table 4.2-17
1996 Plant Growth Estimation

Location	Period	Density	% of cover			
		stems/m^2	*Elodea*	*Myriophyllum*	*Zannichellia*	*Potamogeton*
Site I	July	10.2	88	8	4	0
	September	172	48	3.4	16	2
Site II	July	19	81	15	4	0
	September	23	67	25	4	4

Source: St-Germain *et al.* 1997

Heath Steele Tailings Pond:

The Heath Steele tailings pond is located 50 km north of Miramichi in northeastern New Brunswick. Data collection activities at the active Heath Steele tailings pond were similar to those undertaken at the Brenda Mine tailings area. A field trial was carried out at the Heath Steele tailings pond, specifically in the Lower Cell, to establish a biologically supported water cover with the objectives of inhibiting sulphide tailings oxidation, and reducing solids resuspension.

The trial was designed to assess the effect of various parameters on plant propagation and growth, wind and wave action, sediment type, plant species and plant sandwich shape. A total of 160 plant sandwiches were placed in two areas of the tailings pond referred to as the West and East sites. A control sandwich plot was also placed in the donor lake - Upper Tetagouche Lake. The tested plant species were: *Potamogeton robinsii* (pondweed), *Potamogeton richardsonii* and *Myriophyllum farwelli* (milfoil). The trial plots in the tailings areas did not establish plant growth. The researchers indicated that the high pH (9.5-10.5) of the pond was the probably the reason that a biologically supported water cover was not established. St. Germain *et al.* (1997) provide a good explanation of the effect of pH's greater than 9.4 on carbonate ion removal through $CaCO_3$ precipitation and the inhibition of aquatic plant growth.

Variations of Subaqueous Disposal Include:

- Relocation of exposed tailings to a water pond. This has been accomplished at several mine properties;
- Raising of the water pond level to flood exposed tailings. This may involve substantial works to construct impervious containment structures, reduce seepage losses, and divert additional water flows to a tailings management facility; and
- Disposal of tailings into a flooded pit, or ocean disposal.

Subaqueous disposal can form the basis of a part of an overall closure strategy and be applied to only flooded sections of a tailings management area. In these cases, other decommissioning strategies such as an engineered dry cover or elevated water table technique could be applied to the exposed tailings.

4.2.4 COST

MEND 5.8.1 reviewed the costs of applying various acidic drainage control and mitigation technologies. The results of the study suggest that a self-sustained water cover may represent one of the best technologies from a cost implementation perspective.

Estimated unit costs for several of the acidic drainage control and mitigation strategies assessed in MEND 5.8.1 are shown in Table 4.2-18.

Table 4.2-18
Summary of Estimated Costs for Acid Drainage Control and Mitigation Strategies Applicable to Tailings (in Year 2000 Dollars[(A)] per Hectare of Tailings Footprint)

Control Technology	Cost Estimates		
	Lowest Estimate	Single Estimate	Highest Estimate
Self-Sustained Water Cover	\$14,000[(B)] \$75,000[(C)]		\$270,000 \$370,000
Maintained Water Cover		\$ 12,000 \$ 88,000	
Composite Soil Cover with a Partial Water Cover		\$236,000 \$322,000	
Composite Soil Cover (No Water Cover)	\$245,000 \$309,000		\$326,000 \$440,000
Collect and Treat Acidic Drainage (No Water Cover)	\$66,000 \$227,000		\$103,000 \$252,000
Collect and Treat with Partial Water Cover		\$ 99,000 \$239,000	

Notes:
(A) Year 1994 dollars reported in Table 6.7, MEND 5.8.1, have been escalated at 1%/annum.
(B) Top value represents the capital cost.
(C) Bottom value represents the total cost including capital costs, collection and treatment costs, maintenance and other operating costs as applicable.

The highest estimated unit cost of \$370,000/ha in Table 4.2-18 for the total costs of a self-sustained water cover is based on a scenario where the site is poorly suited for a water cover. Actual costs for water cover programs are presented in Table 4.2-19.

Table 4.2-19
Selected Costs for Sites where Water Covers have been Applied

Site	Comment	Basin Size	Cost
Denison Tailings Management Area (Flooding)	water cover depth averages 1 m	240 ha 60 Mt tailings	~$25.7 M or ~$107,000/ha or $0.43/t
Quirke Waste Management Area (Flooding)	water cover depth averages 1 m	192 ha 44 Mt tailings	$36 M or ~$187,000/ha or $0.82/t
Solbec Tailings Area (Flooding)	1 m depth water cover	66 ha 4 Mt tailings	$2.1 M or $32,000/ha or $0.52/t
Louvicourt Tailings Area (Subaqueous Disposal)	minimum 1.2 m water cover depth	100 ha 14 Mt tailings	~$10.3 M or ~$103,000/ha or $0.73/t
Panel Waste Management Area (Flooding)	water cover depth averages 1 m	123 ha 16 Mt tailings	~$5.5 M or ~$126,000/ha or $0.97/t

Note: In Year 2000 Dollars. Year 1997 costs taken from MEND 7.2b have been escalated at 1%/annum

4.2.5 MEND AND RELEVANT PUBLICATIONS

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MEND 2.11.1a And — 1990. A Preliminary Assessment of Subaqueous Tailings Disposal in Anderson Lake, Manitoba. March.

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MEND 2.11.1b — 1990. Geochemical Assessment of Subaqueous Tailings Disposal in Buttle Lake, British Columbia. March.

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MEND 2.11.4a 1995. Geochemical Assessment of Subaqueous Tailings Disposal in Buttle Lake, British Columbia. 1993 Study Program. May.

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MEND 2.11.9 1998. Design Guide for the Subaqueous Disposal of Reactive Tailings in Constructed Impoundments. March.

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MEND 2.12.1c 2001. Subaqueous Disposal of Reactive Mine Tailings – Louvicourt Mine Test Cells – Geochemical Sampling and Analysis – Final Report.

MEND 2.12.1d 2001. Reactivity Assessment and Subaqueous Oxidation Rate Modelling for Louvicourt Tailings.

MEND 2.12.2e. 2001. Evaluation of Man-Made Subaqueous Disposal Option as a Method of Controlling Oxidation of Sulphide Minerals: Column Studies – Final Report.

MEND 2.13.1a 1998. Wet Barriers on Pyritic Uranium Tailings - Part I and II - Laboratory Lysimeter Studies of Oxidation and Limestone Neutralization Characteristics of Uranium Tailings and Waste Rock. January .

MEND 2.13.1b 1998. Wet Barriers on Pyritic Uranium Tailings - Part III - Laboratory Diffusion Lysimeter Studies of Uranium Tailings Deposited under a Shallow Water Cover. January.

MEND 2.13.2a 1993. Expérimentation de l'inondation du parc des résidus miniers - Solbec Cupra (Phase IV). Rapport Sommaire. Décembre.

MEND 2.13.2b 1994. Flooding of a Mine Tailings Site - Solbec Cupra. Suspension of Solids: Impact and Prevention. March.

MEND 2.13.2c 1994. Inondation artificielle du parc de résidus miniers Solbec Cupra: Études microbiologique et chimique - Rapport final. Mars.

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4.3 SATURATION

Research has demonstrated that the rate of oxygen diffusion through water is orders of magnitude lower than that through air. This important principle provides the basis for the use of moisture saturated pore spaces in tailings as a means of reducing the flux of oxygen from the atmosphere through tailings pore spaces to sulphide minerals. The saturation of pore spaces in waste rock piles as a means of controlling acid generation is normally not an option.

4.3.1 DISCUSSION OF THEORY

4.3.1.1 Oxygen Diffusion in Fine-Grained Soils

The theory related to the diffusion of oxygen through fine-grained soils, such as tailings, has been assessed in numerous studies including but not limited to MEND 2.22.2. The theoretical basis presented in MEND 2.22.2a is reiterated below to demonstrate the effect and benefit of moisture saturation in tailings.

It is often considered, at least in preliminary calculations, that oxygen transport is controlled by molecular diffusion in the gaseous or liquid phases of a fine grained soil, as temperature and pressure gradient effects are neglected (Collin 1987; Nicholson *et al*. 1989).

Oxygen diffusion is associated with concentration gradients between conditions in the atmosphere and the porous media. An equation for oxygen flux for a single layer having a thickness, L, can be written using Fick's Law for steady state or transient conditions, given by Bear (1972) and Shackelford (1991):

$$F(t) = -q_{eq} D^* \frac{\partial C(t)}{\partial z} = -D_e \frac{\partial C(t)}{\partial z} \tag{1}$$

and

$$\frac{\partial C(t)}{\partial t} = D^* \frac{\partial^2 C(t)}{\partial z^2} - k_r^* C(t) \tag{2}$$

with

$$q_{eq} = q_a + H q_w \tag{3}$$

where:

F(t) = diffusive flux of oxygen (kg/m^2/s),
D_e = effective diffusion coefficient of oxygen (m^2/s) and $D_e = \theta_{eq} D^*$,
C(t) = oxygen concentration in the gaseous phase at time t (kg/m^3),
z = depth (m),
t = time (s),

θ_{eq} = equivalent porosity (m^3/m^3), and θ_a and θ_w are the air and water volumetric content respectively,

H = solubility constant of gas in water, or Henry's constant which is 2.95×10^{-2} at 25°C

D* = diffusion coefficient (m^2/s), and

k_r^* = reaction rate constant (for sulphide oxidation) and $k_r^* = k_r/\theta_{eq.}$

In the above equation, θ_a and θ_w are related to the degree of saturation, S_r, and porosity, n, by the following expression: $\theta_w = nS_r$ and $\theta_a = 1 - nS_r$.

MEND 2.22.2a reports that analytical solutions to the equation in transient conditions (Equation 2) have been given by Crank (1975), for $k_r = 0$ (non-reactive cover system):

$$F(t) = 2C_o\, q_{eq} \sqrt{\frac{D^*}{pt}} \sum_{m=1}^{\infty} \exp\left[-\frac{(2m+1)^2 L^2}{4D^* t}\right] \qquad (4)$$

and

$$C(z,t) = C_o \; erfc \left(\frac{z}{2\sqrt{D^* t}}\right)$$

where

$$erfc \quad (u) = 1 - \frac{2}{\sqrt{p}} \int_o^u \exp(-v^2)\, dv$$

and m is an integer variable. These are obtained for the following limiting conditions:

$C = 0$ for $t = 0$ and $z \geq 0$,

$C = C_0$ for $t \geq 0$ and $z \leq 0$, and

$C = 0$ for $t \geq 0$ and $z \geq L$.

where:

C_0 = the atmospheric oxygen concentration.

These conditions imply a decreasing oxygen concentration in the layer (e.g. a cover) until it is reduced to zero at the base of the layer (e.g. at the top of the reactive tailings).

The solution for steady state conditions (Equation 1) is:

$$F = \frac{-D_e(C_L - C_o)}{L} \qquad (5)$$

where:

C_L = the concentration of oxygen below the cover and L is the cover thickness.

D_e can be calculated from the modified Millington-Shearer model given by Collin (1987) and Aubertin *et al*. (1993):

$$D_e = (1 - S_r)^2 \quad [n\,(1 - S_r)]^{2x}\ D_a + S_r^2\ (n\ S_r)^{2y}\ H\ D_w$$

where:

D_a	=	diffusion coefficient of oxygen in air (1.8×10^{-5} m^2/s),
D_w	=	diffusion coefficient of oxygen in water (2.5×10^{-9} m^2/s), and
x,y	=	material parameters, expressed as function of n and S_r.

To estimate the value of D_e, results of Reardon and Moddle (1985) were used to calibrate the above model and obtain typical parameter values for sand and tailings. As shown in Figure 4.3-1, $D_e \approx D_w$ when $S_r \approx 90\%$ (e.g. the D_e value for the humid porous system approaches that of water).

Based on the above equation (Equation 1), a calculation for steady state flow in a simple cover layer, using typical parameters (i.e., C_o = 21%, n = 0.41, S_r = 0.87, L = 1.1 m, $D_e = 4.61 \times 10^{-9}$ m^2/s), the oxygen flux (F(t)) was determined to be 0.036 $kg/m^2/yr$ O_2. The equations can also be applied to calculate the efficiency of a cover layer for steady-state conditions, as defined by Nicholson *et al.* (1989):

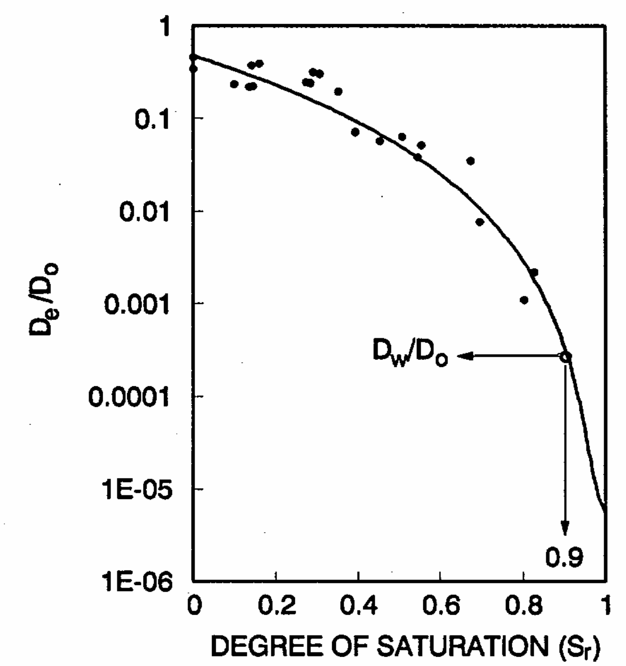
$$E_c = \frac{F_r}{F_c} = \left(\frac{\sqrt{k_r\ (D_e)_r}}{(D_e)_c} L \right) + 1$$

where:

F_r	=	oxygen flux through the reactive tailings without a cover,
F_c	=	oxygen flux through the cover, and
$(D_e)_{r,c}$	=	effective diffusion coefficient in the reactive tailings and cover material.

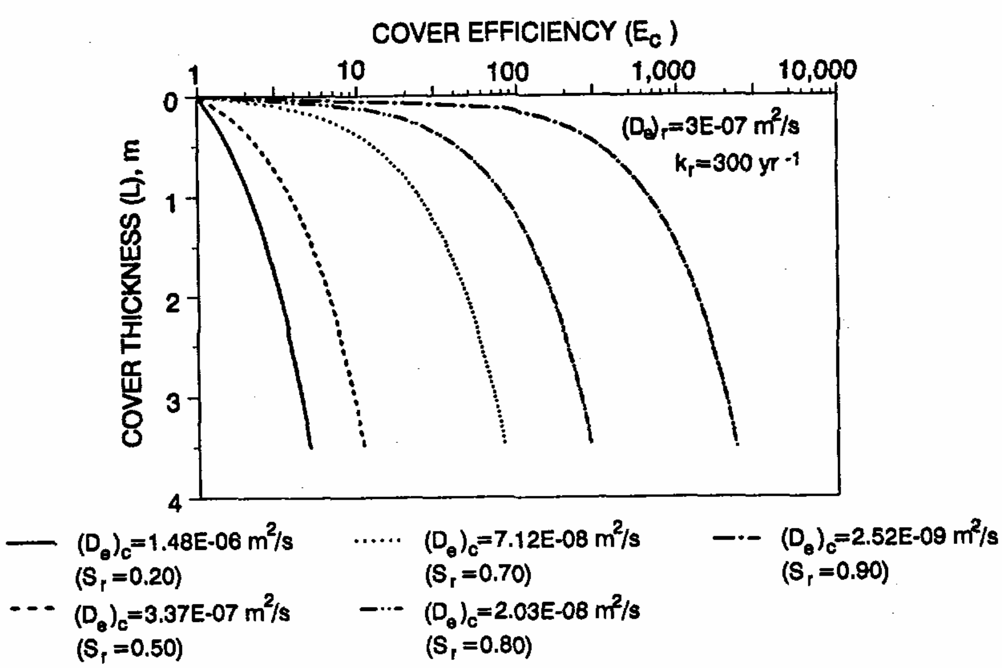
Figure 4.3-2 shows calculated E_c values as a function of L and S_r, for k_r = 300/a. The results compare well with those of Nicholson *et al.* (1989) obtained for slightly different conditions. These show that increasing the thickness of the cover above about 0.5 to 1 m does not significantly change its efficiency. These results show that a cover 1 m thick with S_r = 90% is more efficient than one of 4 m having an S_r = 80%. As such, the degree of saturation is a critical parameter for the design of moisture retaining dry covers. Bussière *et al*. (1995) indicate that a high level of saturation needs to be maintained in the fine-grained, moisture-retaining layer of a dry cover to avoid cracking due to evaporation, and to mitigate the diffusion of atmospheric oxygen through air filled pore spaces.

Figure 4.3-1 **Effective Diffusion Coefficient Expressed as a Function of the Degree of Saturation**



Source: MEND 2.22.2a

Figure 4.3-2 **Relationship Between the Cover Thickness and its Efficiency to Reduce Oxygen Flux**



Source: MEND 2.22.2a

For a complex cover made from several horizontal layers, the oxygen flux can be calculated using Fick's First Law for steady-state flow, equating the flux at each interface, to give the following equation (MEND 2.21.1):

$$\left(\frac{D_i}{L_i}\right)C_{i-1} + \left(\frac{D_{i+1}}{L_{i+1}} - \frac{D_i}{L_i}\right)C_i + \left(\frac{D_{i+1}}{L_{i+1}}\right)C_{i+1} = 0$$

where index, i, is given for layer i. Assuming an atmospheric oxygen concentration at the surface and a zero concentration at the top of the covered reactive tailings, one can calculate that the same flux of oxygen is obtained for the single layer cover with L = 1.1 m, and for a three-layer system in which the capillary barrier (n = 0.41, S_r = 90%) is only 0.5 m thick. Table 4.3-1 compares the two scenarios. This result is possible because the layered system helps to maintain a high degree of saturation in the capillary barrier, which in turn reduces the value of the effective diffusion coefficient that largely controls oxygen flux.

Table 4.3-1
Oxygen Flux Calculations for Cover Systems

Cover	Material	N	S_r	L (m)	D_e (m^2/s)	O_2 flux Kg/m^2/yr
One layer	Tailings	0.41	0.87	1.10	4.61 x 10^{-9}	0.036
Three layers	Sand	0.39	0.10	0.30	1.87 x 10^{-6}	0.036
	Tailings	0.41	0.90	0.50	2.52 x 10^{-9}	
	Sand	0.39	0.10	0.30	1.87 x 10^{-6}	

Source: MEND 2.22.2a; Aachib *et al*. 1994

Elberling *et al*. (1994) report that based on experimental data in tailings, the effective diffusion coefficient (D_e) can be expressed as follows:

$$D_e = D_a t\ (1 - S_r)^a \ + D_w t S_r \cdot H$$

where:

- D_e = effective diffusion coefficient in cover material (m^2/s),
- D_a = diffusion coefficient of oxygen in air (m^2/s),
- D_w = diffusion coefficient of oxygen in water (m^2/s),
- τ = experimental parameter,
- S_r = degree of saturation (volume of water/volume of pore space),
- H = Henry's constant, and
- α = experimental parameter.

Chao *et al*. (1991) devised a mass spectrometer-based method for the rapid measurement of the effective diffusion coefficient in till covers. For compacted till covers, the fitted parameter values were $\tau = 0.032$ and $\alpha = 3.92$. The effective diffusion coefficient with field samples varied from 5.64 x 10^{-7} m^2/s to 1.66 x 10^{-8} m^2/s, depending on the degree of saturation (0.20 to 0.86). The dependence of the effective diffusion coefficient (D_e) on the degree of saturation, and the fitted parameter values determined above ($\alpha = 3.92$) suggests that the diffusion coefficient can vary over five orders of magnitude. The most significant attenuations in D_e occur at high levels of moisture saturation.

4.3.1.2 Tailings Saturation using an Elevated Water Table

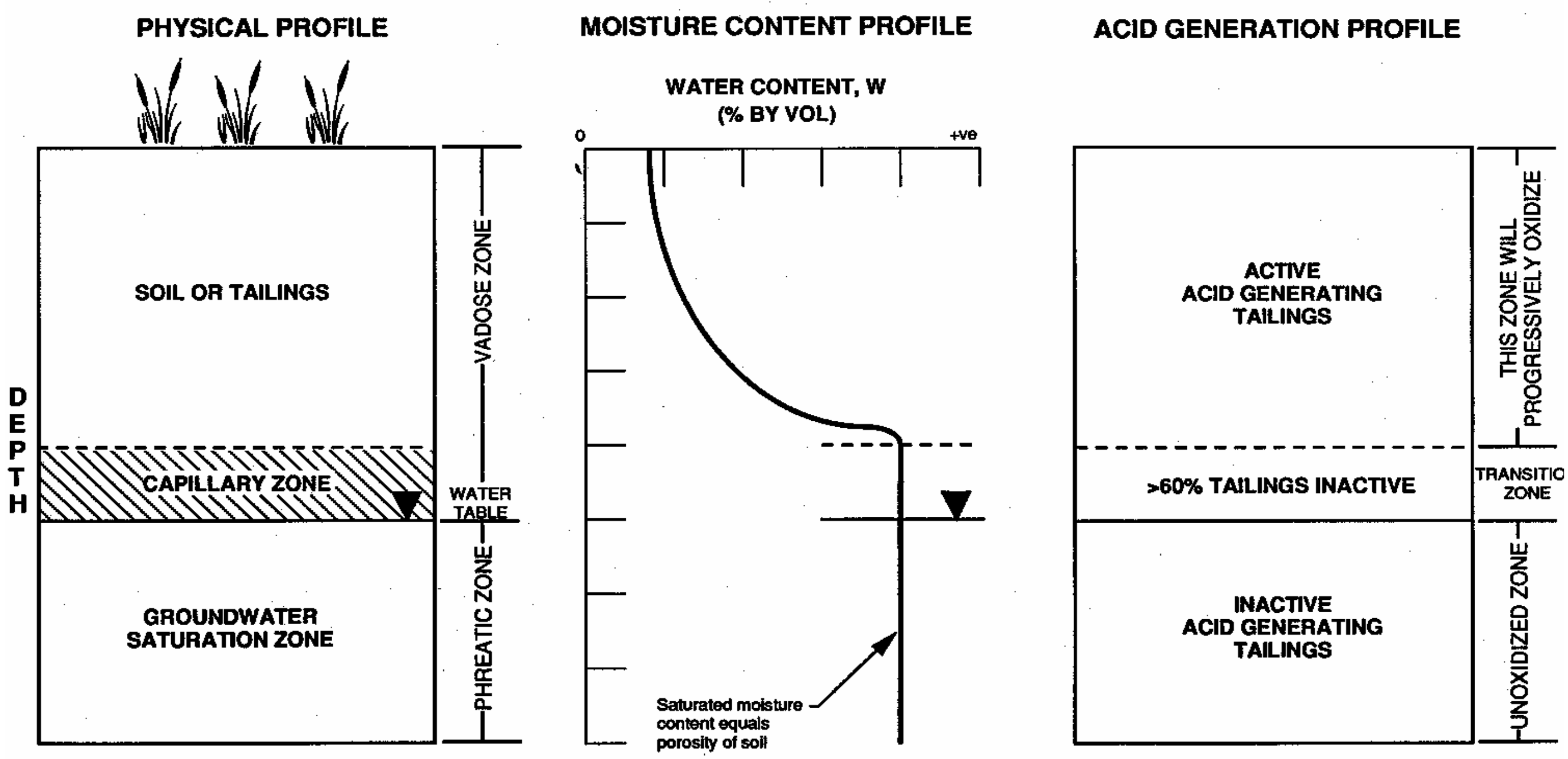
Elevated water table concepts involve the raising of the level of the saturated zone within a tailings mass so that reactive tailings are maintained in a sufficiently saturated state to inhibit sulphide oxidation. The primary role of an elevated water table in tailings is to reduce the total depth of unsaturated tailings exposed to oxygen. As indicated previously, high levels of moisture saturation act as a diffusion barrier to atmospheric oxygen.

An example of a tailings groundwater profile is shown in Figure 4.3-3, where:

- The water table within the tailings represents the upper surface of a zone of saturated tailings, and is a surface upon which the tailings pore water pressure equals the atmospheric pressure. The water table is also referred to as the phreatic surface;
- The vadose zone is located above the water table. The tailings pore spaces in this zone are generally under less than atmospheric pressure, however, some pore spaces may contain air or other gases at atmospheric pressure; and
- The capillary fringe extends upwards from the water table. In the capillary fringe, the tailings pore spaces are filled with water at a pressure that is less than atmospheric, and the pore water is held above the phreatic surface by capillary forces.

The height of the capillary fringe above the water table depends on the interstitial pore size which is related to the particle grain-size distribution. Specifically, the smaller the pore size the greater the surface tension and consequently the higher the capillary rise above the water table. As such, the capillary rise is higher for fine-grained tailings in comparison to that for coarse-grained tailings. In comparison, waste rock piles composed of coarse particles with large interstitial pore spaces do not have elevated water table conditions.

Figure 4.3-3 **Groundwater Profiles**



Source: MEND 2.17.1

Many, but not all, tailings contain a grain-size distribution ranging from medium sand to clay sized particles with 70 to 90% of the material less than 74 µm (200 mesh) in size. During conventional tailings placement, beach type deposits are formed as the particles are hydraulically separated by grain-size, density, and shape. Relatively coarse-grained and denser material settles closer to the discharge point(s) at the top of tailings beaches while fine-grained and less dense tailings particles including slimes tend to settle further downgradient as indicated in Figure 4.3-4.

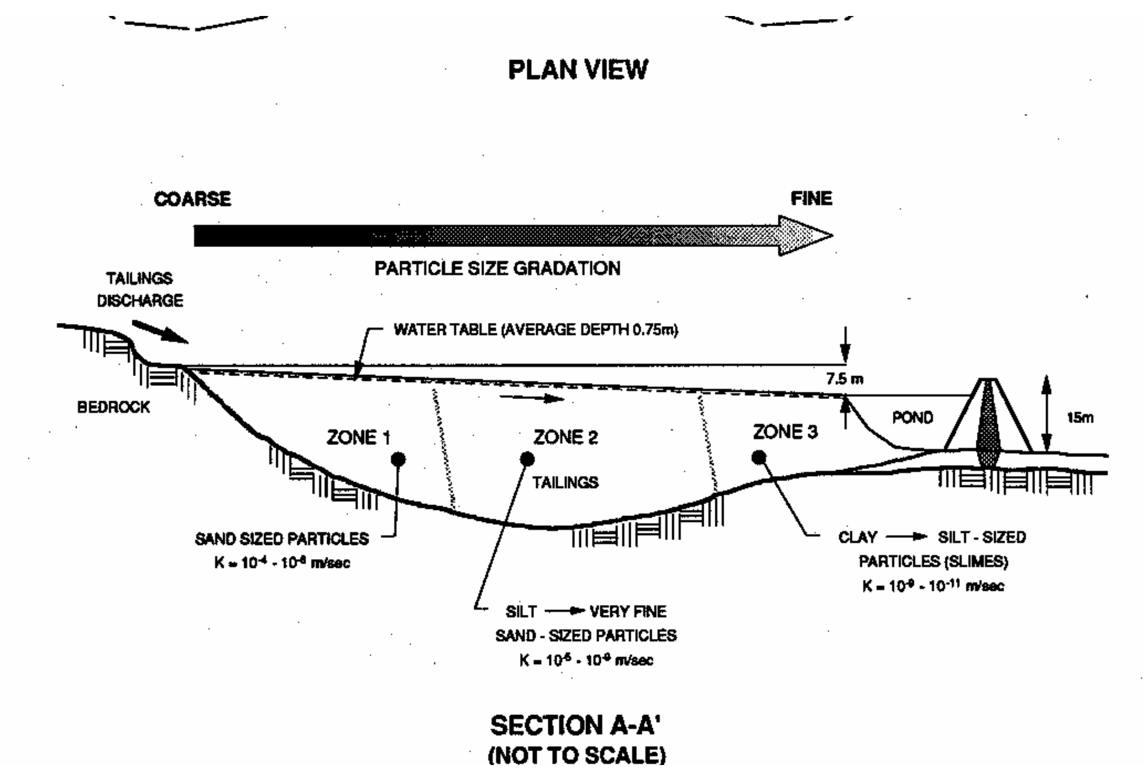
The depth to the water table determines the depth of the near-surface zone of exposed and drained sulphide tailings. This zone contains tailings that are readily available for acid generation. The elevated levels of moisture saturation that occur above the water table can extend close to the tailings surface when the water table is shallow or if there is strong capillary action (e.g. non-segregated tailings).

At active tailings sites, the surface layer of sulphide tailings is regularly wetted and the tailings water table is recharged as additional tailings are deposited. Once tailings deposition ceases, the degree of saturation of the interstitial pore spaces in the near-surface tailings zone may decrease and allow atmospheric oxygen to diffuse to exposed, unsaturated sulphide minerals. This process is common in many tailings areas.

The depth from the tailings surface to the water table that is sufficient to reduce the rate of sulphide oxidation is not immediately obvious. An elevation that is close to, but below the tailings surface is best from the perspective of reducing the diffusion of oxygen into the tailings, but has two drawbacks:

- A water table at the surface can lead to physical instability of the tailings surface. Granular porous media such as tailings are most stable when drained or at least under negative pore pressure - in tailings, this condition occurs above the water table. As the water table elevation in tailings increases, the effective stress in the media decreases and the shear stress required to move the tailings decreases. This increases the potential for the tailings to liquefy and flow as a result of a disturbance. Tailings containment structures must be capable of safely accommodating the raising of the water level within tailings, and a water pond (if present); and
- A shallow water table can lead to the discharge of near surface tailings pore water to surface runoff during precipitation events. The rapid discharge of shallow tailings pore water during rainfall events at the Nordic tailings area at Elliot Lake, Ontario, was reported by Blowes and Gillham (1988).

Figure 4.3-4 **Particle Size Distribution and Hydraulic Conductivity with Distance from Source**



Source: MEND 2.17.1

As a guideline, the depth to the water table should not fall below the Air Entry Value (AEV) of the tailings at the surface. The AEV value is a measure of the suction pressure created to begin draining water from the tailings. This suction can pull water from the water table upwards, thus creating near-saturated conditions well above the water table. Thickened and paste (non-segregated) tailings have very high suction values (in the order of several metres) and have the potential to retain high levels of moisture saturation above the water table.

The rates of sulphide oxidation below an elevated water table would be expected to be similar to rates under a surface water cover. As suggested by the theoretical analysis described previously, these rates will be controlled by the infiltration of water containing dissolved oxygen rather than by atmospheric oxygen diffusion. Under saturated conditions, diffusion rates become so low that infiltration dominates but at very small rates. According to Nicholson *et al*. (1989), these rates would be on the order of 0.09 mole/m^2/a (based on 10 mg/L of dissolved oxygen and an infiltration rate of 0.3 m/a).

4.3.2 DISCUSSION OF MEND RESEARCH

MEND 2.17.1 reviewed the use of elevated water tables as a method to control and reduce acid generation in tailings. The study identified the following three methods of creating elevated water conditions:

1. Modify the Water Balance of the Tailings;
2. Enhance the Water Retention Characteristics of the Tailings; and
3. Construct Groundwater Flow Barriers within the Tailings.

4.3.2.1 Modify the Water Balance of the Tailings

At some sites, the water balance of a tailings management facility can be modified to increase the elevation of the water table within the tailings. The water input must be higher, or output lower, to effect a rise in the water table elevation. Precipitation, runoff flowing to the tailings, groundwater, mill discharge, and local surface water drainage systems represent potential sources of water to tailings, while evaporation, runoff flowing from the tailings, groundwater flow, water pond discharge, and seepage represent water losses (Figure 4.3-5).

The tailings water balance may be modified by:

- Increasing water input to the tailings through the diversion of a natural drainage system to the tailings. This may include the collection of runoff from the surrounding watershed;
- Reducing tailings pore water seepage losses through the construction of low permeability perimeter containment structures and/or perimeter seepage barriers; and

- Reducing water losses due to evaporation, and/or improving the infiltration of water into the tailings.

4.3.2.2 Enhance the Water Retention Characteristics of the Tailings

<u>Thickened tailings:</u>

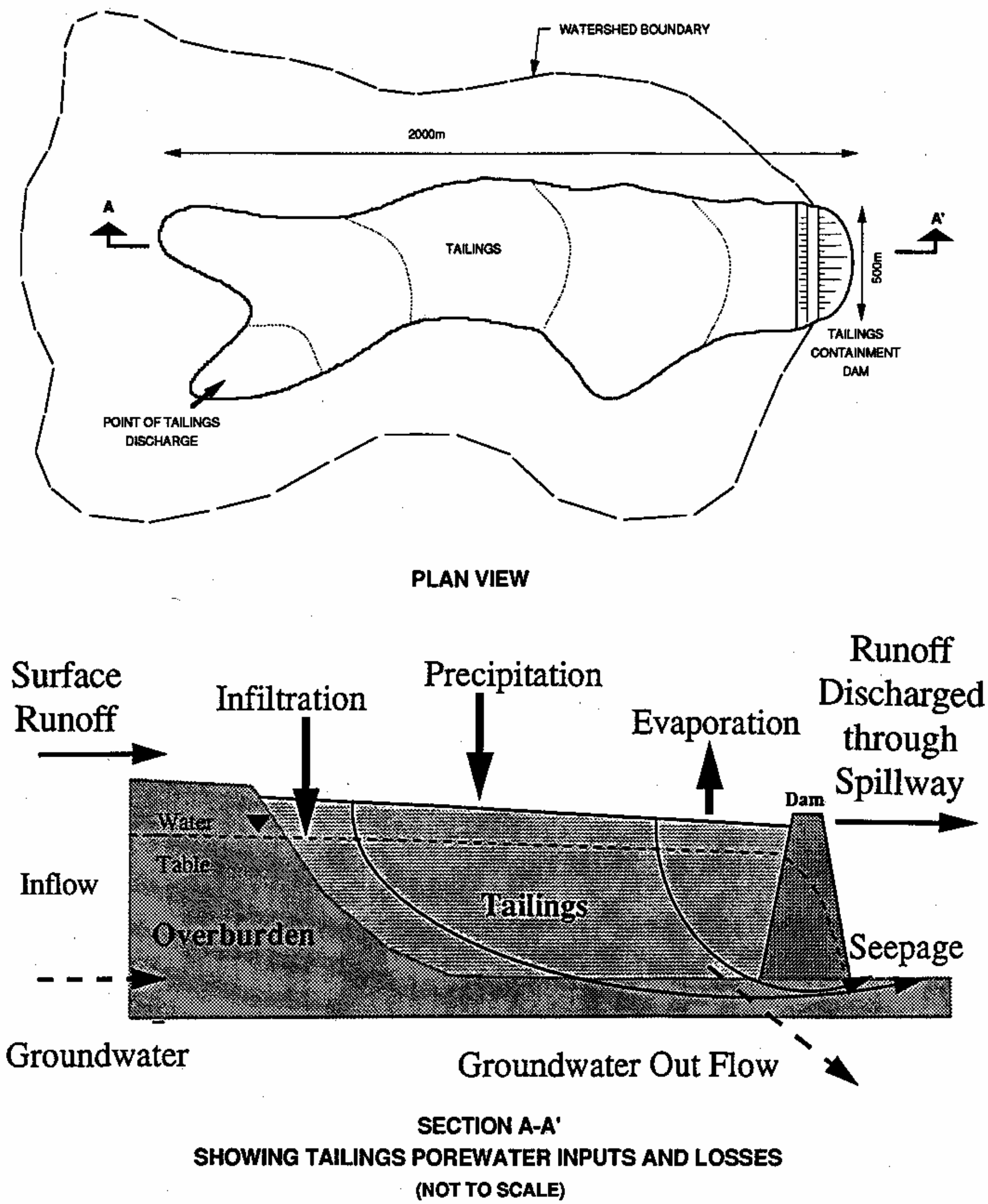
Thickened tailings create a very high level of saturation with the capillary fringe extending upwards to near surface. Thickened tailings are generally defined as tailings that are comprised of greater than 50% solids that are disposed into flat-sloped (e.g. 2 to 6% slope) mounds forming a cone. Falconbridge's Kidd Metallurgical Division in Timmins, Ontario, was the first metallurgical complex to use the thickened tailings disposal (TTD) system. The system has been progressively improved to its present status where tailings are thickened in an Outokumpu high-rate thickener, to excess of 60% solids and then discharged from an elevated position to form a cone-shaped mound. As the approach was developed by Dr. E.I. Robinsky (1975, 1978, 1999, 2000), the tailings mound is also known as the Robinsky cone. The Robinsky cone has been studied under numerous projects including, but not limited to, MEND 2.23.2ab, MEND 2.23.2c and MEND 2.23.2d.

With respect to acid generation, the most important characteristic of thickened tailings is the formation of a homogeneous mass of low hydraulic conductivity and high moisture content tailings. Robinsky *et al*. (1991) indicated that thickening promotes the development of a shallow water table and the near-saturation of surface tailings. These aspects are also discussed by Williams and Seddon (1999) in their review of thickened tailings discharge experience in Australia.

MEND-sponsored studies at the Kidd Metallurgical Division's tailings management area (MEND 2.23.2) have revealed that these tailings have an air entry value of approximately 6 m (or 60 kPa), as shown in Figure 4.3-6. This suggests that the tailings will remain effectively saturated up to 6 m above the water table.

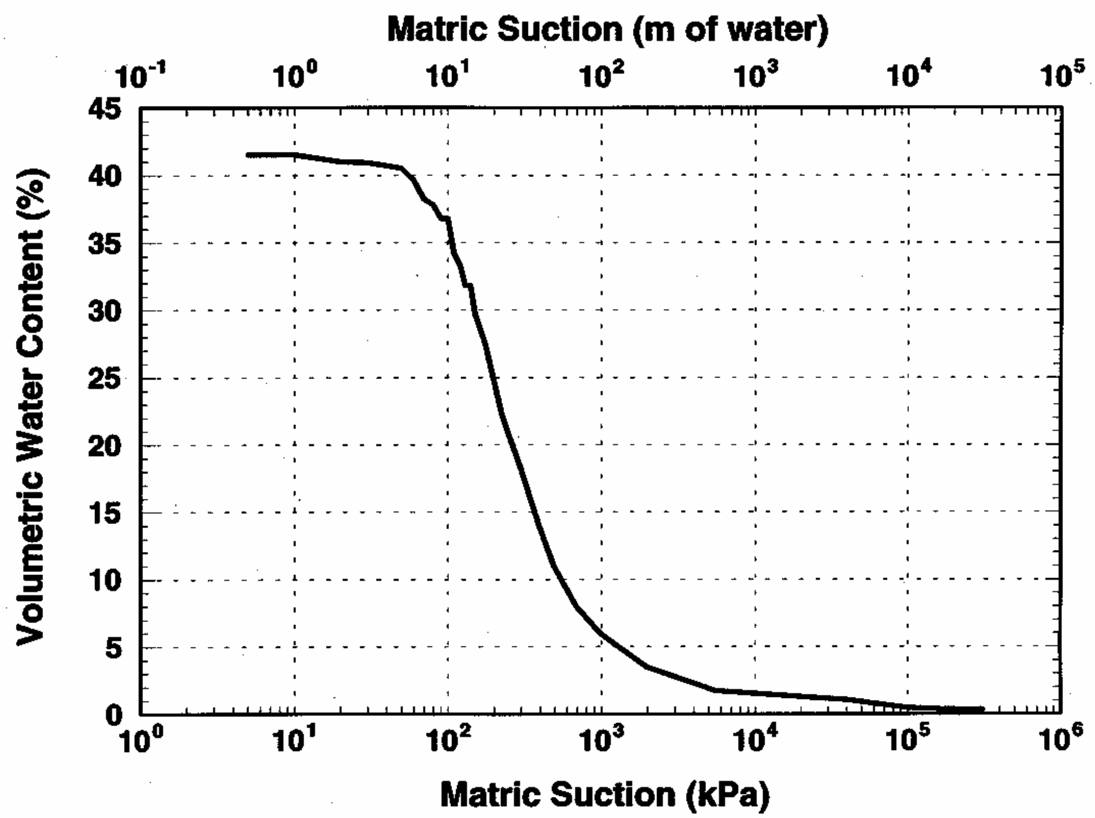
In concept, non-acid-generating tailings produced at the mine site could be used to cap unthickened sulphide tailings (Bussière *et al.* 1998; MEND 2.22.3) during the final years of operation. The near surface tailings would have excellent water-retention properties, which would in turn substantially reduce the depth of oxidation. Following closure, a soil cover could be placed over the thickened tailings cap to reduce water losses due to evaporation and encourage capillary rise into the non-acid generating tailings cover.

Figure 4.3-5 **Tailings Pore Water Inputs and Losses**



Source: MEND 2.17.1

Figure 4.3-6 Drainage Curve of the Kidd Creek Tailings



Source: MEND 2.23.2ab (after Yang *et al.* 1992)

<u>Paste tailings:</u>

Pastes are dense, viscous mixtures of tailings and water which, unlike slurries, do not segregate when allowed to rest (Cincilla *et al.* 1997). Landriault (1995) reports that large-tonnage tank dewatering (paste production) systems have been used for paste fill production at two Canadian mines. The dewatering tank system has achieved consistent operational performance, allowing for the development of a range of potential applications for paste production and disposal, both underground and on surface. Cincilla *et al.* (1997) also report that developments in equipment design have made practical the consistent production of low-moisture content tailings/water paste mixtures with unique rheological and water-retention properties.

Interest has grown in the application of paste technology to the surface disposal of tailings. Tenbergen (2000) reports that paste fill operations are becoming more common and offer advantages over traditional backfilling and surface disposal methods. Applications would have to suit local conditions since every paste application is unique due to the mineralogical and size distribution characteristics of the tailings, the process history, and the application of the paste.

A paste has a relatively low water content of 10% to 25% (Brackebusch 1994). Cement and larger particles of aggregate can be added to a paste without greatly changing the pipeline transport characteristics. A paste may bleed water when it is allowed to remain undisturbed for several hours. The angle of repose of paste ranges from 5% to nearly 30%.

Naylor *et al*. (1997) describe the tailings paste production and storage operation used at the Kinross Gold Corporation, MaCassa Mine in Kirkland Lake, Ontario. The operation involved the mixing of tailings paste with alluvial sand to produce an engineered quality paste backfill for use underground. Underground disposal of sulphide tailings may be beneficial at sites where sufficient tailings storage capacity is not available on surface. Dodd and Paynter (1997) describe the paste backfill plant operation at the Golden Giant Mine in Marathon, Ontario, where a 6 inch slump paste is produced and distributed for use underground. The reduction in the quantity of tailings requiring disposal in a tailings disposal facility on surface allowed a planned expansion of the tailings disposal facility to be deferred, and reduced in size. The mechanical properties of paste backfill, based on field testing, are described in Ouellet and Servant (2000).

The potential advantages of paste technology for tailings disposal are reduced land usage and lower retaining structures. Paste technology may be used to produce a homogeneous tailings deposit which can be compacted to support reclamation equipment. The low permeability of consolidated paste would be expected to assist in inhibiting sulphide oxidation. Nguyen and Doell (1996) indicate that major advantages of paste tailings disposal include:

- No segregation of fine and coarse tailings particles;
- The dewatering of tailings prior to disposal;

- Reduced ice lens formation in comparison to conventional disposal in cold weather; and
- Pastes can be stacked due to their high angles of repose, and hence improve the storage capacity of a tailings management area.

4.3.2.3 Groundwater Flow Barriers within Tailings

In MEND 2.17.1, an alternate approach to creating an elevated water table in tailings was evaluated using two-dimensional computer modelling techniques. In brief, the approach involves the construction of flow barriers within tailings to reduce the often preferential downgradient flow of pore water. The envisaged effect would be to raise the water table in the areas upgradient of the barriers. A number of options were considered to attain an increase in the water table elevation in a tailings impoundment. These included decreasing outflow by the reconstruction of a perimeter dam, decreasing evaporation to increase infiltration, and use of groundwater flow or hydraulic barriers constructed *in situ*. The modelling study investigated the relative magnitude of effect of each of these options in elevating the water table. The results show that all of the methods considered may increase the water table elevation.

Slurry trench cutoff walls were used to reduce impacts to the groundwater system adjacent to a tailings impoundment in northern Ontario. The permeability testing of the soil-bentonite and cement-bentonite (CB) slurry trench cutoff walls is described in Plewes and Semenoff (1999). The average mass permeability of the CB was approximately 5 x 10^{-6} cm/s – a value consistent with literature for CB slurry trench walls.

4.3.3 APPLICATION AND LIMITATIONS

The applications of elevated water table concepts offer additional potential reasonable cost closure options for tailings management areas. Site-specific conditions will determine the suitability of these concepts for use at a mine property.

Elevated water tables have been an intrinsic component of closure strategies for tailings management facilities but have only recently become proposed or applied as a principal basis of closure plans. As such, experience in the application and performance assessment of elevated water tables is recent and being accumulated. The developing state of knowledge is evidenced by the relatively limited data available regarding the application of elevated water tables to sulphide tailings impoundments. Nine sites (Table 4.3-2), however, provide some degree of relevant experience:

- Five sites (the Kidd Metallurgical Division in Ontario, the Greens Creek Mine in Alaska, the Cluff Lake Mine in Saskatchewan, Les Mines Selbaie in Québec, and the Elura Mine in Australia) have used thickened tailings disposal. These sites represent a broad range of site conditions, and tailings disposal and closure strategies; and

- Three sites (the Falconbridge New Tailings site, the Dona Lake Mine site, and the Stanrock Mine site - all in Ontario) involve the modification of the water balance of the tailings. At Falconbridge, research has been carried out to investigate the important relationships between oxygen consumption rates within tailings, the degree of saturation, and depth to the tailings water table. At Dona Lake, a closure plan is being implemented which relies in part on the elevation of the water table within the coarse tailings beach area. At Stanrock, spigotted tailings dams were replaced with engineered water retaining structures (Ludgate *et al*. 2000).

Table 4.3-2
Sites That Provide Relevant Experience

Site Study No.	Mine Site	Approximate Location	Elevated Water Table Concept
1	Elura	New South Wales, Australia	Thickened Tailings
2	Greens Creek	Alaska, USA	Thickened Tailings
3	Cluff Lake	Lake Athabaska District, Saskatchewan	Thickened Tailings
4	Les Mines Selbaie	Joutel, Québec	Thickened Tailings
5	Kidd Creek	Timmins, Ontario	Thickened Tailings
6	Falconbridge New Tailings	Falconbridge, Ontario	Modified Water Balance
7	Dona Lake	Pickle Lake, Ontario	Modified Water Balance
8	Stanrock	Elliot Lake, Ontario	Modified Water Balance
9	Sturgeon Lake	Ignace, Ontario	Combination of a Modified Water Balance and a Groundwater Flow Barrier

Source: MEND 2.17.1

The Sturgeon Lake site in Ontario involves a combination of a modified water balance and a groundwater flow barrier.

Chemical treatment of acidic drainage may be required, in some instances, after the application of an elevated water table concept. Sources of acid production could include sulphide solids in the near-surface and not sufficiently saturated zone, and fluctuations in the water table elevation. The effect of water table fluctuation could be mitigated in some cases by the use of a cover to reduce evaporative losses, increase infiltration, or to elevate the water table in the cover. The additional and interim use of chemical treatment may be cost beneficial when compared to other options.

Numerical modelling techniques can be used to predict the performance of elevated water table concepts in controlling or reducing acidic drainage. These techniques draw upon soil science and civil engineering methods of predicting water movement in soil, and can be complemented by numerical models for the prediction of acidic production in tailings. These models are important as they provide a means of preliminarily assessing the performance of a closure option, and of addressing the effect of changes in ambient conditions (e.g. drought conditions).

4.3.4 COST

Preliminary and conceptual cost comparisons were carried out in MEND 2.17.1 based on the closure of three types of tailings impoundments: a tailings stack; a valley impoundment underlain by a pervious zone; and a valley impoundment underlain by an impervious zone. Closure options included the perpetual collection and treatment of acidic drainage, the use of an engineered cover, and applications of elevated water table concepts. The first order estimates of closure costs indicate that elevated water table concepts, when suitable, can provide significant closure cost savings in comparison to collection and treatment, and the use of an engineered dry cover.

4.3.5. MEND AND RELEVANT PUBLICATIONS

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4.4 DRY COVERS

Dry cover systems as a closure option for management and decommissioning of waste rock and tailings is a common prevention and control technique used at numerous sites around the world. The objectives of dry cover systems are to minimize the influx of water and provide an oxygen diffusion barrier to minimize the influx of oxygen. Apart from these functions, dry covers are expected to be resistant to erosion and provide support for vegetation.

Dry covers can be simple or complex, ranging from a single layer of earthen material to several layers of different material types, including native soils, non-reactive tailings and/or waste rock, geosynthetic materials, and oxygen consuming organic materials. Multi-layer cover systems utilize the capillary barrier concept to keep one (or more) of its layers near saturation under all climatic conditions. This creates a "blanket" of water over the reactive waste material, which reduces the influx of atmospheric oxygen and subsequent production of acidic drainage.

Research projects on "dry" covers have been conducted under the auspices of the BC AMD Task Force and the MEND program since 1988. Table 4.4-1 lists the key MEND projects related to dry covers.

Table 4.4-1
Dry Covers Projects

MEND Project	Title
1.25.1	Soilcover Users Manual Version 2.0
1.41.4	Whistle Mine Waste Rock Study
2.20.1	Evaluation of Alternate Dry Covers for the Inhibition of Acid Mine Drainage from Tailings
2.21.1	Development of Laboratory Methodologies for Evaluating the Effectiveness of Reactive Tailings Covers
2.21.2	Field Evaluation of the Effectiveness of Engineered Soil Covers for Reactive Tailings: Volume 1-Laboratory and Field Tests; Volume 2-University Contracts
2.21.3a	Review of Soil Cover Technologies for Acid Mine Drainage – A Peer Review of the Waite Amulet and Heath Steele Soil Covers
2.21.3b**	Peer Review on Covers made with Non-Traditional and Solid Waste Compost
2.21.4**	Design, Construction and Monitoring of Earthen Covers for Waste Rock and Tailings
2.22.2a	Évaluation en laboratoire de barrières sèches construites à partir de résidus miniers
2.22.2b	Études de laboratoire sur l'efficacité de recouvrement construites à partir de résidus miniers
2.22.2c	Études sur les barrières sèches construites à partir de résidus miniers, Phase II – Essais en place
2.22.3**	Valorisation des résidus minier : une approche intégrée
2.22.4ae	Construction and Instrumentation of a Multi-Layer Cover – Les Terrains Aurifères
2.22.4af	Construction et instrumentation d'une couverture multicouche au site Les Terrains Aurifères Québec, Canada
2.22.4be	Field Performance of the Les Terrains Aurifères Composite Dry Cover

Table 4.4-1
Dry Covers Projects (continued)

2.22.4bf	Suivi du comportement du recouvrement multicouche – Les Terrains Aurifères
2.24.1	Manual of Methods Used in the Revegetation of Reactive Sulphide Tailings Basins
2.25.1a	Reclamation of Sulphide Tailings Using Municipal Solid Waste Compost: Literature Review and Recommendations
2.25.1b	Reclamation of Sulphide Tailings Using Municipal Solid Waste Compost: Laboratory Studies
2.25.2*	Design of an Organic Barrier using Wood Waste – East Sullivan
2.25.3	Evaluation of the Use of Covers for Reducing Acid Generation from Strathcona Tailings
2.25.5	Degradation Rate of Wood Waste Covers at the East Sullivan Tailings Impoundment
2.31.1a	Heath Steele Waste Rock Study, Phases I to III
2.31.1b	Engineering Design and Construction Phase IV – Composite Soil Cover Acid Waste Rock Study Heath Steele Mines
2.31.1b	Monitoring Report Phase IV – Composite Soil Cover Acid Waste Rock Study Heath Steele Mines
2.31.1c	Monitoring Program 1995-96 Phase V – Composite Soil Cover on Waste Rock Pile 7/12 at Heath Steele, New Brunswick
2.34.1	Evaluation of Field-scale Application of a Shotcrete Cover on Acid Generating Rock
2.35.2b	Evaluation of Techniques for Preventing Acidic Rock Drainage: Final Report
5.8.1	Economic Evaluation of Acid Mine Drainage Technologies
6.2	Polymer-Modified Clay as Impermeable Barriers for Acid Mining Tailings
W.010	Dry Covers Technologies Workshop, Sudbury, April 1996
W.015	Prevention Technologies for Acid Mine Drainage, Fredericton, November 1997
W.016	Acidic Drainage Workshops, Fredericton, Moncton, March 1998
BC.03**	7th Annual BC MEM – MEND 2000 Metal Leaching / ARD Workshop (Dry Covers, Predicting Solubility Constraints, Research and New Developments), Vancouver, November 2000
ME.02	Case Studies on Wet and Dry Covers for Tailings and Waste Rock, Sudbury, September 1999
IW.04	Dry Covers for Mine Tailings and Waste Rock, Vancouver, May 1997

* Open file report
** Not available at time of printing

The focus of discussion within this section is on the control of sulphide oxidation within reactive tailings and waste rock storage facilities using dry covers. Application of dry covers to non-reactive wastes is also addressed. Specifically, Section 4.4 discusses:

- Background information related to dry cover systems (4.4.1);
- Alternative dry cover system designs (4.4.2);
- Performance monitoring of dry cover systems (4.4.3);
- Case studies of dry cover systems for tailings and waste rock (4.4.4);
- Sustainable performance of dry cover systems (4.4.5); and
- Costs associated with construction of dry cover systems (4.4.6).

4.4.1 BACKGROUND

The background discusses the scope of the conceptual system, the purpose of dry cover systems, key theoretical concepts related to dry covers, the design objectives of dry cover systems and the factors that influence them and finally, design philosophies for dry cover systems.

4.4.1.1 Scope of the Conceptual System

Figure 4.4-1 illustrates the scope of the conceptual system involving a dry cover system on a waste rock pile. The scope includes the:

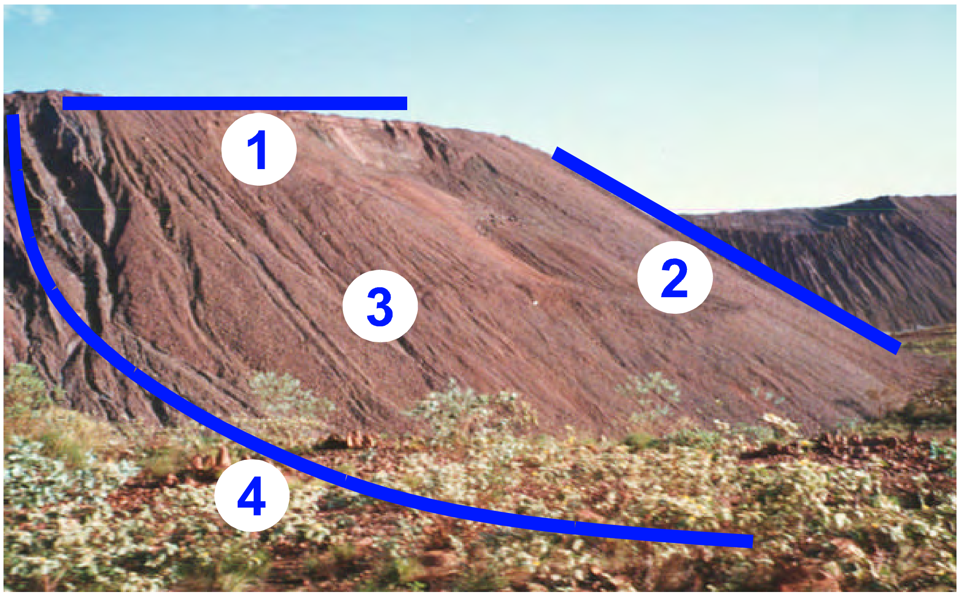
1. Performance of the dry cover on a relatively horizontal surface;
2. Performance of the dry cover on a sloping surface;
3. Internal hydraulic and geochemical performance of the waste material; and
4. Influence of basal flow as a result of placing the waste material on a valley wall, a groundwater discharge area, and/or historic surface water path.

Integration at the conceptual and detailed design stages of each component of the scope as listed above is the key to implementing the optimum dry cover system with respect to technical and economic feasibility. It will also ensure the best opportunity for long-term sustainable cover system performance as well as for developing a credible closure strategy.

In general, the first item is well understood and addressed during the design of dry covers. However, the second and fourth items will significantly influence the metal loading released from the waste material. For example, some documented case studies of “cover system failures” are in fact a result of the cover system being designed for a horizontal surface while being constructed on a sloping surface. The performance of a dry cover on a sloping surface can be much different as compared to a horizontal surface and the difference in performance relates to site climate conditions, the slope length and angle, and the material properties. Numerous other documented “cover system failures” can be attributed to the influence of basal flow resulting from placing the waste on valley walls, basins, groundwater discharge features, and historic surface water paths. In these cases, the release of acidic drainage from the waste storage facility following cover placement was not due to incident precipitation on the surface, but rather sub-surface basal flow leaching oxidation products from the storage facility.

A discussion on the reactivity of the waste underlying the dry cover and its influence on developing cover system objectives is provided in Section 4.4.1.4. However, it is suffice to mention here that the geochemical and hydraulic characteristics of the underlying waste have a significant influence on cover system performance.

Figure 4.4-1 **Scope of the Conceptual System: Cover Performance on a 1) Horizontal and 2) Sloping Surface, 3) Internal Hydraulic and Geochemical Performance, and 4) Influence of Basal Flow**



4.4.1.2 Purpose of Dry Cover Systems

The primary purpose of placing dry cover systems over reactive waste material is to minimize further degradation of the receiving environment following closure of the waste impoundment. This requires long-term control of the quality of surface runoff and seepage waters from the waste facility to protect both the local surface and groundwater systems. In addition, protection of the local air quality requires long-term control of dust and hazardous gas emissions from the surface of some waste storage facilities (e.g. uranium tailings impoundment).

Another important purpose of dry cover systems is to provide a medium for establishing a sustainable vegetation cover that is consistent with the current and final land use of the area. Achieving this goal will minimize the effects of water and wind erosion on the surface of the cover system, thereby providing stability of the engineered structure in the long term. The establishment of a vegetation cover using native species will also potentially restore wildlife habitat lost from the development of the waste facility. Vegetation can also be vital to the performance of dry cover systems, as discussed in the following sub-section.

4.4.1.3 Key Theoretical Concepts

A key point with respect to dry covers is that by definition they are part of an unsaturated system exposed to the atmosphere. The term “unsaturated” implies that the pore spaces within the cover system are not completely filled with water. This is particularly true for dry covers placed on waste rock piles. It is also a valid description of dry covers placed on tailings impoundments because a significant amount of tailings surface area is typically above the water table. The components of the unsaturated system include the atmosphere, vegetation, cover layer(s), and the underlying waste as well as any natural foundation materials. It is fundamental therefore that the design, implementation, and monitoring of dry cover systems be undertaken from an unsaturated perspective, whether it be with respect to hydraulic or geochemical performance.

The key theoretical concepts related to the design of dry cover systems are the soil-water characteristic curve and hydraulic conductivity function, the capillary barrier concept, and the relationship between degree of saturation and the diffusion of oxygen. MEND 2.21.4, MEND 2.21.3a and others describe these concepts in great detail; however, they are introduced briefly below.

<u>Soil-Water Characteristic Curve and Hydraulic Conductivity Function</u>

The soil-water characteristic curve (SWCC) is one of the essential components to dry cover system design. It represents the relationship between moisture content (or degree of saturation) and negative pore-water pressure (or suction) for a given soil. The SWCC, also known as the moisture retention curve, is useful in understanding the capability of a soil to store and release water under drainage and evaporation conditions. Figure 4.4-2 presents a schematic illustration of the SWCC for different soil types. The finer textured material retains moisture to a greater extent than the coarser textured material. In addition, a well-graded material will gradually decrease in moisture content, while a uniform material will be subjected to a sharp drop in moisture content. The negative pore-water pressure required to initiate drainage of an initially saturated soil is called the air entry value (AEV). The SWCC is obtained from a laboratory test in which the volumetric water content of a soil sample is measured at different applied suctions.

Hydraulic conductivity refers to the ability of a soil to transmit moisture, and at low negative pore-water pressures (i.e. below the AEV) is generally equal to the saturated hydraulic conductivity. Hydraulic conductivity is a function of pore-water pressure, and because moisture content is a function of pore-water pressure, hydraulic conductivity can also be plotted against moisture content. The former relationship is called the hydraulic conductivity function, and is shown in Figure 4.4-2. Important to note is the intersection of the hydraulic conductivity function for different soils. For example, a sand may have a higher hydraulic conductivity than a clay near saturation (i.e. low negative pore-water pressure), but may have a lower hydraulic conductivity when dry (i.e. high negative pore-water pressure).

Capillary Barrier Concept

The capillary barrier concept is commonly used in the design of dry cover systems and more specifically, the design of multi-layer cover systems. A capillary barrier results when a fine-grained soil overlays a coarse grained soil, as illustrated in Figure 4.4-3. The design of a capillary barrier is dependent on the hydraulic properties of both the coarse and fine soils. Capillary barriers, unlike compacted barriers, do not rely solely on low hydraulic conductivity to restrict moisture movement into underlying material. Processes that increase hydraulic conductivity, such as desiccation and freeze-thaw, do not necessarily decrease the effectiveness of a capillary barrier (MEND 2.21.4).

Rasmusson and Erikson (1986), Nicholson *et al.* (1989), Morel-Seytoux (1992), MEND 2.21.4, MEND 2.22.2a and others describe the capillary barrier concept in detail; however, a brief description is provided here for completeness. The lower coarse textured soil may drain to a condition of residual moisture content if conditions allow. The residual suction for coarse textured material is relatively low. The overlying fine-grained soil will not drain at this low suction and as a result, it remains in a saturated condition. This “capillary” break will occur during drainage whenever the residual suction of the lower coarse textured soil is less than the AEV of the upper fine-grained soil. A coarse-grained layer overlying a fine-grained soil layer may also be included in the design of a capillary barrier system to reduce evaporation from the fine-grained layer. The upper coarse textured soil layer can reduce runoff, if the intensity is not too extreme, because it provides for storage of water following infiltration, thereby allowing some water to reach the underlying fine textured soil and satisfy any antecedent moisture losses.

Figure 4.4-2 **The Soil-Water Characteristic Curve and Hydraulic Conductivity Function for Different Soil Types**

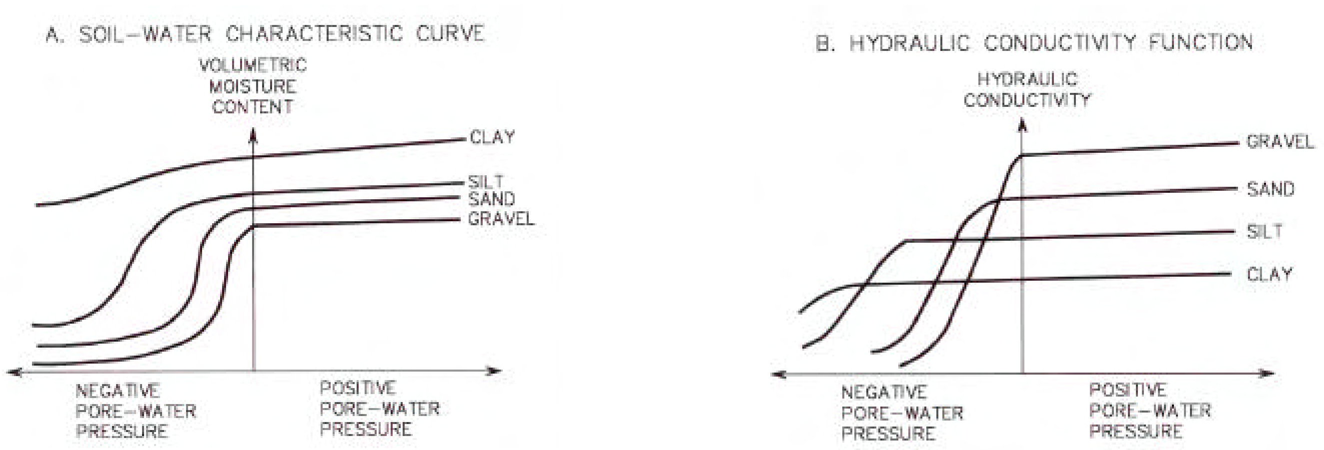
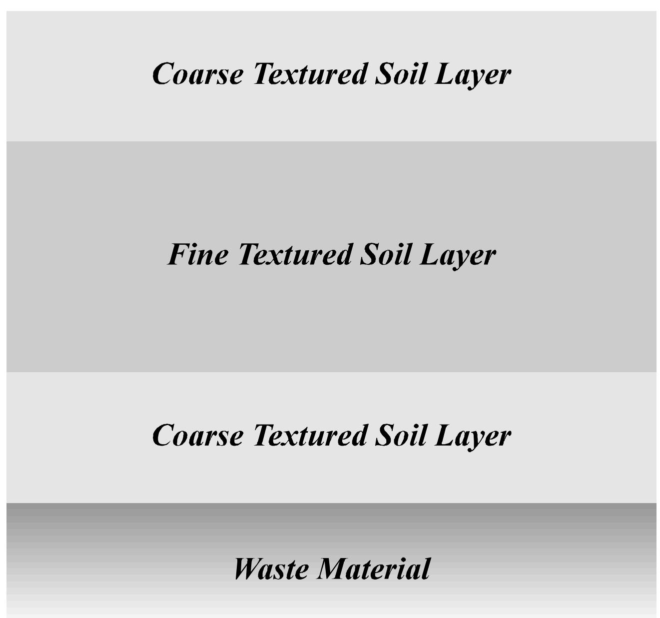


Figure 4.4-3 **A Multi-Layer Cover System over Waste Material**



Source: O’Kane 1996

In summary, the capillary barrier concept is utilized in the design of dry cover systems to keep a central, fine-grained layer near saturation under all climatic conditions. This in turn limits the ingress of oxygen due to low oxygen diffusion conditions; this phenomenon is explained in greater detail below. In addition, the lower permeability of the fine-grained soil layer (usually compacted), combined with the lower capillary barrier, provides a control on net infiltration to the underlying waste material. Capillary barriers are considered as an alternative in arid and semi-arid climates where maintaining a layer within the cover system at high saturation conditions (i.e. a compacted layer) may not be possible.

Relationship between Degree of Saturation and Diffusion of Oxygen

Limiting the ingress of atmospheric oxygen is generally the key design objective of a multi-layer or composite dry cover system placed over reactive waste material. Earthen materials can act as oxygen barriers provided a high degree of saturation is maintained within the soil. Degree of saturation represents the percentage of the volume of voids in the soil that contain water. If all the voids in the soil material are water-filled, then the predominant mechanism for oxygen to travel into the underlying waste is diffusion through the pore-water (Nicholson *et al.* 1989).

Figure 4.4-4 illustrates the effect of the degree of saturation of a soil on the diffusion coefficient for oxygen. The effect on the oxygen diffusion coefficient for soils with a degree of saturation between 0 and 50% is minimal. The oxygen diffusion coefficient in the soil is approximately equal to the diffusion coefficient in air. There is a substantial decrease in the oxygen diffusion coefficient at higher degrees of saturation.

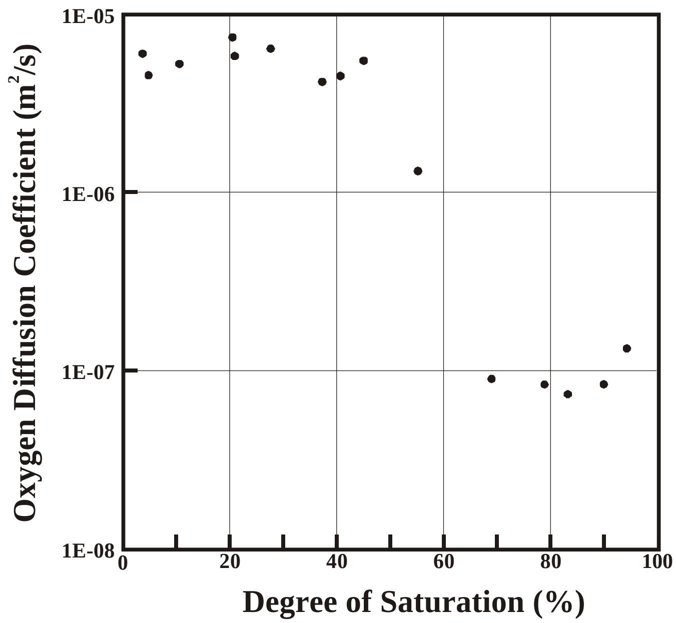
MEND 2.22.2a,b,c produced an extensive set of data on the relationship between the degree of saturation and oxygen diffusion coefficient. One of the findings of this study was that the diffusion of oxygen through the water phase of a relatively saturated earthen material should not be neglected, especially when conducting oxygen diffusion laboratory tests. In summary, research has demonstrated that an effective barrier to oxygen diffusion will result if the degree of saturation of a soil layer can be maintained greater than approximately 85 to 90%.

4.4.1.4 Dry Cover Design Objectives

The two principal design objectives of dry cover systems are:

1. To function as an oxygen ingress barrier for the underlying waste material by maintaining a high degree of saturation within a layer of the cover system, thereby minimizing the effective oxygen diffusion coefficient and ultimately controlling the flow of oxygen across the cover system; and
2. To function as a water infiltration barrier for the underlying waste material as a result of the presence of a low permeability layer and/or a moisture storage and release layer.

Figure 4.4-4 **Effect of Degree of Saturation on the Oxygen Diffusion Coefficient**



Source: Yanful 1993

Additional design objectives for dry cover systems placed on reactive tailings and/or waste rock can include:

- Control of consolidation and differential settlement;
- Oxygen consumption (i.e. organic cover materials);
- Reaction inhibition (i.e. incorporate limestone at the surface which does not prevent oxidation but can control the rate of oxidation); and
- Control of upward capillary movement of process water constituents/oxidation products.

The two principal design objectives of dry cover systems are described in more detail below. A classification for dry cover systems is also presented.

Limiting the Influx of Atmospheric Oxygen

Controlling the influx of atmospheric oxygen is a key component of the design of a closure system for disposal facilities containing reactive waste material. Water covers have generally been accepted as the most suitable method of limiting atmospheric oxygen to sulphide-bearing mine waste in semi-humid to humid climates. However, water covers are not feasible at all locations due to various technical factors (e.g. surface topography, hydrological conditions, and long-term stability of retaining structures), as well as various social and economic factors.

Dry cover systems are an alternative where flooding is not possible of feasible. Multi-layer cover systems utilize the capillary barrier concept to keep one (or more) of its layers near saturation under all climatic conditions. This creates a "blanket" of water, or a "water cover", over the reactive waste material. The relatively high degree of saturation ensures that the effective coefficient of oxygen diffusion is low (Figure 4.4-4), which will lead to a control on the influx of atmospheric oxygen to the underlying waste material.

The influx of atmospheric oxygen to reactive waste material can also be limited by incorporating oxygen consuming materials in dry cover systems. The primary function of these barriers is to reduce the ambient oxygen concentration at the waste material/cover interface by consumption of oxygen. Almost invariably, the cover systems contain organic matter, primarily lignocellulosics such as wood chips, wood wastes, peat, sewage sludge, compost, hay, straw, silage, and paper mill sludge (MEND 2.20.1). Oxidation of an organic layer material will eventually decline as the remaining material becomes more humidified and more resistant to further decomposition (Elliott *et al.* 1997). As a result, the long-term efficiency of oxygen consumption barriers is a key element in the design of cover systems that incorporate such barriers (MEND 2.25.5; Tassé 2000).

Limiting the Influx of Atmospheric Water

The net infiltrative flux through a cover system is also an important consideration in the design of a closure system for a mine waste disposal facility. The objective is to control/limit the quantity of water that flows downward through the cover to the underlying waste material because the infiltrating water ultimately contributes to subsequent production of acidic drainage. The net infiltrative flux is a function of the total precipitation, evaporative flux, transpiration, change in soil moisture storage, and runoff. Each of these factors are in turn influenced by a variety of conditions. For example, runoff and infiltration rates are a function of rainfall intensity, surface topography, vegetation, soil properties, and soil moisture conditions. Evaporation and/or evapotranspiration from the cover surface is a strongly coupled process that depends on atmospheric conditions, soil/waste properties, and soil/waste conditions. In addition, it is clear that runoff, as well as run-on, are site-specific considerations.

It is difficult and usually not economically feasible in arid and semi-arid climates to construct a cover system, which contains a layer that remains highly saturated, thereby reducing the influx of atmospheric oxygen. The cover system will be subjected to extended dry periods and therefore, the effect of evapotranspiration will be significant. However, subjecting the cover system to evaporative demands can be beneficial in arid climates and result in a reduction of infiltration to the underlying waste material. A cover surface layer possessing sufficient storage capacity can be used to retain water during a precipitation event or freshet. Subsequent to the increase in moisture storage in this layer, it would release a significant portion of pore water to the atmosphere by evapotranspiration during extended dry periods, thereby reducing the net infiltration across the soil cover system. The objective is to control acidic drainage as a result of controlling the transport mechanism (i.e. water) into the waste material. A cover system with the above objectives is often referred to as a “moisture store and release” type cover system.

An issue that arises with respect to a dry cover system designed to only limit net moisture percolation to the underlying waste is the question of decreasing seepage only, leading to higher concentrations and ultimately the same loading to the environment. In general, there is not complete agreement as to whether very low net percolation rates will lead to the same loading or a reduced loading. It is argued that the low percolation rates associated with a properly designed store and release cover system (with no oxygen control) will eventually lead to contaminant release. Conversely, it can be argued that there must be a reduction in loading for a percolation rate given that: “zero flow corresponds to zero loading release”. In addition, at lower percolation rates the leachable areas of waste rock, for example, will be greatly reduced (albeit the finer textured material will contain higher concentrations of leachable contaminants). Finally, even a store and release dry cover system will provide protection from mechanical weathering and breakdown of waste rock (i.e. freeze/thaw and wet/dry cycles), thus ensuring that the source of contaminant loading does not rapidly increase following decommissioning.

Classification of Dry Cover Systems

The design objectives for dry cover systems form the basis for classification of dry cover systems (Table 4.4-2). MEND 2.21.3a classified cover systems that control acid generation as: oxygen transport barriers; oxygen consuming barriers; reaction inhibiting barriers; and store and release infiltration barriers. Each of these dry cover system types has been discussed, with the exception of reaction inhibiting barriers.

The function of dry cover systems with a reaction-inhibiting barrier is to provide an environment that results in a significant reduction of the intrinsic sulphide oxidation rates (MEND 2.20.1). Materials such as flyash and limestone can be incorporated into the cover system to provide alkalinity. This results in an increase in the pH of the waste material pore water, which in turn reduces the rate of sulphide oxidation.
Further discussion and examples of the various types of dry cover systems are included in Sections 4.4.2 and 4.4.4.

Table 4.4-2
Summary of Dry Cover System Classifications

Dry Cover Classification	Primary Role of Cover in Inhibition of AMD
Oxygen transport barriers	Act to retain moisture and hence provides a low diffusion barrier to atmospheric oxygen
Oxygen consuming barriers	Act as an oxygen consuming sink to provide low oxygen concentrations at the interface
Reaction inhibiting barriers	Act to inhibit reactions, neutralizes pH
Store and release infiltration barriers	Act to minimize moisture flux by maximizing near surface storage of moisture with subsequent release by evapotranspiration

Source: MEND 2.21.3a

4.4.1.5 Factors Influencing Design Objectives

Several factors influence and often dictate the design objectives of a dry cover system. The key factors are likely: climate conditions; waste material reactivity; type of waste material (i.e. tailings or waste rock); hydrogeologic setting; and basal inflow conditions. These factors are discussed below.

Climate Conditions

The climate conditions at the mine site are a key factor in determining the dry cover system objectives. Is it generally a “dry” site (potential evaporation greatly exceeds annual precipitation), or is it generally a “wet” site (annual precipitation meets or exceeds annual potential evaporation)? However, caution is required when using these “annual” criteria for characterizing wet climates at a site. Numerous sites exist in Canada where precipitation exceeds potential evaporation on an annual basis; however, the site typically experiences hot, dry summer months where evaporation greatly exceeds rainfall. These dry summer conditions can make it difficult to design a cover system that meets all objectives throughout the year.

Reactivity of the Waste

The reactivity of the waste material is an important factor in determining the cover system objectives. For example, in the case of inert or non-reactive tailings the only contaminants in tailings seepage are those originally introduced to the impoundment with the process water.

These contaminants can include elevated concentrations of copper or cyanide for example, and all major ions, resulting in a very high ionic strength tailings pore water. During the operating stage the contaminant concentrations in tailings seepage will remain relatively constant, reflecting process water chemistry (assuming the mill circuit is not changed). The tailings seepage will still be dominated by process water for most of the inevitable drain-down period following decommissioning. Only towards the end of this period will it gradually be diluted with "fresh" recharge water. In the long term all contaminants introduced with the process water will be flushed from the tailings. A "higher" quality cover will delay the flushing but will also be associated with higher contaminant concentrations or loadings. This dry cover section generally addresses reactive waste; however, the above discussion is offered to provide a perspective with respect to non-reactive waste.

Reactive waste will usually indicate that a "higher" quality cover system is required to control further reactions and provide a comparatively finite source term for seepage from the covered waste material. For example, if the tailings discussed above are reactive they may oxidize, thereby releasing additional contaminants (i.e. oxidation products such as sulphate and metals) into the tailings pore water. Significant oxidation of the tailings typically does not start until well into the drain-down period when the tailings are dewatered allowing air to move into the tailings profile. The degree of contamination resulting from tailings oxidation (both with respect to make-up of contaminants and contaminant concentrations) depends on the buffering capacity of the system (i.e. buffering capacity of the tailings as well as resident process water). The tailings pore water will become acidic quickly if there is limited or no buffering capacity, resulting in accelerated oxidation and a large increase in sulphate and dissolved metals. In contrast, if there is significant alkalinity (buffering potential) in the tailings, the acidity released during sulphide oxidation will be neutralized, maintaining a circum-neutral pH. The rate of oxidation is limited at this pH and many metals of concern are immobile; however, some contaminants may stay in solution at elevated concentrations albeit at concentrations much lower than in the case of no buffering. Clearly then, the level of reactivity and buffering capacity of the waste will determine the design objectives of the cover system. In addition, the solubility of some metals is higher at circum-neutral pH, as compared to acidic conditions.

Type of Waste

Assuming the waste material is sulphidic and reactive (with or without buffering capacity), whether the material is tailings or waste rock will also determine the cover system objectives. The texture of the waste material has an impact on determining the cover system objectives; a tailings material will typically be poorly drained, higher in moisture content, and finer textured, while waste rock is typically drained, coarse textured and with comparatively low moisture contents. These opposing conditions will determine the cover system objectives from a construction perspective. Moist fine tailings can be extremely difficult for cover placement, while the integrity of the waste rock surface is typically not an issue except when placing a synthetic cover material (i.e. geomembrane). The differing texture between waste rock and tailings also influences cover system objectives due to the predominant mechanisms of oxygen transport. Oxygen diffusion is the predominant transport mechanism for tailings. Oxygen transport in waste rock piles can be dominated by diffusion, but advective and convective oxygen transport can be a significant, if not the overriding component in many situations. Finally, waste rock piles are typically associated with steeper and longer side slopes as compared to tailings impoundments, which are contained by dams.

Hydrogeologic Setting and Basal Flow

The hydrogeologic setting of a waste disposal facility will significantly control the dry cover system objectives. For example, many sites are characterized by a tailings basin located within a groundwater system with significant lateral groundwater transport (i.e. the tailings water table is controlled by the groundwater system and not incident precipitation falling on the tailings surface). In these cases, leaching of process water and remnant acidity will take place regardless of the ability of the cover system to control water infiltration. The higher quality cover often does not have the cost benefit to make it an economic solution; consequently a lower quality cover coupled with a collection and treatment system becomes a viable closure option.

Waste rock is often end-dumped from valley walls and slopes. These walls and slopes will invariably have historic surface water paths. The waste rock placed over the paths will be subjected to seasonal flushing resulting from the water table rising past the waste rock-valley wall interface into the waste rock. Alternatively, waste rock placed on fractured bedrock and faults where groundwater flow is focused can also be subject to seasonal flushing of stored acidity and metals. This long-term seasonal flushing of the waste rock will influence the dry cover system objectives.

4.4.1.6 Design Philosophy for Dry Cover Systems

Two design philosophies exist for dry cover systems; that of designing an engineered structure, the other of designing an engineered system (MEND 2.21.3a; Figure 4.4-5). The primary design goal of an engineered structure is to isolate waste rock and tailings from the surrounding environment. The design of an engineered structure is based primarily on the long-term average steady flux conditions. The advantage of this approach is design simplicity; however, a major disadvantage is difficulty in extrapolating the design to new conditions, such as climate, waste volumes or geometry, without full-scale prototype testing. The design of an engineered structure is often conservative and as a result, may not provide an economically viable design.

The goal of engineered systems is the integration of the waste rock or tailings into the long term, natural evolution of the site. The design of an engineered system is a dynamic soil/waste system coupled with the site-specific environment, which includes climate and vegetation. The main disadvantage with engineered systems is the high degree of theoretical and analytical complexity required in the design phase. The interactions between the waste material and the environment must be analyzed in a dynamic and fully coupled manner. Although design complexity is greater than for an engineered structure, the design of engineered systems allow freedom of design to incorporate local soil materials and native vegetation and to develop economically viable systems.

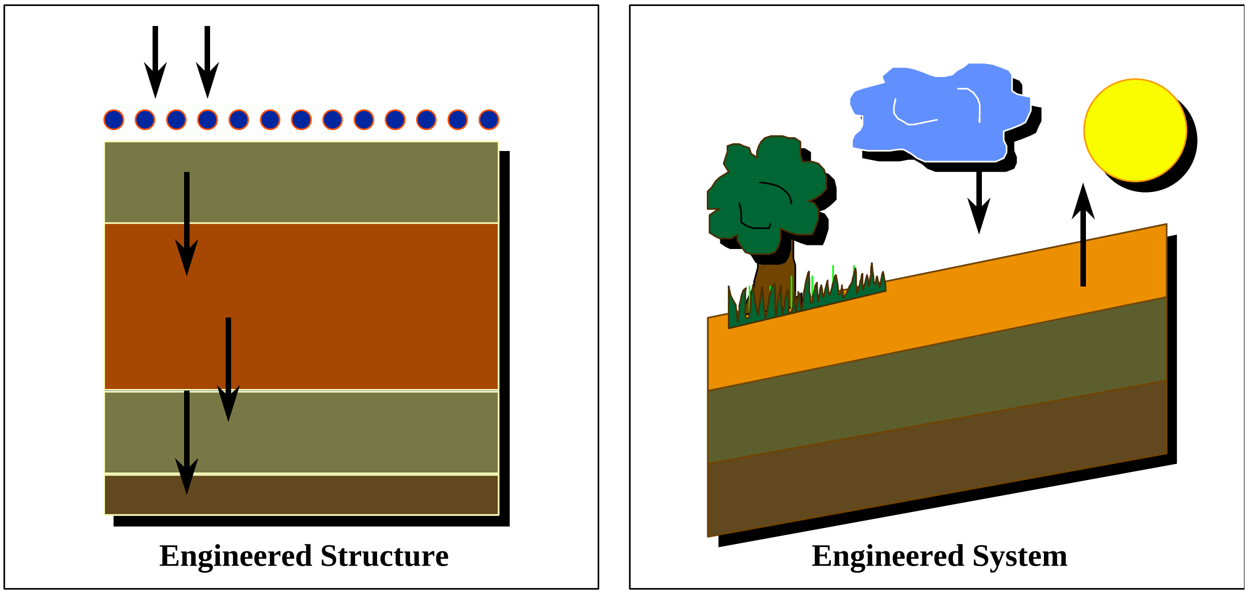
4.4.2 ALTERNATIVE DRY COVER SYSTEM DESIGNS

Figure 4.4-6 shows the base method cover system design and variations of the base method cover system design. It should be noted that there are numerous combinations of the variations presented in Figure 4.4-6, but in the interest of clarity only the basic variations are shown. The simplest case (i.e. the base method cover system design) is typically evaluated first during the preliminary cover design phase, and then complexity added until the desired design objectives are met. In general, increasing complexity in the design of a cover system implies increased cover system performance, which also entails increased costs.

Factors that control the economic and technical feasibility of a cover system for a particular site include, but are certainly not limited to:

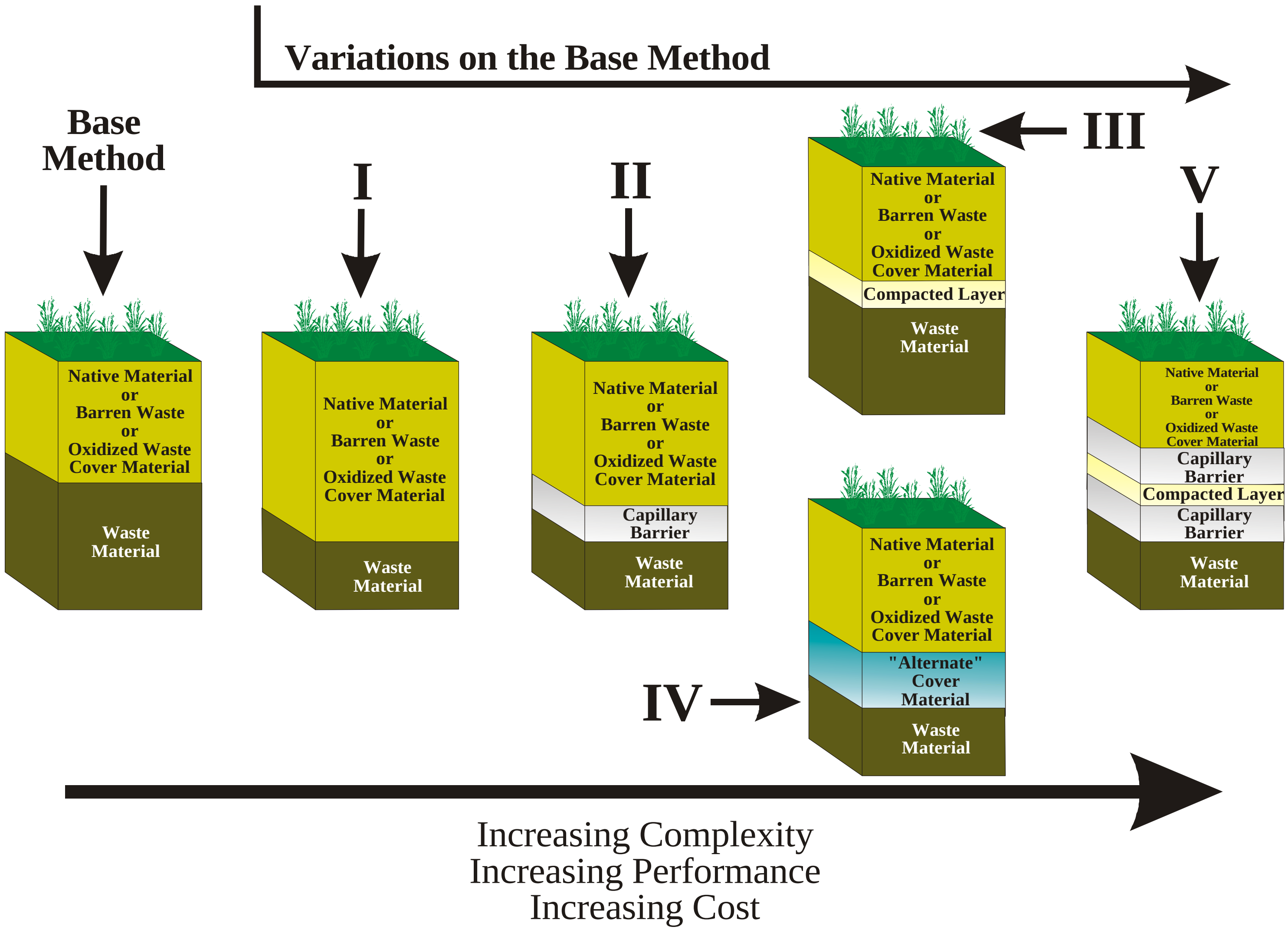
- Site climate conditions;
- Availability of cover material(s) and distance to borrow source(s);
- Cover and waste material properties and conditions;
- Surface topography;
- Soil and waste material evolution; and
- Vegetation conditions.

Figure 4.4-5 **Design Philosophies of Dry Cover Systems: Engineered Structure Versus Engineered System**



Source: MEND 2.21.3a

Figure 4.4-6 **Schematic Illustration of the Base Method Cover System and Variations of the Base Method**



Source: Adapted from Swanson *et al.* 1999

MEND 2.20.1 evaluated alternate dry cover systems for the inhibition of AMD from tailings and in the process, identified several possible sources of cover materials. "Barren waste" in the subsequent discussion of alternate cover system designs is considered to be non-reactive waste rock or tailings. This material is not considered as "special cover materials", but rather a logical cover material source available to most, if not all, mine sites. "Oxidized waste cover material" isconsidered to be near surface waste rock that at one time contained sulphide minerals, but now is free of sulphides and oxidation products through natural weathering over geologic time.

The base method and variations of the base method are discussed below and shown in Figure 4.4-6.

1. <u>Base Method</u> – Non-compacted cover material placed directly on the waste material. This material is usually native material, barren waste material, or oxidized waste material. The primary objective of this cover system is establishment of a sustainable vegetation cover, although reduction of net percolation could also be viewed as a prime objective. This design is most commonly used to cover waste materials that are non-reactive.

2. <u>Base Method Variation I</u> – Non-compacted cover material placed directly on the waste material, but with an increase in the thickness of the non-compacted material. This cover system is an attempt to increase the ability of the dry cover to reduce net percolation of moisture to the underlying waste by increasing the available moisture storage capacity (i.e. store and release of moisture). Establishing a sustainable vegetated cover is also an objective.

3. <u>Base Method Variation II</u> – A capillary barrier material is placed directly on the underlying waste and overlain by non-compacted material. The primary objective of including the capillary barrier material is to provide a hydraulic discontinuity between the underlying waste and the overlying non-compacted cover material. The capillary barrier material is added if potential exists for capillary rise of contaminants (i.e. process water and/or oxidation products) to impact the sustainability of vegetation or the quality of surface water. In addition, the capillary barrier will provide the hydraulic discontinuity to limit percolation to the underlying waste for low to moderately high infiltration conditions.

4. <u>Base Method Variation III</u> – A compacted layer is placed directly on the underlying waste and overlain by the non-compacted material. The objective is to provide a hydraulic barrier to percolation of water due to the low permeability of the compacted material. In addition, the compacted material will typically have the ability to retain moisture under significant drainage and evaporative conditions. Hence, a barrier to oxygen ingress can be achieved by the presence of the compacted material because of the high saturation levels maintained in the compacted layer. The compacted material can consist of native material, run-of-mine waste material (barren or oxidized), or tailings (inert or de-sulphurized). The upper layer of non-compacted material stores and releases moisture as well as provides a medium for vegetation development.

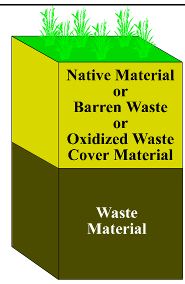
5. Base Method Variation IV – An alternate cover material is placed on the underlying waste and overlain by non-compacted material. The alternate cover material can be an organic layer of material such as municipal solid waste compost, wood waste, peat, etc. although typically these organic materials would not be overlain by a non-compacted layer. These materials have the potential to provide physical barriers to oxygen ingress (i.e. high saturation layers) but also to consume atmospheric oxygen through decomposition. These materials also have the potential to function as vegetation growth mediums, thereby limiting the requirement for the non-compacted native material. The alternate material can also consist of synthetic materials such as cementitious material, shotcrete, flyash mixtures, geopolymers, flexible membrane liners such as geomembranes and geosynthetic clay liners, and ameliorated cover material barriers where bentonite (with or without polymers) or flyash are added to enhance performance of the compacted cover material.

6. Base Method Variation V – This variation of the base method cover system design includes a capillary barrier material placed directly on the waste material. The capillary barrier material is usually a uniform material (i.e. like beach sand) that is coarser in texture than the compacted material. A compacted layer and an upper capillary barrier material then overlie the lower capillary barrier material. The three layers are typically overlain by a vegetation growth medium. It can be feasible for the upper capillary barrier material to act as a growth medium if the compacted material has significant fines and moisture retention, thus allowing the overlying capillary barrier material to be finer textured and function as a zone for root development.

Figures 4.4-7 to 4.4-12 inclusive show the base method cover system design and variations of the base method, together with the objectives of each dry cover system design and references to MEND and other case studies. The location of the case study, climate, and waste material type are also given in each figure. The case studies are then discussed in Section 4.4.4. The case studies presented in addition to the MEND projects were limited to those with readily available information from conference proceedings and refereed journals to allow the reader to obtain detailed information with a minimal effort. A bibliography of MEND papers, with reference to the MEND project numbers, is included as Appendix C. Also, it should be noted that not all of the "non-MEND" case studies presented in Figures 4.4-7 through 4.4-12 are discussed in Section 4.4.4.

Reference is made to a "predictive numerical model" (MEND 1.25.1) at the beginning of Figures 4.4-7 to 4.4-12 inclusive. The model being referred to is SoilCover, which is a one-dimensional, coupled heat and mass transfer program that uses a physically based method for predicting the exchange of heat and moisture between the atmosphere and a soil surface (Wilson *et al.* 1994). The most recent version of this model (SoilCover 2000, Version 5) is available on the World Wide Web (www.vadose-science.com). MEND 2.21.3a and MEND 2.21.4 describe a methodology for numerical modelling and list other software packages that may be used in the design of dry cover systems.

Figure 4.4-7 **Summary of Base Method Dry Cover System Design Case Studies**



Potential Primary Design Objectives

1. Establish vegetation
2. Dust suppression
3. Control radon exfiltration
4. Consolidation (tailings)
5. Control water infiltration to underlying waste
6. Control waste material surface runoff water quality

MEND Project or Other Case Studies	**Site or Description**	**Climate**	**Waste Material (Design Objectives)**
• MEND 1.25.1	Predictive numerical model		Tailings and waste rock
• MEND 2.24.1	Revegetation manual		Tailings (1)
• MEND 5.8.1	Economic evaluation tools		Tailings and waste rock (1, 6)
• Gardiner *et al.* 1997 • O’Kane *et al.* 1999b	Kimberley, B.C.	Arid-summer, wet-winter	Tailings (1, 2, 5, 6)
• Wels 2000	Questa, NM, USA	Arid	Tailings (1, 2, 5, 6)
• Wels *et al.* 1999	Wismut, Germany (also includes numerous other references)	Wet	Tailings (1, 2, 3, 4, 5, 6)
• Gobla 1999	Summitville, Colorado, USA	Dry and wet	Waste rock (1, 5, 6)
• Renken 1998	Literature review and study		Tailings (1)
• Woyshner and Swarbrick 1997	Kidd Creek, Timmins, Ontario	Wet	Tailings (5)
• Berdusco and O’Brien 1999	Elkford, B.C.	Semi-arid (reclamation high elevation)	Waste rock (1)
• Wimberley 1999	South Africa	Arid	Waste rock (1, 5, 6)
• Mulligan *et al.* 1999	Kidston, Queensland, Australia	Semi-arid tropical (reclamation)	Tailings (1)
• Cellan 1998 • Burnside *et al.* 1999	Hawthorne, Nevada, USA	Arid	Heap leach (1)

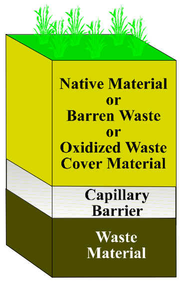
Figure 4.4-8 **Summary of Variation on Base Method Dry Cover System Design Case Studies – Increase Thickness of Non-Compacted Material**

Potential Primary Design Objectives

1. Establish vegetation
2. Dust suppression
3. Control radon exfiltration
4. Consolidation (tailings)
5. Control water infiltration to underlying waste
6. Control waste material surface runoff water quality

MEND Project or Other Case Studies	**Site or Description**	**Climate**	**Waste Material (Design Objectives)**
• MEND 1.25.1	Predictive numerical model		Tailings and waste rock
• Bews *et al.* 1997 • Durham *et al.* 1999 • Durham 1999	Kidston, Queensland, Australia	Semi-arid tropical	Waste rock (1, 5, 6)
• Wimberley 1999	South Africa	Arid	Waste rock (1, 5, 6)
• Rohde 1992	Helena, Montana, USA	Semi-arid	Waste rock (1, 5)
• Swanson 1995 • Wilson *et al.* 1995 • Filipek *et al.* 1996	Montana, USA	Semi-arid	Waste rock (1, 5, 6)
• O'Kane *et al.* 1998a • O'Kane *et al.* 1999a • Watkins *et al.* 2000 • O'Kane *et al.* 2000	Newman, Western Australia	Arid tropical	Waste rock (1, 5, 6)
• Zhan *et al.* 2001a • Zhan *et al.* 2001b	Nevada, USA	Semi-arid	Spent heap leach (1, 5)
• Marcus 1997	Port Hardy, B.C.	Wet	Waste rock (1, 5)

Figure 4.4-9 **Summary of Variation on the Base Method Dry Cover System Design Case Studies – Addition of Capillary Barrier Material**

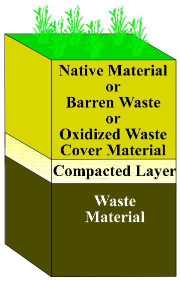


Potential Primary Design Objectives

1. Establish vegetation
2. Dust suppression
3. Control radon exfiltration
4. Consolidation (tailings)
5. Control water infiltration to underlying waste
6. Control waste material surface runoff water quality

MEND Project or Other Case Studies	**Site or Description**	**Climate**	**Waste Material (Design Objectives)**
• MEND 1.25.1	Predictive numerical model		Tailings and waste rock
• Gardiner *et al.* 1997 • O'Kane *et al.* 1999b	Kimberley, B.C.	Arid-summer, wet-winter	Tailings (1, 2, 5, 6)
• Rohde 1992	Helena, Montana, USA	Semi-arid	Waste rock (1, 5)

Figure 4.4-10 **Summary of Variation on Base Method Dry Cover System Design Case Studies – Addition of Compacted Layer**



Potential Primary Design Objectives

1. Control ingress of oxygen by advection, convection, diffusion (waste rock), diffusion (tailings)
2. Control water infiltration to underlying waste
3. Establish vegetation
4. Control radon exfiltration
5. Control upward movement of process water (tailings) and oxidation products

MEND Project or Other Case Studies	**Site or Description**	**Climate**	**Waste Material (Design Objectives)**
• MEND 1.25.1	Predictive numerical model		Tailings and waste rock
• MEND 2.20.1	Literature review		Tailings and waste rock (1, 2, 3, 4 organic, inorganic)
• MEND 2.35.2b	Laboratory study		Waste rock (1, 2, 3, 4, 5 organic, inorganic)
• Gardiner *et al.* 1997 • O'Kane *et al.* 1999b	Kimberley, B.C.	Arid-summer, wet-winter	Tailings (1, 2, 3, 5)
• Firth and van der Linden 1997	Indonesia	Wet tropical	Waste rock (1, 2, 3)
• Harries and Ritchie 1981; 1985 and 1987 • Bennett *et al.* 1988 • Applegate and Kraatz 1991 • Timms and Bennett 2000	Rum Jungle, Northern Territory, Australia	Wet tropical	Waste rock (1, 2, 3)
• Bews *et al.* 1997 • Durham *et al.* 1999 • Durham 1999	Kidston, Queensland, Australia	Semi-arid tropical	Waste rock (2, 3)
• Gallinger 1988 • Swanson 1995 • O'Kane 1996 • Wilson *et al.* 1995 • Owuputi *et al.* 1995 • Aziz and Ferguson 1997 • Wilson *et al.* 1997 • O'Kane *et al.* 1998b • Saretzky 1998 • Saretzky and Wilson 2000	Equity Silver, Houston, B.C.	Wet	Waste rock (1, 2, 3)
• Gossow 1985	Hesse, Germany	Wet	Nuclear waste (2, 3, 4)
• Lindvall *et al.* 1997	Aitik, Sweden	Wet	Waste rock (1, 2, 3)
• Murray *et al.* 1988	Halifax Airport, Nova Scotia	Wet	Waste rock (1, 2)
• EPA 1996 • Duex 2000	Richmond Hill, South Dakota, USA	Wet	Waste rock (2, 3)
• Orr 1995	Mt. Leyshon, Queensland, Australia	Wet	Waste rock (1, 2, 3)
• Lundgren 1997	Bersbo, Sweden	Wet	Waste rock (2, 3)
• O'Kane *et al.* 1998c	Myra Falls, Campbell River, B.C.	Wet	Waste rock (1, 2, 3)

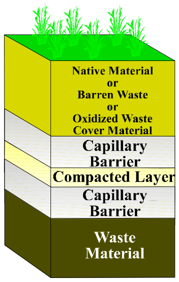
Figure 4.4-11 **Summary of Variation on Base Method Dry Cover System Design Case Studies – Addition of "Alternate" Cover Material**

Potential Primary Design Objectives

1. Control ingress of oxygen by advection, convection, diffusion (waste rock), diffusion (tailings)
2. Atmospheric oxygen consumption (inorganic and organic)
3. Control water infiltration to underlying waste
4. Establish vegetation
5. Control radon exfiltration
6. Control upward movement of process water (tailings) and oxidation products

MEND Project or Other Case Studies	Site or Description	Climate	Waste Material (Design Objectives)
• MEND 1.25.1	Predictive numerical model		Tailings and waste rock
• MEND 2.20.1	Literature review		Tailings and waste rock (organic, inorganic)
• MEND 2.25.1b	Literature review		Tailings (2, 3 organic)
• MEND 2.25.1b	Laboratory study		Tailings (2, 3 organic)
• MEND 2.25.3	Laboratory study		Tailings (1, 2, 3 organic and inoranic)
• MEND 2.34.1	Myra Falls, Campbell River, B.C.		Waste rock (3 inorganic)
• MEND 2.35.2b	Laboratory study		Waste rock (2 inorganic)
• MEND 5.8.1	Economic evaluation tools		Tailings and waste rock (3 inorganic)
• MEND 6.2	Laboratory study		Tailings (1, 2, 3 inorganic)
• MEND 2.21.3b (unpublished)	Literature review		Tailings and waste rock (3 inorganic)
• MEND 2.25.2 • Tremblay 1994 • Tassé *et al.* 1997 • Tassé 1999 • Tassé, 2000	East Sullivan, Val d'Or, Québec	Wet	Tailings (1, 2, 3 inorganic)
• Lewis *et al.* 2000	Mine Poirier, Joutel, Québec	Wet	Tailings (1, 3, 4 synthetic)
• Ketallapper and Chrisianson 1999	Summitville, Colorado, USA	Wet	Waste rock (1, 3, 4 synthetic)
• Lundgren and Lindahl 1991	Ranstad, Sweden	Wet	Heap leach (1, 2, 3, 4 bentonite)
• Haug *et al.* 1991	Saskatoon, Saskatchewan	Semi-arid	Tailings (3 bentonite)
• Alvarez and Ridolfi 1999	Kellogg, Idaho, USA	Wet	Waste rock (2, 3 GCL)
• O'Kane *et al.* 1998c	Myra Falls, Campbell River, B.C.	Wet	Waste rock (1, 2, 3 bentonite/flyash)

Figure 4.4-12 **Summary of Variation on Base Method Dry Cover System Design Case Studies – Addition of Capillary Barrier Material Above and Below the Compacted Layer**



Potential Primary Design Objectives

1. Control ingress of oxygen by advection, convection, diffusion (waste rock), diffusion (tailings)
2. Atmospheric oxygen consumption (inorganic and organic)
3. Control water infiltration to underlying waste
4. Establish vegetation
5. Control radon exfiltration
6. Control upward movement of process water (tailings) and oxidation products

MEND Project or Other Case Studies	Site or Description	Climate	Waste Material (Design Objectives)
• MEND 1.25.1	Predictive numerical model		Tailings and waste rock
• MEND 2.21.1	Laboratory study		Tailings (1, 3)
• MEND 2.21.2 • MEND 2.21.3a • Yanful and St-Arnaud 1992	Waite Amulet, Québec	Wet	Tailings (1, 3)
• MEND 2.22.2a • MEND 2.22.2b • Bussière *et al.* 1997 • Benzaazoua *et al.* 1998	Laboratory study		Tailings (1, 3)
• MEND 2.22.2c • Aubertin *et al.* 1997	Val d'Or, Québec (six experimental cells)	Wet	Tailings (1, 3)
• MEND 2.22.3 (unpublished)	Economic evaluation		Tailings (1, 3)
• MEND 2.22.4a • MEND 2.22.4b • Ricard *et al.* 1997 • Ricard *et al.* 1999	Les Terrains Auriféres, Québec	Wet	Tailings (1, 3)
• MEND 2.31.1a • MEND 2.31.1b • MEND 2.31.1c • MEND 2.21.3a • Yanful *et al.* 1993b 1993c	Heath Steele, New Brunswick	Wet	Waste rock (1, 3)
• MEND 2.35.2b	Laboratory study		Waste rock (1, 3)
• Gardiner *et al.* 1997 • O'Kane *et al.* 1999b	Kimberley, B.C.	Arid-summer, wet-winter	Tailings (1, 3, 4, 6)
• Hanselman and Courtin 1999	Elliot Lake, Ontario	Wet	Tailings (1, 2, 3, 4, 6 inorganic)
• Cabral *et al.* 1997	Sherbrooke, Québec	Wet	Tailings (1, 2, 3, 4, 6 organic)
• Cabral *et al.* 1999	Woburn, Québec	Wet	Waste rock (1, 2, 3, 4, 6 organic)
• Simms and Yanful 1999	London, Ontario	Wet	Tailings (1, 3, 4)
• Alvarez and Ridolfi 1999	Kellogg, Idaho, USA	Wet	Waste rock (3, 4)
• Woyshner *et al.* 1997	Rouyn-Noranda, Québec	Wet	Waste rock (1, 3, 4, 6)

4.4.2.1 Effect of Sloped Surfaces on Cover Performance

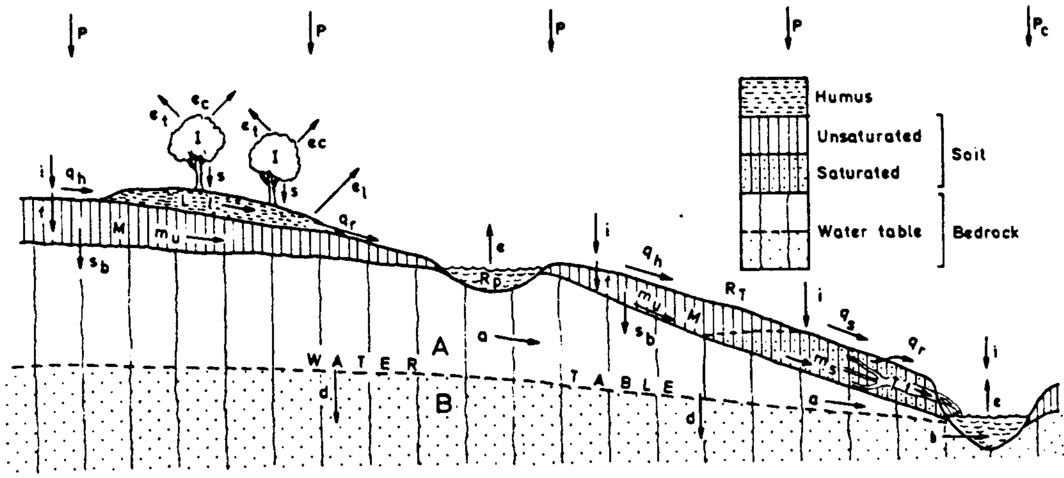
Considerable fundamental and applied research has been devoted to investigate the performance of soil cover systems for mine waste rock and tailings. In general, the literature illustrates that the majority of soil cover designs deal with vertical flow of heat and moisture in horizontal layers, using numerical models as well as laboratory and/or field scale physical models. Field performance monitoring has typically focused on one-dimensional performance. In reality, mine waste rock piles have steep slopes as a result of construction methods and placement configurations. The hydraulic performance of a soil cover system placed on these slopes, and its ability to function as oxygen ingress and water infiltration control systems, will be different than that predicted by idealized one-dimensional numerical models (Boldt-Leppin *et al.* 1999).

The topographically controlled groundwater flow system developed in fine-grained soils in waste piles is analogous to natural groundwater flow on hill slopes. The distribution of the hydraulic heads is controlled by the topography, raising the water table to conform to the topography. The flow system is mainly influenced by low hydraulic gradients controlled by the slope, low permeability, high capillary forces due to the fine-grained soils, and spatial homogeneity.

Figure 4.4-13 is a schematic illustration of the components of a hill slope hydrologic cycle. It is clear from Figure 4.4-13 that side slopes of waste rock piles are analogues to natural hill slopes. A significant amount of research that could be applied to waste rock cover systems has been completed studying the hydrology of natural hill slopes (Boldt-Leppin *et al.* 1999).

MEND 2.22.4b and Aubertin *et al.* (1997b) completed two-dimensional numerical modelling of unsaturated inclined layers (silt layer between an upper and lower sand layer). The modelling was conducted to assess the degree of saturation in a sloped cover system designed to prevent gas transport across the cover. The effects of a 4% slope on de-saturation of the cover layers after 60 days of drought were demonstrated. The computed saturation profile in the two sand layers was similar to the one-dimensional simulation; however, the profile in the fine-grained silt layer showed a marked de-saturation, particularly in the upper part of the layer. The boundary conditions in the simulation together with the hydrological regime created conditions where the moisture retention in the silt layer was not sufficient to maintain saturation near the top of the slope. The result illustrates that for the materials used in this example, a 4% slope is sufficient to de-saturate the top portion of a relatively short slope length. The numerical model calculations are based on an actual case of a cover system constructed in northwestern Québec (MEND 2.22.4b). This field investigation shows that the water content profile is greatly affected by the position along the slope. The performance observed with the placement of hydraulic barrier test cells indicates that this type of structure would be effective when supplementary intervention is needed in the slopes. MEND 2.22.4b and Bussière *et al.* (1998; 2000b) provide further details on the effects of side slopes on the efficiency of capillary barriers to control AMD.

Figure 4.4-13 **Components of the Hill Slope Hydrologic Cycle**



Precipitation (gross rainfall)	P	Horton overland flow	q_h
Channel precipitation	P_c	Saturated overland flow	q_s
Precipitation intensity	i	Return flow	q_r
Evapotranspiration	e_t	Pipe flow	t
Canopy interception loss	e_c	Pipe storage	T
Interception and canopy storage	I	Unsaturated throughflow	m_u
Stemflow and drip	s	Saturated throughflow	m_s
Litter flow	l	Soil-moisture storage	M
Litter interception loss	e_l	Seepage into bedrock	s_b
Litter storage	L	Interflow in bedrock	a
Evaporation	e	Aeration zone storage	A
Depression storage	R_P	Deep seepage	d
Detention storage	R_T	Baseflow	b
Infiltration	f	Groundwater storage	B

Source: Chorley 1978

In general, the performance of a capillary barrier cover system on a slope will be a function of climate conditions, slope angle and length, as well as cover material and waste material properties. However, a capillary barrier cover system that functions properly at one site may not function properly at another due to a change in one of the factors listed in Section 4.4.1.5. Particular caution is required when transferring a cover system designed for a sloped surface from one site to the next.

4.4.3 DRY COVER SYSTEM PERFORMANCE MONITORING

One of the most common technologies used for the evaluation of dry cover system performance is water quality analyses of seepage discharged from the waste storage facility. Water samples are typically collected from collection ditches around the perimeter of the waste impoundment or from monitoring wells installed below or downgradient of the waste site. This approach empirically describes a waste rock or tailings storage facility through monitoring of its cumulative effect at the base (Morin and Hutt 1994). The success of revegetation on the surface of waste rock dumps and tailings facilities can also be an indicator of pyrite oxidation. Harries and Ritchie (1981; 1985; 1987), Yanful *et al.* (1993c), and O'Kane *et al.* (1998b) illustrated that gaseous oxygen profiles and temperature profiles can also serve as a tool for the evaluation of cover system performance because the profiles indicate the internal behavior of the dump.

Direct measurement of field performance is the state-of-the-art methodology for measuring performance of a dry cover system. Field performance monitoring can be implemented during the design stage with test cover plots or following construction of the full-scale cover (e.g. MEND 2.21.2; Woyshner and Swarbrick 1997; Aubertin *et al.* 1997a; O'Kane *et al.* 1998a, 1998b, 1998c, 1999b; MEND 2.22.4a,b; MEND 2.31.1c). Direct measurement of field performance of a dry cover system is the best method for demonstrating to regulatory agencies and the public that the cover system will perform as designed. The main objectives of field performance monitoring are to:

- Obtain a water balance for the site;
- Obtain an accurate set of field data to calibrate a numerical model;
- Develop confidence with all stakeholders with respect to cover system performance; and
- Develop an understanding for key characteristics and processes that control performance.

The desired field performance monitoring system should include monitoring of the various components that influence the performance of a dry cover system. These components are shown schematically in Figure 4.4-14, and are related as follows:

$$\text{PERC} = \Delta \text{S} + \text{NSI, and} \qquad (1)$$

$$\text{NSI} = \text{PPT} - \text{AET} - \text{RO} \qquad (2)$$

where:

PERC = net percolation into the underlying waste material from the base of the cover;
ΔS = change in moisture storage within the cover layers;
NSI = net surface infiltration;
PPT = precipitation;
AET = actual evapotranspiration; and
RO = runoff.

Table 4.4-3 lists typical methods of measurement for the various components of a field performance monitoring system (MEND 2.21.4). Many of the sensors listed in Table 4.4-3 can be connected to an automated data acquisition system, which typically includes a datalogger, multiplexer, and a solar panel/rechargeable battery power source. The use of automated systems for data collection greatly reduces the need for human intervention and in particular, the requirements of mine site personnel (Ayres *et al.* 1996). Cellular, radio, or telephone communication can simplify data collection requirements at many sites.

MEND 2.21.4 provides a detailed overview of field performance monitoring for dry cover systems. Some key points related to meteorological monitoring, measurement of moisture storage changes and lysimeters are provided below.

Meteorological Monitoring

Measurement of site precipitation is the most crucial of site meteorological measurements. Rainfall during the summer should be measured by several inexpensive tipping bucket rain gauges for large sites to quantify spatial differences. Rain gauges should also be located at all test plot sites. Snowfall should be measured with an all-season precipitation gauge and in addition, regular depth/density measurements of the snowpack should be collected with increasing frequency as spring freshet approaches.
Other meteorological parameters that should be monitored include air temperature, relative humidity, wind speed and direction, and net solar radiation. This data is required as input to the model (e.g. SoilCover) and can be used to evaluate the potential evaporation for the site.

Monitoring of Moisture Storage Changes

In general, water content sensors alone are needed to measure changes in moisture storage within a cover system; however, it is recommended that soil suction sensors be installed at the same location and depths of the water content sensors in each of the cover system layers. This enables a field soil-water characteristic curve (SWCC) to be developed for each material in the cover system (see Section 4.4.1.3 for a discussion on SWCC). A soil suction sensor can be thought of as the “piezometer of the unsaturated zone”. Measurement of soil suction is a fundamental

Figure 4.4-14 **Schematic of Field Performance Monitoring System for Dry Covers Systems**

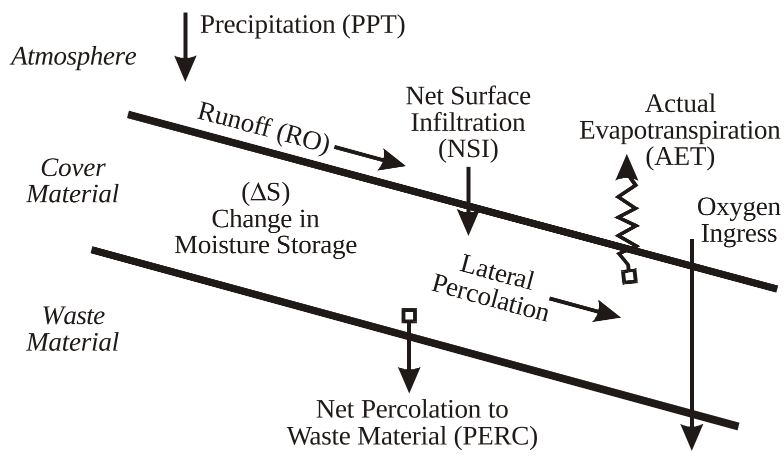


Table 4.4-3
Typical Methods of Measurement for the Components of a Field Performance Monitoring System

Parameter Measured	Typical Method(s) of Measurement
Precipitation	Tipping bucket rain gauge (for rainfall) All-season precipitation gauge (for snowfall) Snow survey (depth and density of snowpack)
Actual evapotranspiration	Bowen ratio energy balance (BREB) Weighing lysimeter
Moisture content	Time domain reflectometry (TDR) Frequency domain reflectometry (FDR) Neutron scattering
Negative pore water pressure (or soil suction)	Thermal conductivity sensor Jet-fill tensiometer
Positive pore water pressure	Standpipe piezometer Pneumatic piezometer Electric piezometer
Net percolation	Lysimeter Suction sensor gradients
Temperature	Thermocouple Thermistor
Gaseous oxygen	Oxygen analyser (with sampling ports) Oxygen consumption test

component of the monitoring program because soil suction describes the stress state of the cover system. In other words, a change in the degree of saturation or water content of the cover system is caused by a change in soil suction within the cover material. Soil suction determines the volume of water present within the soil pores and subsequently the hydraulic conductivity of the porous material, as related by the hydraulic conductivity function.

Field soil suction values also provide a means of verifying field water content measurements. The hydraulic conductivity of the unsaturated cover material can be determined based on the response time of the soil suction sensor. Finally, the hydraulic gradient, or direction of moisture flow, can be evaluated based on soil suction measurements. The hydraulic gradient cannot be determined from water content measurements, particularly in layered systems, which describes most natural and engineered systems.

All methods listed in Table 4.4-3 for monitoring *in situ* water content will require field calibration to obtain quantitative measurements of volumetric water content. The reason is that volumetric water content is a function of texture and density, and the factory calibration curve does not account for site-specific field conditions. It is fundamental to note that all volumetric water content sensors have been designed for the agriculture industry. The materials encountered in the agriculture industry can be much different than those encountered in the mining industry. Nonetheless, qualitative measurements can also be useful and should not be discounted.

<u>Monitoring of Net Percolation</u>

Measurement of the net water flux from the base of the cover layers into the underlying waste material is likely the most important component of a cover system monitoring program. In general, the design and installation of lysimeters to monitor evaporative fluxes as well as net infiltration (PERC in Figure 4.4-14) is well understood and implemented in the soil science discipline. However, the design of lysimeters for dry cover system monitoring programs in the mining industry have typically not included fundamental lysimeter design aspects established in the soil science discipline. As a result, most lysimeter measurements used to monitor dry cover system performance should be used with caution unless a thorough review of the design and installation of the lysimeter is undertaken.

A lysimeter installed to monitor net infiltration from the base of a cover system is part of an unsaturated system. Flow into the lysimeter may occur during saturated conditions; however, flow will primarily occur during unsaturated conditions. The flux through the cover is a function of the properties of not only the cover material, but also the material underlying the cover, which in turn controls the suction at the base of the cover. In addition, the lysimeter establishes an artificial water table boundary condition below the cover that is different than outside the lysimeter. The design of field lysimeters requires that the geometry of the lysimeter (crosssectional area and depth), hydraulic properties of the backfill (SWCC and saturated hydraulic conductivity), and the cover response (flux) be integrated so that the suction at the cover layer/waste material interface is the same both inside and outside of the confines of the field lysimeter.

Bews *et al.* (1997; 1999) and Simms and Yanful (2000) provide further insight into the design and performance of field lysimeters for monitoring net percolation through dry cover systems.

4.4.4 CASE STUDIES

This section presents case studies of dry cover designs and performance monitoring. The case studies are separated into a section addressing tailings and a section addressing waste rock. Each section first focuses on those studies completed in conjunction with MEND, followed by the non-MEND case studies.

4.4.4.1 Tailings Dry Cover System Case Studies

Waite Amulet (MEND 2.21.1, 2.21.2)

The evaluation of covers for the Waite Amulet tailings area consisted of performance monitoring of field test plots at the decommissioned tailings site near Rouyn-Noranda, Québec and laboratory experiments at the Noranda Technology Centre and universities (MEND 2.21.1; 2.21.2).

Four test plots, consisting of two composite soil covers, one geomembrane cover, and a control (tailings without cover) were constructed at the Waite Amulet site. Each test plot was instrumented to measure gaseous oxygen concentrations underlying the cover layers, cover material moisture conditions, *in situ* temperature, and pore water quality at various depths. In addition, a collection basin lysimeter, initially filled with unoxidized tailings, was installed below each cover to measure both the quantity and quality of seepage water.

The composite soil cover consisted of a 60 cm thick, compacted, silty clay layer placed between two sand layers, each 30 cm thick. Different proctor densities and molding water contents were used for the two compacted layers. A final 10 cm gravel crust blanketed the cover system to minimize erosion. Figure 4.4-15 is a cross section through the Waite Amulet tailings test plots.

These cover layer depths were selected to provide maximum reduction in oxygen ingress and a sufficient safety factor to minimize the effects of adverse climatic conditions such as freezing and thawing. The design of the cover was based on the results of a laboratory study that concluded that the composite cover would be able to resist significant moisture losses during dry climate conditions. The geomembrane cover consisted of an 80 mil (2 mm thick) high-density polyethylene (HDPE) liner material placed between a fine sand and underlying coarse sand layers.

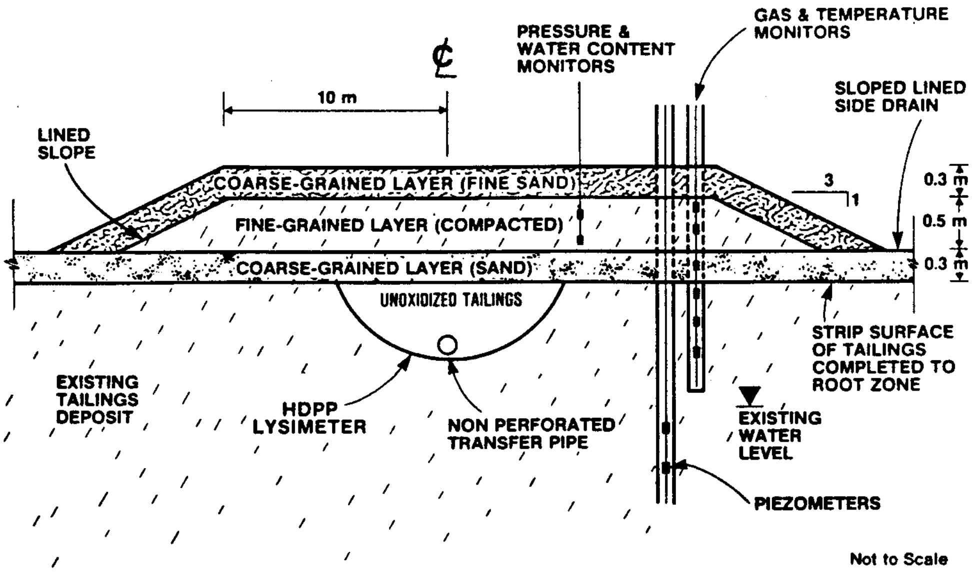
The laboratory study consisted of six column tests to simulate covered and uncovered tailings. The covered tailings consisted of a 30 cm thick clay layer placed between two sand layers, each 15 cm thick. The soils were similar to those used in the construction of the field test plots. Un-oxidized tailings used in the laboratory experiments were collected from the deep saturated zone of the south end section of the Waite Amulet tailings impoundment. The covered and uncovered tailings were subjected to cyclic wetting and drying, at laboratory temperatures. Gaseous oxygen concentration, moisture conditions, temperature, and drainage water quality were monitored during the laboratory study. The covered tailings did not produce any seepage during normal wetting or rain application because of the low hydraulic conductivity of the compacted clay layer. A large majority of the water applied to the surface of the covered tailings during the column test reported as runoff. The covered tailings were periodically flushed (by-passing the soil cover) to collect pore water and assess the degree of sulphide oxidation.

Results of the laboratory, field and modelling studies indicated that ingress of oxygen was reduced by 91 to 99% due to the presence of the cover material as compared to the uncovered tailings. Hydrologic modelling indicated that water percolation through the composite cover was approximately 4% of the incident precipitation. Field lysimeter data indicated that 6% of the annual incident precipitation percolated to the underlying tailings. This represented a reduction of 80% in the total annual infiltration as compared to the uncovered tailings. It was concluded that freezing and thawing did not adversely affect the cover and that no future negative effects would be anticipated, based on field results and results from the laboratory freeze-thaw studies. It was also found that the long-term stability of the HDPE cover was not a major concern except for the possible effects of equipment, burrowing animals and sunlight.

MEND 2.21.3a, which provides a critical, peer review of this project, states that the soil cover system installed at the Waite Amulet site performed satisfactorily. It was also stated that the results of the research indicated that the capillary barrier concept was attainable under field conditions and will result in a reduction of the influx of infiltration and oxygen, thereby reducing the potential for acid generation. One concern raised in the MEND 2.21.3a report was the costs associated with the construction of the Waite Amulet cover system, which was approximately $250,000/ha. However, it is noted that this is for a test plot and not a full-scale cover. Costs would be reduced for a full-scale cover system.

Details on the Waite Amulet project can also be found in Yanful (1991), Yanful and St-Arnaud (1991; 1992), Yanful and Aubé (1993), Woyshner and Yanful (1993), and Yanful *et al.* (1993a).

Figure 4.4-15 **Cross-Section through Composite Soil Cover Test Plot at Waite Amulet Tailings**



Source: Yanful and St-Arnaud 1991

Les Terrains Aurifères (MEND 2.22.4)

Several laboratory and field studies were carried out under MEND prior to the design and construction of a full-scale cover system for the tailings facility at the Les Terrains Aurifères (LTA) site near Malartic, Québec (MEND 2.22.4). The results of the studies conducted prior to the LTA project were instrumental in the final design of the tailings cover system, and are summarized briefly below.

Laboratory studies were completed between 1991 and 1996 at l'École Polytechnique de Montréal on the feasibility of using clean (i.e. non-acid generating) tailings for the moisture retention layer in a multi-layer cover system (MEND 2.22.2a, 2.22.2b). The impetus for this research was to find a lower cost alternative for the fine-grained layer for sites where clean cover material is not available near the site. Clean tailings can be obtained from milling sulphide-free ore, or by using a desulphurization process that has been investigated by MEND 2.22.3, Bussière *et al.* (1997), Benzaazoua *et al.* (1998), and others. The studies have shown that if high saturation (≥ 90%) can be maintained in the cover through capillary barrier effects, then a layer of fine material (i.e. tailings) sandwiched between two sand layers will effectively reduce the oxygen flux to the reactive tailings materials by a factor of 1,000 of more. Theoretically, the efficiency of a dry cover then becomes comparable to that of a water cover of the same thickness (MEND 2.22.2a).

Six experimental cells were constructed in 1995 on a site near Val d'Or, Québec to evaluate, on a larger scale and under more realistic conditions, the performance of multi-layer covers built with clean tailings (MEND 2.22.2c). A laboratory study was conducted in parallel to this field program to characterize the material properties and evaluate, under well controlled conditions, the behavior of the same cover systems constructed in the field. MEND 2.22.2b presents the material properties and the content of the various laboratory columns. Preliminary results from the column testing revealed the various layered cover systems had a similar hydraulic behavior to the layered system evaluated in MEND 2.22.2a.

MEND 2.22.2c and Bussière and Aubertin (1999) describe the experimental cells and the instrumentation installed in each test cover, as well as present and discuss the results of nearly four years of monitoring. Each of the five test covers (one cell was a control plot) had three layers placed on reactive tailings. The top and bottom layers consisted of 0.3 m and 0.4 m of relatively coarse sand, respectively. The fine material layer, placed between the two sand layers, was either made of clean tailings with different thicknesses (three cells: 0.3, 0.6 and 0.9 m), of a natural silty soil (one cell: 0.6 m), or a mixture of bentonite and clean tailings (one cell: 0.3 m). The field performance monitoring program consisted of routine measurements of volumetric water content, matric suction, oxygen flux, and chemical composition of the leachate.

The key monitoring results from the MEND 2.22.2c field project include the *in situ* water content profiles and oxygen fluxes measured through the various test covers. The water content profile showed that the degree of saturation in the fine-grained layer of each test cover remained high (usually above 85%). The oxygen consumption tests demonstrated that after the first year, the oxygen flux through the covers became very low compared to the control (uncovered) cell. The conclusion from all the different monitoring results is that clean tailings can be used as fine-grained material in a multi-layer cover system to control the production of AMD from sulphidic tailings (MEND 2.22.2c; Bussière *et al.* 2000a).

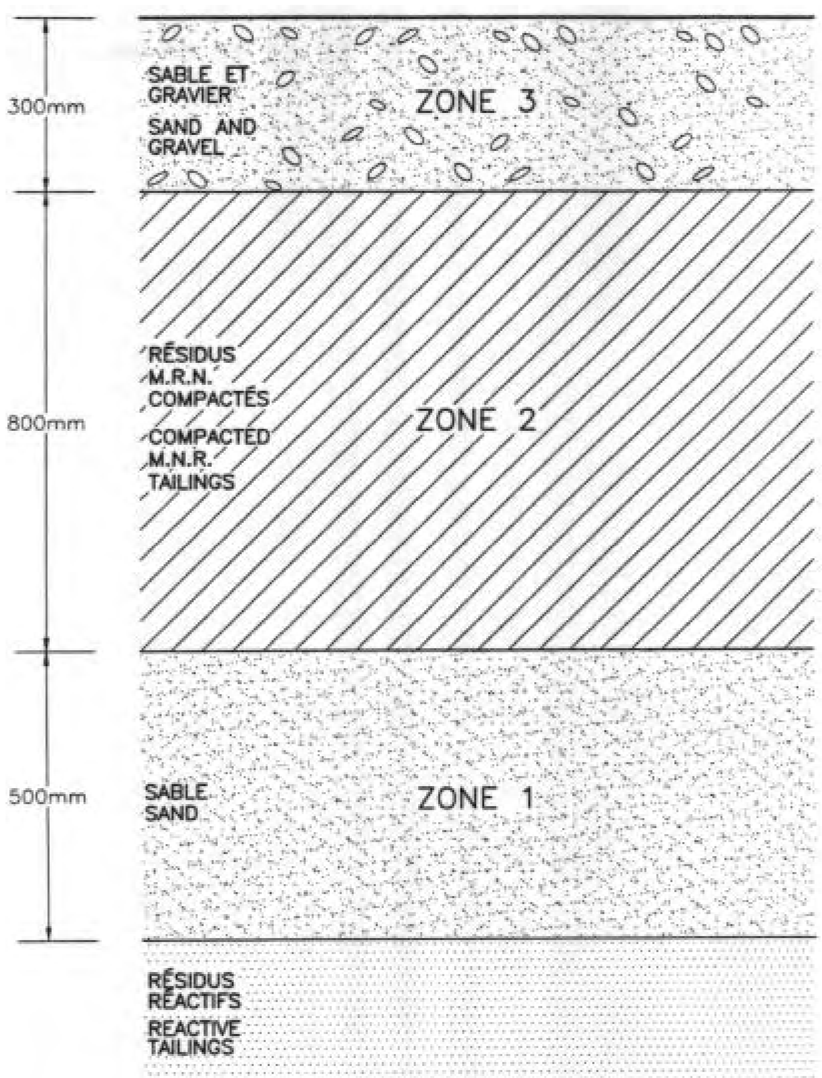
The design, construction and instrumentation of the dry cover system built on the LTA tailings impoundment and the external slopes is described by MEND 2.22.4a, MEND 2.22.4b and Ricard *et al.* (1997; 1999). A multi-layer cover system, using clean tailings for the moisture retention layer, was selected as the closure option based on the promising results obtained from MEND 2.22.2a,b,c. The goal of this project (MEND 2.22.4) was to assess the large-scale performance of a composite cover placed on the acid-generating tailings impoundment.

The LTA project is a landmark tailings cover system construction project because it is likely the first large-scale multi-layer cover system implemented in Canada. Approximately 60 ha of reactive tailings were covered with 0.5 m of sand placed directly on the tailings (Figure 4.4-16). This capillary barrier layer was overlain with 0.8 m of locally available compacted clean tailings. The compacted clean tailings were overlain with an upper layer capillary barrier material consisting of 0.3 m sand and gravel. The optimum configuration was calculated using a simplified one-dimensional cover design model and a two-dimensional saturated-unsaturated numerical model. Input data was based on laboratory characterization tests.

Construction of the 1.6 m thick composite cover on top of the LTA tailings impoundment proved to be a challenge due to the elevated phreatic surface in the tailings. The Zone 1 layer (Figure 4.4-16) needed to be placed and compacted during winter, while the underlying tailings surface was frozen, in order to enable heavy equipment circulation and prevent migration of LTA tailings particles into the Zone 1 layer (MEND 2.22.4a). Cover construction was completed during the winter of 1995/1996 and the summer of 1996 using a comprehensive QA/QC program. The final cost of the LTA composite cover, an area of almost 60 hectares, was $3.9M or $65,000/ha.

Field performance monitoring of the constructed cover system commenced in the fall of 1996 and continued for a two-year period. Several instrumented stations were installed on top of the impoundment and in the outer slopes of the dykes for monitoring *in situ* volumetric water content, *in situ* matric suction, and oxygen fluxes through the cover system (MEND 2.22.4b). The field performance monitoring demonstrated that the multi-layer dry cover system performed more efficiently than anticipated, based on the measured moisture conditions.

Figure 4.4-16 **Stratigraphy of the LTA Dry Cover System**



Source: MEND 2.22.4a

The key findings of this study are (MEND 2.22.4b):

- The fine material layer remained saturated, on the slopes and on the horizontal surface;
- The capillary barrier developed continuous flow conditions during significant recharge events, particularly at the base of slopes. This promoted a momentary increase in the degree of saturation of the cover resulting in a temporary increase in the net water infiltration rate;
- The presence of plateaus on the slopes significantly increased the up-slope water retention in the cover system by reducing the continuous discharge surface; and
- The mean tailings oxygen ingress flux was reduced by 95% over the observation period.

The LTA project confirmed that using locally available material (i.e. non-reactive tailings) for cover construction is a technically and financially viable closure option. Further details on certain aspects of this project can be found in McMullen *et al*. (1997), Aubertin *et al*. (1997a; 1997b), and Bussière *et al*. (1997; 1998).

Oxygen Consuming Organic Cover Systems: Background Studies (MEND 2.25.1)

Large quantities of organic material are now stockpiled, or may be available in the near future, from urban and industrial sources. The cities in Ontario are capable of producing approximately 680,000 tonnes of municipal solid waste (MSW) compost annually and create comparable amounts of sewage sludge, which is currently landfilled (MEND 2.25.1a). Peat from bogs in the Canadian Shield region, although not a waste material, represents a vast renewable source of organic matter. Peat bogs are often found near base metal and precious metal mines.

These materials may provide effective and affordable solutions to the reclamation of acidic mine tailings. In essence, a source of waste from one industry could be used to resolve problems associated with wastes from another industry (i.e. “two wrongs make a right”). A literature review of the physical and chemical characteristics of MSW compost and other organic materials (MEND 2.25.1a) illustrated that an organic cover on sulphide tailings could be beneficial in suppressing tailings oxidation and consequently AMD as a result of:

1. A Physical Oxygen Barrier*:* the organic layer may be saturated with water over at least part of its depth;
2. An Oxygen Consuming Barrier*:* the continued decomposition of organic material may create a large biological oxygen demand which can act as a sink for meteoric oxygen and dissolved oxygen within infiltrating water;
3. Chemical Inhibition*:* compounds and decomposition products in the organic material that leach into the tailings may inhibit the growth and metabolism of sulphate producing (acidifying) bacteria;

4. Chemical Amelioration*:* organic compounds in the organic material may cause the reductive dissolution of iron oxides (either directly or indirectly by providing metabolic substrates for bacteria), the reduction of sulphate, and the prevention of indirect ferrous sulphide oxidation and acid generation; and
5. By Reduced Water Infiltration*:* the decomposition and resultant compaction of an organic cover layer may result in the decrease of the hydraulic conductivity of the cover. This would decrease infiltration, thus decrease seepage of tailings pore water to groundwater.

MEND 2.25.1a stated that MSW compost or other organic wastes will be useful for the mining industry only if they provide a permanent, socially acceptable, and cost-effective solution to tailings abandonment. Transportation costs from the major MSW sources to remote tailings sites may be prohibitive for a mining company even if the MSW compost is provided at no charge or is subsidized (MEND 2.25.1a).

MEND 2.25.1b summarizes the results of a laboratory study of using MSW compost as a sulphide tailings cover material, which was conducted based on the positive findings of MEND 2.25.1a. Column studies of mature and fresh MSW compost over sulphide tailings were completed. MEND 2.25.1b concluded that:

- There is strong evidence of a reversal of AMD processes in the oxidized tailings;
- The mobilization of iron and sulphate was enhanced by the establishment of strong reducing conditions and the availability of organic substrates for the reductive dissolution of iron oxides by reducing bacteria;
- The compost and sand cover layer models were more effective at maintaining anoxic or other ameliorative conditions in the tailings than was the ploughed model during the nine-month simulation;
- Fresh compost treatments are much more effective than mature compost in maintaining high water content and strong reducing conditions at the compost-tailings interface;
- Fresh compost cover layers showed a greater resistance to the conduction of water that, along with high water content, would seal off the tailings from infiltration of atmospheric oxygen, precipitation, and surface water thus forming a physical oxygen barrier; and
- Compost quality tests showed that leachate from mature and fresh compost present a low environmental risk for use on mine lands.

The study results from MEND 2.25.1a and MEND 2.25.1b have provided much of the basis for continued work on the area of oxygen consuming organic cover systems. Key issues that need to be addressed further are degradation of the organic material (MEND 2.25.5; Tassé 2000) and potential replenishment of the organic matter. The potential for remobilization of metals also requires further consideration.

Two studies involving the use of organic matter in a dry cover system for reactive tailings (Strathcona – MEND 2.25.3; East Sullivan Mine – MEND 2.25.2 and 2.25.5) are discussed below.

Strathcona Tailings: Organic and Inorganic Cover Materials (MEND 2.25.3)

MEND 2.25.3 summarizes an evaluation of the effectiveness of organic covers in reducing acid generation from sulphidic tailings. Stogran and Wiseman (1995) and Elliott *et al.* (1997) also provide details of the study. The materials evaluated were lime-stabilized sewage sludge (LSSS), municipal solid waste (MSW) compost, and peat. An inorganic material (desulphurized tailings, DST) was also evaluated as part of the project for comparative purposes because of potential production of this material at operating mines. The evaluation included physical and chemical characterization of the tailings and potential cover materials, salt migration column tests, and pilot-scale tests.

The main objective of the study was to comparatively evaluate the effectiveness of various organic cover materials at limiting or reducing the impacts of acid generation on the environment from acid generating tailings. The organic cover materials tested demonstrated that there were significant differences in the ability of each material to provide a beneficial tailings cover. The MSW compost and LSSS were found to be very good oxygen barriers as a result of acting as an oxygen sink. In contrast, the peat cover, which had a high humus component and was therefore resistant to further decomposition, was observed to be very susceptible to oxygen diffusion (MEND 2.25.3).

The test results reported illustrate that the LSSS performed best as a tailings cover (i.e. limit ingress of oxygen to underlying sulphidic tailings). The peat showed the least favourable cover material characteristics. The desulphurized tailings also performed well but as a fine textured physical barrier for oxygen diffusion, as opposed to an oxygen sink with the LSSS cover material. MEND 2.25.3 recommended that field trials were required to properly evaluate the potential of the two promising cover materials, whether on their own or blended to improve economics. Longevity is one of the concerns for the LSSS as a potential cover material (i.e. at what point would the organic matter have to replenished).

East Sullivan Mine: Oxygen Consuming Organic Cover Material

The East Sullivan Mine is located near Val d'Or in northwestern Québec. Metals extracted from the mine included copper, zinc, gold, and silver between 1949 and 1966 resulting in approximately 15 Mt of tailings (3.6% sulphur) with a surficial area of 1.36 km^2 (Tassé *et al.* 1997). A variety of organic wastes (bark, fiberboard, pulpwood, and sanding dust) were deposited on the tailings since 1984. Historically up to 6 m of organic waste material was placed although more recently material placement was limited to 2 m and fiberboard excluded.

Comparison of water quality from covered and uncovered portions of the tailings illustrated the potential of the organic cover material in mitigating AMD. Measured pH increased from 4.5 to 7 from 1988 to 1992, respectively, with a corresponding order of magnitude decrease in zinc concentrations (Tremblay 1994). Oxygen and nitrogen decreased while methane and carbon increased within the one metre cover (Tremblay 1994).

Field test plots 2 m high and 20 m^2 were constructed based on the promising water quality results obtained (MEND 2.25.2). Pore gas monitoring revealed that oxygen concentrations were reduced to 4% at depth with a corresponding increase of carbon dioxide to over 10%. Pankewich *et al.* (1998) provides a summary of the isotopic geochemistry of biogenic gases released by the East Sullivan cover system.

Tassé (1999, 2000) concluded that the long-term efficiency of a wood waste cover strongly relies on the water saturation that can be reached and maintained at the base of the wood waste cover. The anaerobic environment would then help to keep the amounts of nutrients and readily metabolizable organic substrates at a high level for favourable oxygen consumption. The implication is that an oxygen consuming organic cover system may be better suited for “wet” climate environments where the term “wet” implies that annual precipitation meets or exceeds potential evaporation. MEND 2.25.5 and Tassé (1999) stated that old wood wastes are less efficient than recent ones and that well drained woodpiles (i.e. wood waste covers) are much less efficient than those with a water saturated base. This is a positive result with respect to the use of wood waste as a cover material for tailings. However, this could imply that a well-drained waste material such as waste rock may not be a suitable long-term underlying material for oxygen consuming organic covers. MEND 2.35.2b concluded that a wood bark cover is not a good technique for reducing acid generation in sulphide-bearing waste rock. Alternatively an organic layer could be one component of a waste rock cover system. However, as with all “alternate” cover materials the feasibility is closely associated with transportation of the organic material to the mine site.

There are other practical considerations making oxygen consuming cover materials for waste rock less promising. Waste rock piles have much steeper side slopes, which makes it difficult to place the organic material. In addition, as a result of end dumping the coarsest waste rock will typically be located on the side slopes thus enhancing airflow. It would be difficult to place the organic material effectively at these locations without a sub-base layer of material and recontouring of the side slopes.

Organic oxygen consuming covers for reactive tailings appears to be a viable solution for closure, provided the site climate conditions are favourable. However, use of organic matter for covering sulphide-bearing waste rock requires fundamental and applied (i.e. field trials) research.

Cominco Ltd., Kimberley Operations, British Columbia

The Cominco Ltd. Sullivan Mine is scheduled for closure in the year 2001 after more than 90 years of continuous operation. The iron-lead-zinc sulphide deposit, discovered in 1892, is located in Kimberley, British Columbia. Tailings disposal began in 1923 following development of a differential flotation process capable of separating complex ore into lead, zinc, and iron concentrates. A total of 90 million tonne of tailings containing pyrrhotite and pyrite have been discharged since 1923 to ponds occupying 373 hectares of land (Gardiner *et al.* 1997).

The siliceous tailings ponds are located in a relatively flat area with bedrock and till deposits forming topographic highs to the north and south. A thin veneer of till rests on the majority of the bedrock highs. However, relatively continuous till deposits exist between the bedrock highs and range in thickness up to 30 m. Discontinuous glaciofluvial deposits of sand and gravel cover the till surface. Prior to tailings discharge, the pond area was a marshy flat that probably represented a groundwater discharge area (Day and Harpley 1993).

A number of AMD abatement technologies are incorporated into the reclamation plan for the tailings pond and include: collection and treatment of seepage and surface water; interception and diversion of an up gradient surface watercourse; and construction of a soil cover system.

A system of ditches, pumps, and pipelines are in place to collect surface seepage from the tailings pond dykes. The seepage is combined with mine water and tailings effluent before being pumped to a treatment plant. The main source of AMD following closure, in terms of both volume and metal loading, will be drainage from underground workings (Gardiner *et al.* 1997). Hence, the need to treat mine drainage following closure was a key factor in determining the optimum soil cover system design for the tailings ponds. The drainage water treatment plant uses lime in a high-density sludge circuit that neutralizes the acid and precipitates metal into a free draining, non-leaching sludge (Murdock *et al.* 1994). The plant is designed to treat a normal maximum of 18,000 L/min, but can be operated at 27,000 L/min. Sludge is discharged to a pond with a life expectancy of 150 years at expected post-closure production rates. Mine drainage and seepage will be treated during a five- six-month period each year during winter and spring depending upon the precipitation. Improvement to the effectiveness of seepage and groundwater collection in the tailings pond area is a continuous effort. Construction and monitoring of the siliceous tailings field test plots was part of an effort to evaluate alternate soil cover systems to control the ingress of oxygen and infiltration of water to the underlying tailings while also providing a medium for sustainable vegetation.

The test plots were constructed using a coarse low-density reject rock from the mill (float rock) and a cobbly non-plastic till. A total of seven test plots were constructed, four in 1993 and three in 1994. The three additional test plots were constructed in 1994 because the field lysimeters installed in the 1993 test plots were damaged during placement of the cover materials. The design of these three plots were identical to Plots 2, 3, and 4 and were designated Plots 2s, 3s, and 4s. The dimensions and configuration of each test plot are shown in Figure 4.4-17.

The float rock layer was intended to limit migration of salts to the root zone and function as a capillary break below a layer within the soil cover system that remained at a high degree of saturation. The float rock has provided a continuous hydraulic break between the overlying growth medium and the underlying tailings. Hence, vegetation biomass has increased during the life of the test plots as well as for historic reclamation test plots studied at the site since 1978.

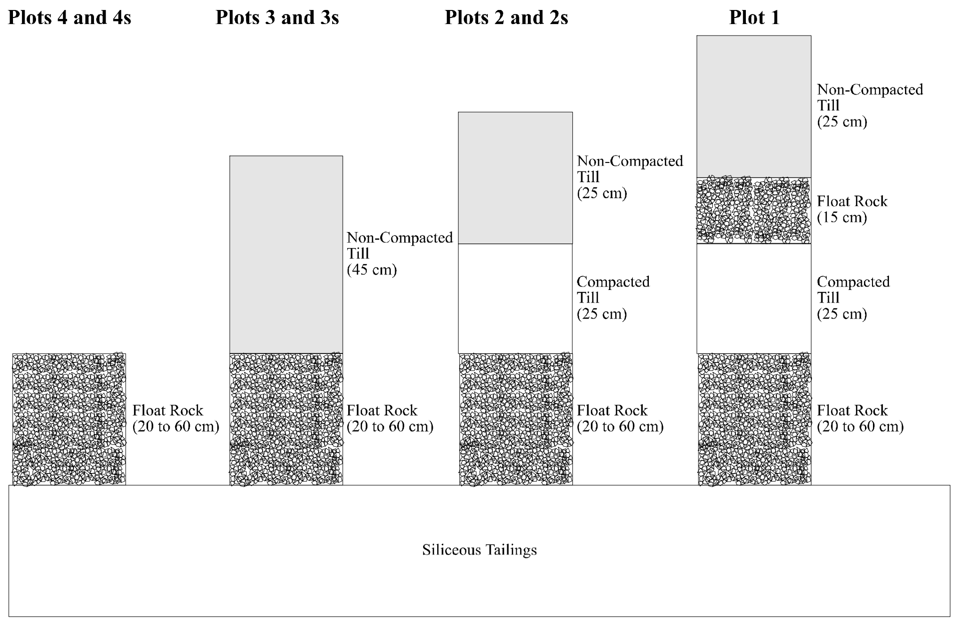
The test plots have not controlled the ingress of oxygen since their construction. In general, the cover material decreases in moisture content during the summer following moist winter and spring conditions. This performance was observed for the test plot 1 design where a capillary barrier material overlays the compacted layer. The compacted layers of the test plots did not possess sufficient moisture retention to prevent migration of moisture to the soil-atmosphere interface (i.e. the top of the cover) for subsequent evapotranspiration during the summer.

The volume of water entering the siliceous tailings underlying the test plots was a function of three site-specific factors: the test plot profile (geometry, material type and preparation), the amount of rainfall during the fall (i.e. year round climate), and the amount of spring snowmelt (hence snowpack). The site can be characterized, as experiencing an arid to semi-arid climate on an annual basis because an annual moisture deficit is prevalent. However, while arid conditions are prevalent during the summer the site typically experiences semi-humid to humid conditions during late fall, winter, and spring. The consequences of climatic variation at the site on soil cover system design and performance are significant because a system that maintains a tension saturated layer during dry climate conditions while also controlling water infiltration is not technically feasible using the available cover materials.

In summary, climate conditions at the site are a key factor controlling performance of the test plots. Higher annual incident precipitation will generally lead to an increase in percolation to the tailings underlying a given test plot. In general, precipitation that contributes to snowpack, or occurs during the winter, will increase net percolation to the tailings while summer rainfall is buffered by the presence of cover material (i.e. moisture store and release) and does not percolate to the underlying tailings (O'Kane *et al.* 1999b).

Gardiner *et al.* (1997) summarized that no cost benefit existed to construct the "ideal" cover system using a capillary break or a highly engineered compacted layer. The fact that the site is committed to collection and treatment of AMD from all sources, not just the tailings, using a state-of-the-art treatment system, was a controlling factor in the design of the dry cover system for the tailings.

Figure 4.4-17 **Schematic Diagram of the Cominco Ltd. Kimberley Operations Siliceous Tailings Test Plot Profiles**



Source: O'Kane *et al.* 1999b

Mine Poirier, Québec

Poirier was an underground copper and zinc mine located in northern Québec near Joutel that operated between 1965 and 1975. Milling of the ores produced approximately 5 million tonnes of high sulphide tailings, which were deposited in a surface impoundment spanning an area of nearly 46 hectares. A thin layer of spilled tailings had accumulated at several locations external to the main impoundment, having a combined volume of about 0.3 million tonnes and covering an area of nearly 28 hectares. Seepage from the tailings impoundment and spill areas has contributed acidity and metals to two nearby creeks (Lewis *et al.* 2000). An important feature of the site, however, is the presence of a low permeability clay layer beneath the main tailings deposit, which acts as a barrier to limit groundwater contamination.

Rio Algom undertook an assessment of the property in 1996 for the preparation of a reclamation plan. Lewis *et al.* (2000) provide details on the reclamation plan, which represents the culmination of a series of studies directed at developing an environmentally sound method of returning the site to a natural state. The primary objective of the reclamation plan was to reduce chemical loadings from the tailings basin to reduce ecological risk to an acceptable level and allow sections of the nearby creeks to recover naturally. The following five options were considered to reduce loadings from the site:

1. Soil cover: a 1 m cover of clay;
2. Geomembrane liner with a protection layer: 0.5 m of clay and 1 to 1.5 m of till;
3. Collect and treat: collection ponds, treatment plants and sludge disposal;
4. Clay cover with frost protection: 1 m of clay and 1.5 m of till; and
5. Bury tailings: in the muskeg adjacent to the site.

Option 2 became the preferred option following discussions with provincial government officials and the prediction of loadings from the tailings basin to the receiving environment for each option. This option involves a cover consisting of a geomembrane liner over the main tailings deposit and relocated mine waste materials (e.g. spilled tailings), together with a 1 m thick soil layer to protect the liner. The purpose of the liner is to substantially reduce (essentially eliminate) infiltration through the cover (Lewis *et al.* 2000). The cover is also designed to function as an oxygen barrier to prevent further oxidation of the tailings.

Construction of the dry cover system over the Poirier tailings basin began with preparatory works in the fall of 1998 (Lewis *et al.* 2000). Gathering the spilled tailings and other reactive materials and placing them in the central waste pile of the tailings basin was one of the first task completed. Cleanup of the spilled tailings required the removal of all materials down to the clay layer. This involved removal of muskeg, trees and bushes over the layer of peat and clay and the bulk of the work was done in late fall and winter when the ground was frozen. The internal tailings beach was graded and compacted in preparation for the application of the geomembrane liner, which required removal of tree stumps, some rock and decant structures. A support layer of sand was placed below the liner in several sections. Installation of the geomembrane liner was completed in 1999, and consisted of placing two different types of liner. A smooth 60 mil high-density polyethylene (HDPE) liner was used on the beach and over the central waste piles (surface area of 37 ha), while a textured 80 mil HDPE liner was used on the slopes, with a geonet drainage layer above the textured liner. A 0.5 m thick layer of clay was placed on all of the liner to protect the liner, and provide a secondary natural sealing media for any imperfections that may occur in the liner (Lewis *et al.* 2000). A layer of till, ranging in thickness from 0.5 m on the beach and waste pile area to 1.5 m on the slopes, was subsequently placed over the clay layer. This layer provides physical protection of the clay by inhibiting frost penetration and provides a suitable medium for vegetation (Lewis *et al.* 2000). Completion of cover system construction and commencement of field performance monitoring occurred in 2000.

The Poirier mine site has the distinction of being the first full-scale demonstration project in Canada of utilizing a geomembrane in a dry cover system. The reclamation plan for the Poirier site will provide considerable improvements to the surrounding habitat (Lewis *et al.* 2000). Further research is required with respect to the longevity of geomembrane liners as a component of a dry cover system.

<u>Other Case Studies Using Alternate Cover Materials</u>

Pulp and paper (P&P) residues were used as capillary barrier material in a field test cover system for reactive tailings at the Eustis mine site near Sherbrooke, Québec (Cabral *et al.* 1997). The P&P residues originated from paper deinking processes and were described as a spongy, partially saturated material, capable of absorbing large quantities of water. The deinking residue layer acts as an oxygen barrier in two ways: 1) due to its high moisture retention capacity, it keeps a high degree of saturation; and 2) due to the high organic content of the deinking residues and microbial activity, the barrier consumes atmospheric oxygen (Cabral *et al.* 1999). The approximate stratigraphy of the field test plot consisted of non-oxidized tailings of variable thickness, overlain by 1.5 m of compacted, fresh deinking residues and finally, a 0.2 m layer of a compost-deinking residue mixture (Cabral *et al.* 1997). The field performance monitoring program consisted of measurements of volumetric water content, temperature, gaseous oxygen concentrations and permeability. The key monitoring results are the oxygen profiles and permeability measurements, which show the cover system is functioning as both an oxygen barrier and water infiltration barrier. Additional monitoring results and details on this test cover project are reported in Cabral *et al.* (1997). Additional sites in the Sherbrooke area using deinking residues from the pulp and paper industry include Moulton Hill and Albert Mine. A small field-scale application (Clinton Mine, near Woburn, Québec) that used deinking residues for the moisture retention layer in a waste rock dry cover system is reported in Cabral *et al.* (1999).

McGregor (1997) reported on a field investigation of a "self-sealing/self-healing"(SS/SH) barrier undertaken at Falconbridge Ltd.'s East Mine tailings near Sudbury, Ontario. The test barrier was installed in four stages by first excavating the overlying oxidized tailings, then reactive material was placed and compacted followed by high Fe-S material (mill washings), and finally reapplication of a layer of overlying oxidized tailings. The interface of the overlying mill washings and the underlying reactive material forms the SS/SH barrier. The reaction of dissolved compounds form precipitates at the interface of the two materials. The aqueous concentrations decrease, thereby creating a concentration gradient, which leads to diffusion of additional reagents and further precipitation until the pores of the parent materials are filled. The field measured average vertical hydraulic conductivity of the interface was four orders of magnitude lower that the surrounding tailings (McGregor 1997). The pore gas diffusivity of oxygen across the SS/SH barrier was less than 1/1000 of the overlying tailings. This resulted in nominal gaseous oxygen concentrations of 3.4% below the barrier, as compared to 18.5% above the barrier. McGregor (1997) reported that the cost for installation of the SS/SH barrier would range from $30,000 to $50,000 per ha. It is noted, however, that a native cover material would likely be required to establish vegetation for long-term closure considerations using this technique, thus adding to the unit cost per hectare. No further work has been reported on the SS/SH barrier system.

4.4.4.2 Waste Rock Dry Cover System Case Studies

Heath Steele (MEND 2.31.1)

A waste rock study was initiated in the spring of 1988 at the Heath Steele Mines (HSM) site in New Brunswick. The objectives of the study were to develop strategies for the long-term management of acid generating waste rock, to evaluate the performance of a soil cover, and to assess the cover effectiveness as a method for long-term management of AMD.
The project was developed and conducted in the following five phases:

Phase I: Selection of four waste rock piles for monitoring and evaluation (MEND 2.31.1a);

Phase II: Installation of monitoring equipment in the four piles identified in Phase I to define waste rock characteristics and background data (MEND 2.31.1a);

Phase III: Geotechnical and geochemical column testing to evaluate the performance characteristics of potential cover system designs (MEND 2.31.1a);

Phase IV: Placement of a soil cover system on Pile 7/12 and performance monitoring (MEND 2.31.1b); and

Phase V: Continued monitoring at Pile 7/12 (MEND 2.31.1c).

The Noranda Technology Centre reviewed and tested a range of cover options and recommended the following composite cover for Pile 7/12 (surface area of 0.25 hectares):

- a 30 cm base granular layer;
- a 60 cm saturated glacial till layer;
- a 30 cm overlying coarse grained granular layer; and
- a 10 cm erosion protection layer.

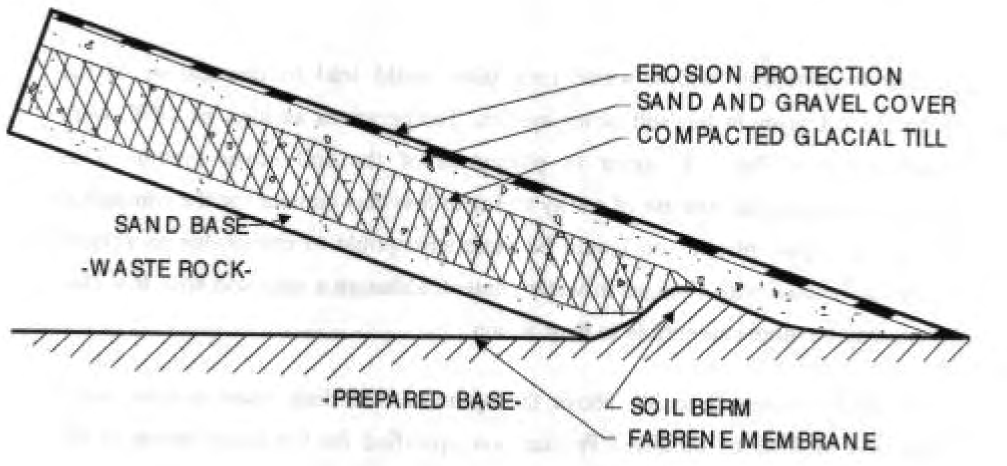
In 1989, Pile 7/12 was moved to its present location and reconstructed on a prepared sand base, underlain by an impermeable membrane to permit the collection of leachate at the base of the pile before and after placement of the soil cover. A dry cover system with the above specifications was constructed in late summer of 1991. Figure 4.4-18 shows the construction details of the soil cover system used on Pile 7/12.

Monitoring of oxygen and temperature conditions continued monthly at all four piles after the placement of the cover on Pile 7/12. Infiltration rates and water quality were also measured at Pile 7/12, as was the moisture conditions of the compacted till layer.

The following conclusions were made after the completion of Phase V (1995-1996 monitoring program):

- The influx of oxygen to the pile was minimized, with internal waste rock gaseous oxygen concentrations typically being well below 1% (20.9% equals atmospheric conditions);
- The volume of acid leachate escaping from the pile was drastically reduced representing only two percent of total precipitation incident on the pile;
- The loading of metals and sulphate decreased by 99%;
- The potential cost of lime for treatment was reduced by $187/yr per 1,000 tonne of waste rock due to reduced seepage volume, and a further $8.70/yr per 1,000 tonne of waste rock due to gradual seepage quality improvement;
- It will take many decades to flush out the stored oxidation products resulting from the Pile 7/12 material being exposed to atmospheric conditions prior to the start of the project (one pore volume was estimated at 30 years); and
- Evidence exists that de-watering of part of the cover system is a concern for the long-term effectiveness of the cover.

Figure 4.4-18 **Soil Cover Construction Details at Heath Steele**



Source: Yanful *et al.* 1993c

Ultimately, composite soil covers were found to be an effective method for reducing the oxidation reaction in sulphidic waste rock piles, thus significantly reducing their impact on the environment. The active layer (i.e. compacted layer) must be designed, constructed, and sustained so that its integrity and moisture content are maintained to achieve this objective. Composite soil covers are suitable for areas only where precipitation enables the active layer to maintain its moisture content, such as at the Heath Steele site. Ongoing maintenance would also be required to prevent the establishment of trees or shrubs on the cover, the roots of which could threaten the integrity of its sealing ability.

MEND 2.31.1c reports that the cover system has resulted in a cost savings for lime in the range of \$196/yr per 1,000 tonnes of waste rock. Approximately 94% of the savings were observed shortly after construction of the cover due to reduced flushing flows through the cover. Other benefits for treatment include a low volume of flow to be treated and effluent water quality consistency.

MEND 2.21.3a, which provides a critical, peer review of this project, states that the soil cover system installed at the Heath Steele site performed satisfactorily. It was also stated that the results of the research indicated that the capillary barrier concept was attainable under field conditions and will result in a reduction of the influx of infiltration and oxygen, thereby reducing the potential for acid generation.

Yanful *et al.* (1993b), Yanful *et al.* (1993c), Bell *et al.* (1994) and Bennett *et al.* (1995) also reported detailed summaries on certain aspects of this project.

Myra Falls Operations (MEND 2.34.1)

Northwest Geochem, in conjunction with Powertech Labs Inc., developed and tested a cementitious material, which incorporated mine tailings as a cover for acid generating waste rock at Boliden's Myra Falls Operations near Campbell River on Vancouver Island (MEND 2.34.1). The primary purpose was to evaluate the long-term stability of shotcrete in a field environment. A large-scale field application of a shotcrete cover on a waste rock pile was conducted (3500 m^2). Mixes utilizing imported aggregate and mine tailings were tested. Visual inspection of the shotcrete cap over the three-year test period indicated that the overall durability of the material was good (MEND 2.34.1). No frost damage was evident and no movement of the cap was detected. Some cracks were observed and likely correlated with areas where the shotcrete was applied at less than the 75 mm thickness specified for the test.

The results of the laboratory testing indicated that the compressive strength of the mixtures exceeded the design objective. The toughness index and flexural strength were lower than standard values for shotcrete. Some reduction in compressive strength was observed in the tailings mix after 400 days. It is believed this loss in strength was a result of oxidation of the sulphide minerals in the tailings material used in the shotcrete. Permeability of the shotcrete was approximately 10^{-14} m/s in the aggregate mix to 10^{-10} m/s in the tailings mix.

The major conclusions with respect to long-term stability were:

- The success of the cover depends on the stability of the waste rock dump. Geotechnical studies are required to estimate any movement of the final design slope; and
- It remains to be determined what the effects will be on the shotcrete cover as a result of placing overburden and vegetation.

Jones and Wong (1994) reported that the total cost of the shotcrete cover application was \$18.46 per m^2, with a significant amount of the cost being the transportation of aggregate to the site (\$7.10 per m^2). These costs do not include placement of the required growth medium layer overlying the shotcrete. It is important to note that MEND 2.34.1 did not address the effectiveness of the shotcrete cover in restricting acid generation in waste rock (i.e. oxygen ingress and water infiltration).

An applied research program was initiated in 1996 to develop a cover system for the waste rock material at the Myra Falls site. The objective was to develop a cover system that controls the ingress of oxygen and infiltration of water, while providing a medium for sustainable vegetation. Four test plots (three multi-layer cover systems and one control plot) were constructed in 1998 (O'Kane *et al.* 1998c); the design of the test cover systems was based on laboratory characterization and numerical modelling studies conducted at the University of Saskatchewan. Two of the test covers consisted of a compacted layer of modified local till, one amended with bentonite and the other with flyash, and an overlying layer of non-compacted till. The third test cover consisted of a compacted till layer and an overlying non-compacted till layer. Field instruments were installed to monitor volumetric water content, soil suction, temperature, net percolation, and gaseous oxygen concentrations. Preliminary results show that the cover system with a compacted layer of till ameliorated with bentonite is performing best with respect to oxygen ingress and water infiltration. However, the optimum cover system design for the waste rock material will be determined following further investigations at the site. A significant design consideration is the fact that the waste rock material is situated on a valley wall and thus, subjected to basal inflow conditions.

Equity Silver, British Columbia

A soil cover system was installed on the waste rock pile at the Equity Silver Mine in 1991. The site is located in the central interior of British Columbia, approximately 575 km north-northwest of Vancouver. The mine is situated on a plateau in a humid alpine environment. Historical site records indicate the average annual total precipitation is 650 mm with approximately 60% of the precipitation occurring as snow.

Two soil layers, compacted till and non-compacted till, form the soil cover system on the three waste rock piles (Main pile: 52 Mt, 41 ha; Southern Tail pile: 18 Mt, 31 ha; and Bessemer pile: 10 Mt, 29 ha). The Southern Tail pile was created by backfilling waste rock into an open pit. The side slopes of the other two piles were graded to a constant slope with a design maximum of 21^{o}. The piles were covered with a 0.5 m thick compacted till layer. A non-compacted till layer, 0.3 m thick, was placed over the compacted till. The Main pile soil cover system was completed in 1994. The Southern Tail pile soil cover system was completed in 1991. The soil cover system for the Bessemer pile, which was active until mine closure, was completed in the summer of 1996.

The soil cover system was designed to reduce the infiltration of water and the rate of oxygen diffusion into the underlying waste rock material. The compacted layer maintains a high degree of saturation and has a low hydraulic conductivity. Saturation in the compacted layer is required to reduce the diffusion of oxygen across the cover into potential zones of acid generation within the underlying waste rock material. The low hydraulic conductivity of the compacted layer allows the layer to act as a barrier to water infiltration. The non-compacted layer, over top of the compacted layer, was designed to provide protection for the compacted layer against erosion, freeze/thaw, and desiccation. Vegetation was subsequently established on the cover system.

O'Kane *et al.* (1998b), Saretzky (1998), O'Kane (1996), Swanson (1995), and Owuputi *et al.* (1995) conducted field performance measurements, laboratory characterization, and numerical modelling to demonstrate cover system performance at the site. Field test work indicates net infiltration to the underlying waste rock was reduced to less than 5% of the average annual precipitation as a result of the presence of a compacted till and a non-compacted till layer placed on the waste rock surface. However, seepage discharge being collected in perimeter ditches suggest that infiltration rates through the pile are approximately an order of magnitude greater than field measured rates (Saretzky and Wilson 2000). The results of a hydrologic investigation carried out by Saretzky (1998) show that groundwater may be discharging into the waste rock pile, leading to higher than anticipated seepage rates at the toe of the pile. Field drilling and piezometer installation was carried out in 2000 to determine the groundwater regime in the vicinity of the waste rock pile; an interpretation of the test results is forthcoming (Aziz 2000).

O'Kane *et al. (*1998b) and O'Kane (1996) illustrated the gradual decrease in gaseous oxygen concentrations within the waste rock pile (from nearly 10% to less than 3%) following placement of the till cover material. This demonstrated that oxygen consumption was greater than oxygen supply. Swanson (1995) used a field calibrated soil-atmosphere saturated-unsaturated numerical model to demonstrate that oxygen ingress across the till cover layers can be expected to be reduced by as much as 98% of uncovered waste rock conditions. Recent field monitoring results, however, have shown a considerable variation in gaseous oxygen concentrations within the waste rock pile, suggesting potential drying of the cover system or the entry of atmospheric oxygen along the edges of the dump (Aziz 2000). Aziz and Ferguson (1997) and Wilson *et al.* (1997) provide detailed summaries of the cover research work conducted at Equity Silver Mine.

A care and maintenance program has been implemented at Equity Silver Mine to protect the integrity of the dry cover system (Aziz 2000). This involves removing woody species that establish on the cover system and repairing drainage ditches that exist on the cover to shed runoff from rainfall events and snowmelt. In addition, snow that accumulates in north-facing drainage ditches is removed each year prior to spring melt to prevent the ditches from icing over and becoming non-functional (Aziz 2000).

Bersbo, Sweden Pilot Project

The Bersbo, Sweden pilot project was the first full-scale project in Sweden of remediating sulphide wastes using state-of-the-art dry cover system design methodologies. Performance has been monitored since 1989. The 700,000 m^3 of waste rock covered was between 90 and 600 years old. It was relocated from various historic mines in the area into two locations with surface areas of 2.8 ha and 3.5 ha, respectively. The two deposits were covered with a sealing layer and a protective layer designed to control the ingress of oxygen and infiltration of water to the underlying waste rock (Lundgren 1997).

The sealing layer of the smaller deposit (2.8 ha) consisted of a concrete like product made of pulverized coal flyash, cement and water. The mixture was blended into a paste, and grouted into a 0.25 m thick layer of crushed rock aggregate. A 0.5 m layer of compacted till was placed on a thin filter layer of tailings, which had been placed directly on the larger deposit (3.5 ha). The sealing layers of the two deposits were overlain by 2 m of non-compacted till.

Percolation through the covers was monitored with collection lysimeters installed beneath the covers. Percolation through the grout cover system decreased to 12% of incident precipitation as compared to before placement of the cover system. The compacted till cover system decreased net percolation to 3%, as compared to the uncovered condition. The latter “natural” cover system also performed well as an oxygen ingress barrier because the compacted till layer maintained a reasonable high degree of saturation, although problems associated with the upper areas of the sloped surfaces were noted. The cementitious compacted layer did not maintain a high degree of saturation, and as a result it did not perform as an oxygen ingress barrier to the same extent as the compacted till cover system.

Oxygen transport was governed by advective transport processes before placement of the cover material (Lundgren 1997). Oxygen supply did not become a limiting factor for oxidation of the potentially acid forming waste material until the cover material was applied. Gaseous oxygen concentrations were near atmospheric conditions prior to placement of the cover material. The mean gaseous oxygen concentration within the waste rock piles during the period covering 1991-1995 following placement of the cover was 3.2% for the first pile and 0.4% for the second pile.

Rum Jungle Mine, Northern Territory, Australia

Uranium and copper were mined at Rum Jungle in the Northern Territory, Australia between 1954 and 1971. An acidic drainage problem developed during this period. A detailed strategy for rehabilitating the mine site was formulated following exhaustive investigations and studies in the 1970's. It was found that the ingress of water and oxygen from the major overburden heaps were contributing to the problem. The strategy developed therefore involved sealing of the overburden. This reduced pyrite oxidation and the transport of heavy metal pollutants.

A four-year rehabilitation project funded by the Australian Government commenced in 1982 and cost $18.6 million. This involved covering three waste rock piles at the site, two of which (White's and Intermediate) have been extensively monitored to obtain temperature and gaseous oxygen concentration profiles. Field monitoring results have indicated that the rehabilitation operation has proved successful since rehabilitation was completed. Oxidation of pyrite minerals in the heaps has slowed and the quantities of heavy metals reaching the Finnis River were greatly reduced. Conclusions reached include the following:

- The concept of sealing the heaps to prevent the ingress of water and oxygen is a feasible and cost-effective method of controlling, not completely stopping, AMD; and
- The planning, design, implementation and monitoring of the Rum Jungle Rehabilitation Project were well documented. The results have proven useful for assessing a wide range of issues involved in the AMD process because monitoring was a large part of the project well before the actual works commenced.

Bennett *et al*. (1988) and Applegate and Kraatz (1991) present detailed summaries of this project. Harries and Ritchie (1981; 1985; 1987) presented and analyzed gaseous oxygen profiles and temperature profiles obtained from the covered waste rock dumps.

Timms and Bennett (2000) provided further insight into the performance of White's waste rock pile cover system at Rum Jungle fifteen years after installation. The effectiveness of the cover was estimated by comparing the pre- and post-rehabilitation infiltration and overall oxidation rates. Timms and Bennett (2000) stated the following conclusions:

- The infiltration rate into White's dump has increased significantly since the 1994/95 wet season and has been above the design specification of five percent of rainfall for the last five years; however, it is still between five and ten times lower than the estimated rate before rehabilitation; and

- The overall oxidation rate in White's pile is approximately a factor of three lower than the pre-rehabilitation overall oxidation rate; it is remaining steady or increasing slightly with time.

Kidston Gold Mine, Queensland, Australia

Durham (1999) and Durham *et al.* (1999) reported on the performance of two cover system trials constructed at the Kidston Gold Mines in the state of Queensland, Australia. Average annual potential evaporation for the site is approximately 1900 mm and exceeded average annual precipitation by almost 1200 mm. Three years of monitoring demonstrated the promise of a moisture store and release cover system designed for the site, as well as the importance of maintaining good vegetation. The field trial cover systems were constructed using well-graded, non-acid forming run-of-mine waste. A field test plot consisting of 2.5 m of non-compacted oxidized run-of-mine waste was constructed. A second test plot was constructed with a 0.5 m compacted layer and an overlying 1 m layer of non-compacted material. Run-of-mine oxidized waste was used as the construction material for both of the cover materials in the second test plot. No moisture has reported to the lysimeters (see Bews *et al.* (1997) for design details) installed to measure percolation to the underlying waste material during the three-year monitoring period (Durham 1999).

A progressively longer period of time was required to reduce moisture conditions in the run-of-mine cover material to pre-wet season conditions following the wet season precipitation. Vegetation is not as well established as compared to earlier in the monitoring period and appears to be a key factor for successful performance of the store and release cover system trials. Transpiration is a key component of the water balance and appears to significantly reduce moisture conditions in the run-of-mine cover material. It is postulated that nutrient deficiency is the likely contributor to the vegetation problems and is being addressed (Durham 1999).

BHP Iron Ore, Western Australia, Australia

BHP Iron Ore initiated a program in January 1995 at their Mt. Whaleback operation in Newman, Western Australia to develop a decommissioning plan for the waste rock material. The Mt. Whaleback iron ore mine is located in the Hamersley Iron Province in the northwestern corner of Australia and situated adjacent to Newman, Western Australia (WA), approximately 1200 km north-northeast of Perth, WA. Development of the mine started in 1968. The mine currently produces approximately 16 million tonnes of iron ore and moves 50 million tonnes of waste material annually. More than 2 billion tonnes of waste rock were deposited during the past 30 years. Ultimately, the operation will deposit a total of approximately 4 billion tonnes in waste rock piles constructed near the open pit.

The primary research program included the development of technology for the long-term performance of the waste rock piles with respect to vegetation, slope stability, surface runoff, erosion, and water infiltration. Field performance monitoring of moisture store and release cover systems constructed on horizontal and sloped waste rock surfaces are ongoing. The cover systems are designed to release infiltration to the atmosphere as evapotranspiration. The objective is to control acidic drainage by preventing moisture movement into and through the waste rock material (O'Kane *et al.* 1999a). The cover systems are constructed using suitable run-of-mine waste material to minimize closure costs (O'Kane *et al.* 1998a).

Field data collected to date demonstrates that the cover system on a sloped surface performs significantly different as compared to the cover system on a horizontal surface (O'Kane *et al.* 1999a). Evidence that infiltration advanced to a depth of approximately 2 m below the horizontal surface of Test Plot No. 1 (2 m of cover material) and Test Plot No. 2 (4 m of cover material) was obtained during the August 1997 to June 1999 monitoring period. The infiltration did not advance to a depth of 2 m during the same monitoring period at the sloped surface field test plot. It would appear enhanced surface runoff at the sloped surface test plot led to the change in cover performance, and significantly reduced infiltration. The field performance monitoring data for a twenty-one month period demonstrates the potential for success of the "moisture store and release" type cover system at the BHP Iron Ore Mt. Whaleback site.

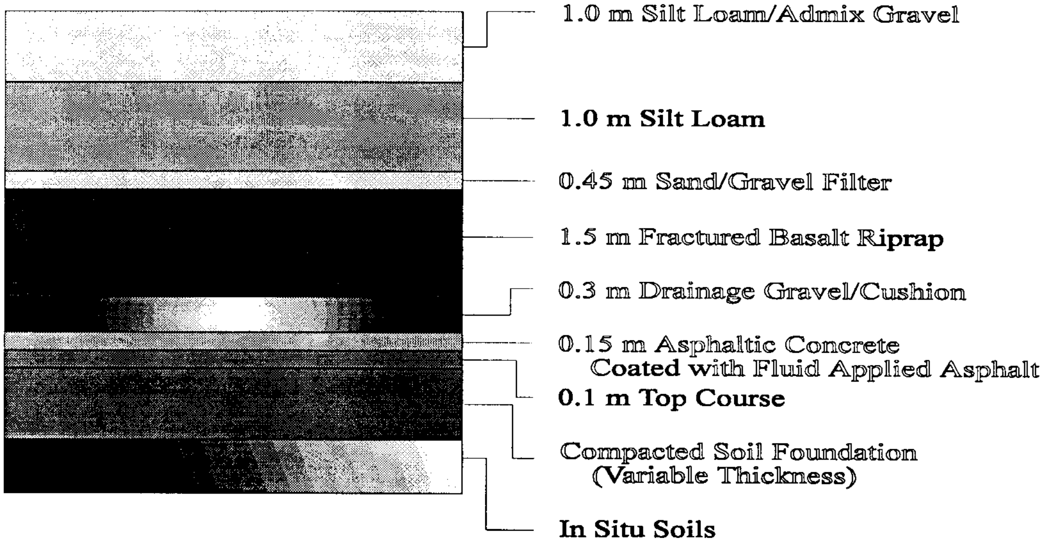
Hanford Site: Low Level Nuclear Waste

The U.S. Department of Energy (DOE) initiated an extended study program at the Hanford site near Richland, Washington in 1985 to isolate and dispose of buried wastes over the long term. The Hanford site contains about 10.3% of all low-level nuclear waste in the Unites States (Fisher 1986). Although it is not an acidic waste site it provides valuable information on the use of layered soil systems and vegetation to control infiltration.

The key objective of the Hanford cover system was to isolate buried waste from environmental dispersion for at least 1,000 years. The Hanford Site Surface Barrier Development Program was initiated to design and test a soil cover system (barrier) that could be used to inhibit water infiltration, plant and animal intrusion, and wind and water erosion (Link *et al.* 1995).

The cover system consists of several layers of different soil materials such as fine soil, sand, gravel, riprap, and asphalt to optimize performance and longevity. A typical cover system is shown in Figure 4.4-19. The top vegetated fine soil layer acts as a medium in which moisture is stored until the process of evaporation and transpiration return the water to the atmosphere. The coarse materials (e.g., sand, gravel, and riprap) below the fine soil layer provide a capillary break that restricts biointrusion. The asphalt concrete layer below the coarse materials is designed to divert any water that percolates through the capillary barrier.

Figure 4.4-19 **Cover System used at Hanford Site**



Source:MEND 2.21.3a

The effectiveness of the soil cover was studied using weighing lysimeters. The results indicated that the capillary barrier in conjunction with the use of vegetation to promote transpiration worked well to prevent deep infiltration.

The effect of surface conditions on soil water storage was also studied with the use of small tube lysimeters. It was found that vegetation caused a greater decrease in storage than non-vegetated plots. Link *et al.* (1995) have concluded that an admix surface (gravel mixed into the fine soil) with vegetation would minimize the chance of wind erosion, while not significantly affecting the storage capability of the cover.

The program was extended to include building of a full-scale prototype soil cover system with a surface area of about 200 hectares. The monitoring plan consisted of the measurement of, runoff of water-sediment mixtures, water infiltration, creep, moisture content, and surface cracking (Petersen *et al.* 1995). The construction cost of the system is estimated to be $300,000 (US) per acre, which is approximately $100 (CND) per m^2. This cost makes such a system unrealistic for use in the mining industry. However, the results of the study are extremely useful for predicting long-term performance of dry cover systems typically found in the mining industry, in particular with respect to biointrusion and capillary barriers.

4.4.4.3 Summary and Discussion

A key point in summarizing the case studies is the relatively short-term monitoring and history of detailed knowledge regarding the application of dry covers for mitigating AMD, in comparison to the long-term environmental liability associated with AMD. This is a result of the short time frame in which AMD has been recognized as one of, if not the major environmental issue facing the mining industry. In general, the issues associated with AMD have only been addressed during the past 20 to 30 years. Dry covers as a method for mitigating AMD have been properly evaluated during the past approximately 20 years. In addition, field case studies with monitoring systems in place to properly evaluate cover system performance have only existed for approximately 10 years. For example, the Equity Silver Mine till cover system was one of the first full-scale engineered cover systems implemented in Canada for a large surface area waste rock pile, and it occurred in the early 1990’s. The consequence of this comparatively short time frame is that it is often difficult to demonstrate the effectiveness of engineered dry cover systems as a long-term closure solution to all interested parties.

The approach to address this challenge lies in continued, and indeed increased, monitoring of dry cover system performance, as well as in the development of improved scientific physically based performance prediction tools (i.e. numerical models). In addition, progressive decommissioning of a potentially acid generating waste (i.e. application of a cover system) during the life of the mine is clearly the optimal closure solution, for a wide-ranging rationale. First and foremost, the funds required for closure are more readily available and closure costs can be spread over the life of the mine rather than concentrated after cessation of operation. Numerous case studies demonstrate that wherever possible, waste rock pile and tailings impoundment construction should be thoroughly conducted with closure in mind. Rehabilitation and decommissioning should be an integral, progressive, and scheduled component of the mining operation, rather than left as a task required after mine operations are completed. Monitoring of the progressive closure efforts, as soon as possible following commencement of mining, will provide information for developing the most technically feasible and economic closure solution.

The second rationale for progressive closure is that remnant oxygen and water within the waste material can be reduced. Therefore, sulphide oxidation can be minimized, as compared to allowing uncontrolled atmospheric oxygen and water entry during the operation of the mine. Minimizing oxygen ingress as well as oxygen movement within the waste, either through advective, convective, or diffusive transport, is likely the key aspect of progressive decommissioning. The objective would be to ensure that pore gas introduced during placement of the waste is the only significant source of internal oxygen. Reducing internal remnant oxygen and water during operation will significantly reduce the residual acidity and metals that are typically associated with waste piles following placement of a dry cover system.

The Millenbach closure (Woyshner *et al.* 1997) is an excellent example. Waste rock (17,500 m^3) was relocated to a tailings dam (0.66 ha) and a multi-layered cover system (with capillary breaks) was constructed over the 1.35 ha surface of the tailings and waste rock. Long-term water quality will be controlled by the new rate of oxidation regulated by the dry cover system. However, water quality is in a transition, or stabilizing period as residual acidity and metals in the pore water flush out and/or precipitate in the tailings and waste rock. The residual acidity and metals are a direct result of allowing the materials to be exposed to atmospheric conditions. The length of flushing and dissolution period was estimated by Woyshner *et al.* (1997) to be 20 to 30 years. This is a significant time given the small volume of waste material.

Wilson *et al.* (1997) used numerical tools to predict that 15 to 20 years of declining seepage would occur from the Equity Silver Mine main pile (52 Mt, 41 ha) before residual levels of seepage were reached following placement of the dry cover system. Another example is the covered waste rock pile at the Heath Steele Mines site (Pile 7/12), where estimates put the natural drain-down of one pile pore volume at about 30 years, and many pore volumes are needed to flush oxidation products that precipitated prior to cover placement (MEND 2.31.1c).

In general, a dry cover system designed to limit oxygen and water will only prevent further contribution to AMD (this can be a significant amount depending on site conditions). In other words, the flushing of remnant acidity and metals, albeit at progressively smaller seepage rates but possibly higher concentrations, will inevitably occur even if an “ideal” cover system is constructed. Therefore, provision for progressive decommissioning during operation will significantly reduce the remnant acidity and metals, and hence the inevitable flushing, and ultimately the need (quantity and time) for collection and treatment following closure.

It is fundamental to understand that short-term monitoring (i.e. 1 to 10 years), whether during progressive decommissioning or after cessation of operations, is simply a brief "snapshot" in time and should not be taken as being indicative of long-term performance. The improved understanding in performance of dry cover AMD control cover system trials is the direct benefit of a field performance monitoring program. Key factors controlling performance will be developed and understood. The database required for field calibration of a robust numerical predictive model can be developed. Accurate, and more importantly, defensible predictions for long-term performance of the AMD control dry cover system can then be obtained using a predictive tool properly calibrated to field conditions.

In summary, the case studies clearly demonstrate the potential of dry covers as a long-term AMD control method. The appropriate dry cover is site-specific and dependent on potential cover materials at the site as well as site climate conditions, as opposed to regional climate conditions (i.e. an automated meteorological station should be installed at the waste storage facility as soon as possible following commencement of operations, or ideally during the exploration phase of the project). The successful application of a dry cover system as a closure measure is fundamentally linked to continued field performance monitoring, as well as development of defensible and robust predictive modelling tools. It is only in this manner that an operator can significantly reduce speculation of long-term dry cover system performance as an AMD mitigation closure method.

4.4.5 SUSTAINABLE PERFORMANCE OF DRY COVER SYSTEMS

Longevity of dry cover systems is a function of many factors that tend to alter the initial and final structure of the cover system materials. These factors are identified in Figure 4.4-20 and include physical, biological, and chemical processes. Chemical processes are more applicable to liner design rather than cover systems as covers seldom have to defend against chemical attack. The most important factors affecting cover performance are physical and biological processes.

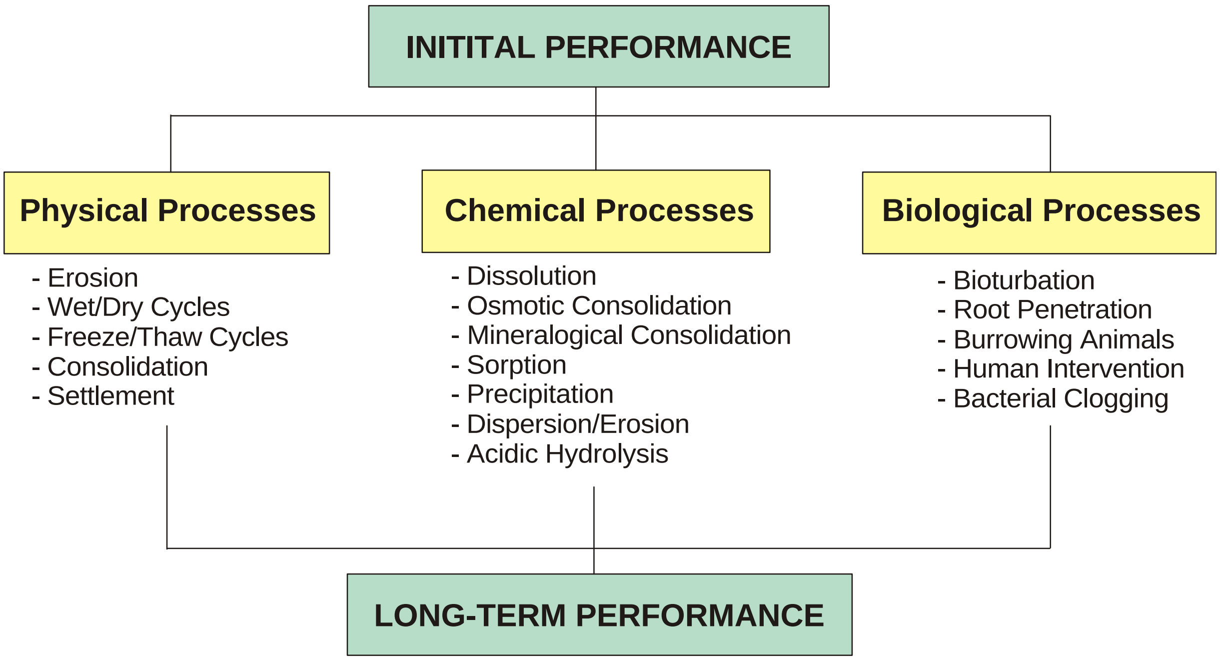
The physical process of erosion can destroy a cover system by either washing away the fine fraction or reducing the thickness of the cover. This has a damaging effect on the hydraulic conductivity and the moisture retention capacity of a cover. The impact from water and/or wind erosion can be minimized by selecting an appropriate material for the uppermost cover layer (i.e. a well-graded, coarse textured material with minimal silt and clay size particles) and by establishing a vegetative growth as soon as possible following cover placement. In addition, the proper design and construction of a surface water management system will reduce the effects of erosion, and minimize the need to repair or stabilize the cover system in the future (Aziz 2000). Poor surface water management and landform instability are the most common factors, if not the most common factor, leading to failure of cover systems around the world. The primary problem is attempting to build engineered structures that "fight" natural processes, as opposed to engineered systems that are natural analogues and become part of the surrounding ecosystem as soon as possible following implementation.

Wet/dry cycles will impact the moisture retention and permeability of a compacted layer designed to be a hydraulic barrier, and/or an oxygen ingress barrier. Compacted clay covers have a high potential for drying and cracking in arid climates. The effective permeability of the cover system will increase as water bypasses the soil matrix and flows through shrinkage cracks (Bronswijk 1991). Potential exists for cracking to occur in arid climates even with a 45 cm layer of protective soil cover overlying the compacted layer (Daniel and Wu 1993). It is common for one to think of Canadian mine sites as being located in "wet" climates and discount the potential of drying and cracking of the compacted layer. However, the reality is that numerous mine sites in Canada are located where hot, dry summers are typical. Hence, the potential for drying and cracking exists unless the cover system design properly addresses this factor that will impact in the longevity of the dry cover system.

Freeze/thaw cycles will also impact the permeability and moisture retention of a compacted layer of a dry cover system. Unfortunately, most Canadian mine sites are subjected to freezing temperatures and as a result the dry cover system design should address this inevitable impact on long-term performance. The greatest impact in performance will occur after the first freeze/thaw cycle although potential exists for a compacted layer to continue to decrease in performance during subsequent freeze/thaw cycles.

The functionality of each layer of the cover system has the potential to diminish through the workings of flora and fauna. Bioturbation is an essential, unavoidable impact on the longevity of a dry cover system and in addition to root penetration should be a key component of design. Bioturbation is the mixing of soil by animals but can also include arboturbation (tree heave) and "stomping" by wind moving trees. All bioturbation is regionally different and site-specific as a result of being influenced by local ecosystems (Heinze *et al.* 1998). Therefore, an acceptable balance in dry cover system design with respect to abiotic and biotic influences must be addressed to compensate for the potential decrease in technical performance. It is necessary to design and install the cover system corresponding to the future use of the area and potential natural succession of flora and fauna.

Figure 4.4-20 **Potential Factors Controlling Dry Cover System Performance**



Source: Haug 1993

A panel discussion was held at a BC/MEND 2000 workshop in Vancouver in November 2000 (MEND BC.03) on the design features, monitoring, and resources required to maintain the performance of a dry cover system in the long term. The two main topics of discussion focused on the design life of dry covers and long-term monitoring and maintenance. Members of the panel discussion stated that a 1000-year dry cover design life is conceivable because man-made structures constructed over 1000 years ago still exist. A consensus was not reached by the panel regarding an appropriate design life for dry cover systems; however, it was noted that the Canadian Nuclear Safety Commission (CNSC) requires that discharges from a uranium waste storage facility be predicted over a 10,000 year period. Normandy Mining Ltd. (Australia's largest gold producer) have developed closure standards at several of their Australian operations that specify the containment structure must be designed to maintain physical stability for a 200 to 500 year time frame.

The panel agreed that dry covers should not be viewed as a walk-away solution, but rather as a control measure for minimizing the impacts from AMD. Therefore, a key issue with respect to maintenance in the long term was whether personnel would be available to conduct the maintenance, as opposed to ensuring that adequate financial assurance would be in place to cover the maintenance costs.

Clearly there is potential for any dry cover system to "fail" and allow contaminated seepage to enter the natural environment. Poorly designed dry cover systems can fail over a 10 to 50 year period, or even up to 100 years after construction. Extending the life of the dry cover system through proper design is possible by taking into account the factors that affect long-term performance. This effort will provide a significant positive impact on the net present value of any contingency plan required for failure of the dry cover system. The key is to prevent the dry cover system from failing in the short term. Rather, the objective should be for the dry cover system to "fail" over geologic time, augmented by minimal maintenance, such that the natural environment is capable of accepting the incremental "failure".

4.4.6 COST

The unit cost for construction of a dry cover system is extremely site-specific. It is difficult to develop an average value because of the impact of the various components that influence cost. For example, the Equity Silver Mine cover system (0.5 m compacted till and 0.3 m non-compacted till) was constructed for approximately $35,000 (CAD) per ha (including reshaping, etc.) (Aziz and Ferguson 1997). The borrow area for the till cover material was immediately adjacent to the waste rock piles at the Equity site and large rubber tired scrapers were used to pick up and place the till. Hence, the cost for construction of the cover system was minimal when coupled with the sound site construction management that occurred. A similar cover system at another site might cost twice or three times more per unit area if the cover material borrow area was at a greater distance and a truck and shovel/backhoe construction technique was used. The Rum Jungle waste rock cover system in Australia (0.225 m compacted layer, 0.25 m moisture retention zone, and 0.15 m erosion resistant zone) was constructed over 51 ha area for approximately $67,000 (AUD) per ha (including reshaping, etc.).

Tailings ponds at the Cominco Ltd. Kimberley Operations occupy 373 ha of land (Gardiner *et al.* 1997). A 0.45 m layer of non-compacted till was estimated in 1996 to cost between $35,000 and $40,000 (CAD) per ha to cover a 42 ha gypsum pond. The borrow area for this material was nearby the tailings but not immediately adjacent. It was estimated that construction of a 0.25 m layer of compacted till (screened to remove stones greater than 5 cm) and overlain by 0.30 m of non-compacted till would cost approximately $60,000 (CAD) per ha (Gardiner *et al.* 1997). It was anticipated that the additional material resulting from a thicker cover system and screening would require the development of a borrow area further from the tailings impoundment, which led to the higher cost per hectare. A comparison of these costs to that of the Equity Silver Mine is a good example of the impact of the distance to the cover material borrow area and “treatment” (i.e. screening) had on the unit cost of construction.

Harries (1997) stated that installation of dry cover systems after mining operations have ceased would likely cost between $50,000 and $100,000 (AUD) per ha. Installation costs at historic sites would likely exceed $100,000 (AUD) per ha (Harries 1997). These costs make a strong case for progressive decommissioning during the life of the mine operation.

The capillary barrier field test plots at Waite Amulet and Heath Steele were estimated to cost between $200,000 and $300,000 (CAD) per ha (MEND 2.21.3a). These costs were expected to significantly decrease for construction of a full-scale capillary barrier cover system. In fact, the final cost of the Les Terraines Aurifèrers composite cover system was $65,000 (CAD) per ha for a 60 ha tailings area, excluding revegetation costs (MEND 2.22.4a). It should be noted that the borrow material for the compacted material and the capillary barrier materials were immediately adjacent to the tailings impoundment that was covered. The preliminary estimate of all reclamation costs (including ditches, foot drains, spillways, etc.) was approximately $93,000 (CAD) per ha (MEND 2.22.4a).

A component of MEND 2.22.2c was the development of a relatively simple model for evaluating the cost of different closure techniques. The model, called Evaluation of the Cost for Reclamation (ECR), allows a relative comparison of the costs incurred by the application of various techniques, namely:

- Chemical treatment (lime) of acidic drainage;
- Use of a water cover with impervious dams;

- Construction of a multi-layer dry cover system with a capillary barrier made with soils or clean tailings; and
- The possible inclusion of a desulphurization process to produce non-acid generating tailings that could be used as the moisture retention layer in a cover system.

The model aims at providing the reader with a simple tool to obtain preliminary estimates of various techniques and to help select the most promising technology for further site-specific investigations. A more detailed description of the spreadsheet based model, along with examples, are provided in MEND 2.22.2c.

Hydraulic placement of cover material is a possibility and an opportunity to reduce construction costs at some sites. This method of construction would take advantage of a milling complex and associated infrastructure (e.g. pumps, pipelines, etc.) already existing at a site. In general, cover materials that would be hydraulically placed over a waste impoundment would be “clean” tailings, which can be obtained from milling sulphide-free ore or by using a desulphurization process (Bussière *et al.* 1997). One such application is at Kennecott’s Ridgeway mine in South Carolina where a clay cap was placed hydraulically over the 320-acre non-acid generating tailings impounment.

4.4.7 MEND AND RELEVANT PUBLICATIONS

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MEND 2.21.2	1993. Field Evaluation of the Effectiveness of Engineered Soil Covers for Reactive Tailings. October.
MEND 2.21.3a	1997. Review of Soil Cover Technologies for Acid Mine Drainage – A Peer Review of the Waite Amulet and Heath Steele Soil Covers. July.
MEND 2.21.3b	2000. Peer Review on Covers Made With Non-Traditional and Solid Waste Compost. Not available at time of publishing.
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4.5 CO-DISPOSAL OF WASTE ROCK AND TAILINGS

4.5.1 DISCUSSION OF THEORY

Waste rock dumps pose a particular challenge in terms of controlling acidic drainage. The traditional approach in disposing waste rock is to maximize the storage volume for the surface area available by maximizing the height of the piles. This creates ideal conditions for sulphide oxidation and acid generation, as the high permeability of the waste rock favours movement of air and water through the pile. Options available for managing acid generation can include flooding, engineered covers and treatment.

The co-disposal concept combines the waste rock and tailings into one waste management facility. The potential advantages may include: a reduced volume of waste as the tailings may occupy void volume in the waste rock; reduced oxygen availability by maintaining saturated or near saturated conditions surrounding the reactive waste rock; and reduced infiltration and seepage production from the waste due to reduced permeability, more surface runoff and higher water retention capacity to enhance evaporative losses.

4.5.2 DISCUSSION OF MEND RESEARCH

One project on co-disposal of tailings and waste rock in test cells was completed in MEND at Les Mines Selbaie (MEND 2.32.2). The objective was to evaluate the concept of co-disposing reactive sulphide waste rock with non-reactive tailings in basins to minimize acidic drainage. The design objective was to create saturated conditions in the reactive waste and minimize groundwater flow in the basin. Phase (a) consisted of field reconnaissance, mineralogical and petrological characterization of both tailings and waste rock and potential basin construction materials, and a preliminary geotechnical design of the basin. The prediction tests carried out on the tailings materials indicated that the tailings were likely to generate acid. The project was based on the concept of co-disposing acid-generating waste rock with neutral tailings. Since the tailings were likely to be acid producers, the final design of the basin was changed to rely on low permeability dykes to maintain saturated conditions in the waste rock and tailings. Phase (b), the construction of the cells to test the co-disposal concept, was cancelled (see section III for more information).

MEND also sponsored a study to evaluate the feasibility of injecting mine tailings into existing reactive waste piles as a method to control acidic drainage (MEND 2.32.3a). A literature review of the various injection techniques and key design parameters was completed. A research program was also proposed to study injection of tailings into waste rock piles and to assess potential benefits of creating conditions similar to those obtained by using co-disposal. This program was well received but too expensive to proceed to the field and was not funded.

4.5.3 APPLICATIONS AND LIMITATIONS

Co-disposal can potentially be applied to any mine site with reactive wastes. Applications may include:

- Consolidation of tailings and waste rock to minimize footprint;
- Elimination of the need for tailings dams or separate waste dumps;
- Creation of an oxygen and or infiltration barrier (as a cover or complete co-mingled deposit); and
- Creation of elevated water tables and/or saturated conditions to inhibit oxidation and reduce infiltration (co-mingled deposit).

The success or failure of the concept would depend upon: potential ARD issues (i.e. if acidic tailings were added to waste rock could ARD conditions be worsened); physical stability of the deposit (i.e. would tailings added to waste rock reduce strength or result in higher losses of erodable fines); chemical stability issues and possible concerns of incorporating milling chemicals into waste rock piles; method of incorporation (blended, layered, processed, etc.); and long-term stability and performance of the co-mingled mass.

For regions where seismic risks are greater, co-disposal offers the potential benefit of maintaining a co-mingled mass that is unsaturated and not susceptible to liquefaction. The challenge at this time is to demonstrate how this mass would be placed and confirm that the co-mingled deposit will retain the properties over the long term.

Co-disposal of tailings into a waste rock matrix to create a man-made cover material has not been attempted, although it offers great promise, especially in regions short of suitable cover materials. Preliminary work by Placer Dome has shown that waste rock can be amended with tailings and other mining wastes (slag and crushed waste rock) to create materials with excellent physical properties. The challenge is to demonstrate that these materials can be economically and practicably formed and applied without creating additional environmental problems (process reagent loss, erosion of fines, stability problems, etc.).

4.5.3.1 The Canadian Experience

Co-disposal is an uncommon practice in Canada. There are, however, many places in Canada where co-disposal has been incidental. This is especially true for underground mining operations where waste rock quantities are small and waste rock was often used for dyke construction and/ or simply disposed in the basins rather than creating a separate repository (e.g. Quirke, Panel and Stanleigh Mines in Elliot Lake). Co-disposal was also proposed for acid waste rock from Voisey’s Bay and Les Mines Selbaie. The one area where true co-disposal was practiced as a closure method was at the former Agnew Lake uranium mine in Ontario. Sludges from water treatment have also been co-disposed within waste rock piles. Examples of selected practices proposed or used in Canada are reviewed below.

1. <u>Les Mines Selbaie</u>

The concept for co-disposal of waste rock with tailings in test cells involved the construction of shallow basins, the placement of highly reactive waste rock in the bottom of the basins, and infilling with alternate layers of less reactive waste rock and non-reactive (or desulphurized) tailings to achieve the most intimate mixing of the tailings as possible. However, because of the difficulties in placing layers and infilling rock voids, the result was a segregated rather than a truly co-mingled deposit. A top layer of non-reactive (or desulphurized) tailings was proposed to complete waste placement in the basins.

The engineering design of the basins was to:

- Maximize the height of the water table in the basins to create saturated conditions in the reactive wastes; and
- Minimize groundwater flow through the basins. This would be accomplished by the use of drainage and low permeability layers placed strategically below (as basin liners) and above (as basin covers) the mine waste to control water movement.

Following closure of the basin, the final steps of this disposal concept would include the placement of top soil and vegetation. This would serve to protect the engineered basin cover from erosion and to improve the aesthetic appearance of the waste management facility.

The main problem with this concept is that co-mingling of the layers was not achieved and the rock layers acted as drains, thus greatly reducing saturation and the effectiveness of the scheme.

2. <u>Rio Algom Quirke Waste Management Area</u>

Waste rock was used in the Quirke waste management area to create tailings cells. Many of these cells were subsequently submerged by tailings beaches. At closure, new dykes were constructed and a terraced flooded tailings basin was created. The flooding plan has been effective in controlling acid production from both the reactive waste and tailings, however, the presence of the buried waste rock dykes created surface drains. These greatly increased the subsurface drainage to the point where at least one cell could not be kept flooded without intervention. This problem was overcome by excavating to the submerged dykes and sealing off the preferential flow paths.

3. Agnew Lake

The Agnew Lake uranium mine employed a combination of underground leaching and surface heap leaching to recover uranium. Ore was blasted and leached in place with the excess (the swell from blasting) brought to surface and leached in a surface heap. Tailings from the operation were precipitates formed from raffinate neutralization. The tailings solids were primarily gypsum and iron hydroxides, which were disposed of in a surface repository. The mine operator at closure was left with an acid generating leach dump and a viscous sludge deposit. Separately these deposits presented long-term liabilities. Testing was carried out to determine if there was merit in combining the wastes. The sludge had a high moisture content and high NP, while the rock was structurally strong but could produce acidic drainage. Testing confirmed that when rock was placed into the tailings, the tailings would infill the voids and allow free water to drain creating a low permeability mix with cementitious properties. The combined mixture was extremely stable, non acid generating and, as a co-disposed mixture, retained all the beneficial properties and none of the detrimental properties of the individual waste streams. The leached ore was deposited into the tailings and the mass monitored for a period of 5 years, after which the property was returned to the crown.

4.5.3.2 International Practice

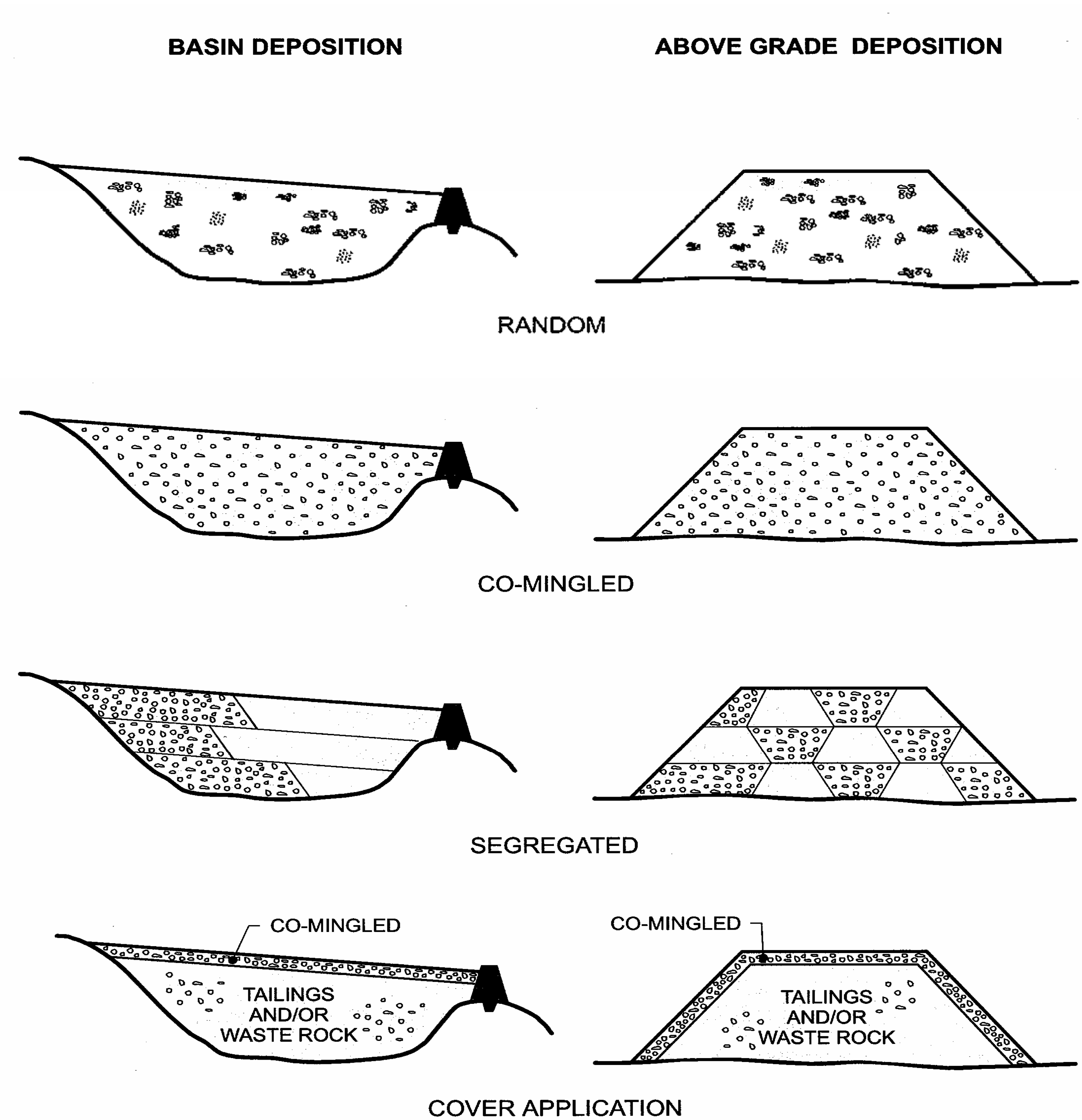
The International Network for Acid Prevention (INAP) retained Klohn Crippen (Klohn Crippen 2001) of Vancouver to complete a literature search of co-disposal practices. A presentation of the findings was completed at an INAP sponsored workshop on co-disposal in Vancouver, November 2000. They indicated that their desktop survey found few examples of co-disposal applications in the mining industry. The most extensive use of co-disposal is in the coal industry where coal tailings fines and coarse rock rejects have been co-disposed in Australia for more than 10 years. In these cases, co-disposal is practiced by slurrying the co-mingled streams and pumping to a conventional tailings repository (pit or tailings basin). Additional details can be found in the INAP report, which should be available in 2001.

4.5.4 BASE METHOD

There are four basic concepts available for co-disposal: random deposition, segregated deposition, co-mingling and cover application as shown on Figure 4.5.1.

With random deposition, waste and tailings rock are simply placed in the disposal area without an intended purpose other than minimizing the footprint (disposal). For above-grade disposal, tailings would likely have to be dewatered unless perimeter dykes were constructed (per Selbaie test cells) or tailings quantities were small enough that the moisture and solids are retained within the waste rock matrix.

Figure 4.5-1 **Concepts for Co-Disposal of Waste Rock and Tailings**



With segregated deposition, rock is disposed in a separate area; this may be in segregated areas, in waste rock dykes used to create tailings cells or in layers or zones within the basin or waste rock pile.

With random or segregated disposal, the physical properties of the waste rock zones within the disposal areas are quite different; typically these areas will have high permeability and form preferential flow paths for drainage. For a basin that will be flooded, covered or treated at closure the primary advantage for co-disposal is simply the consolidation of the waste into one basin. Some additional advantages could also occur in reducing oxygen access to the waste rock if this material was buried below tailings.

Co-mingling has significant benefits as a management option for reactive waste. With co-mingling the objective is to create as near to a homogeneous deposit as possible. This will minimize preferential flow paths which will lower water tables in deposits where elevated water tables are proposed and will reduce void ratios and inhibit oxygen entry to waste piles placed above the water table. The volume reduction may also be significant. For example, a 20 million tonne gold ore deposit with 1:1 ore to waste ratio would require a management area of about 10.5 million m^3 for waste rock (1.9 t/m^3) and 14.3 million m^3 for tailings (1.4 t/m^3), or a total volume of 24.8 million m^3. For a co-mingled co-disposal, the volume required would be reduced to about 21.7 million m^3 or a reduction of about 15 %. For a large open pit with high waste to ore ratio, tailings could be readily incorporated into the pile with no increase in volume.

Co-mingling can be achieved by the blending of dry tailings with waste rock or by pumping tailings slurry or paste into the voids of a waste rock pile. In either case the costs may be high and there is considerable uncertainty as to the effectiveness of such schemes. The prime advantages of such a scheme is to reduce infiltration and effectively eliminate convective air flow into the waste pile. Other than cost, the other major drawback is that it is highly unlikely that waste rock and tailings production would be evenly balanced between storage capacity within the waste rock voids and tailings production.

Co-mingled waste has great potential as a cover material, especially in regions where suitable cover materials are scarce.

4.5.5 VARIATIONS

Historically in Canada, co-disposal was practised not for control of reactive waste rock but rather as a simple and cost-effective management plan, especially where waste production rates were small (i.e. only 1 disposal site to manage). Co-disposal has also been practised and proposed for other mining wastes, including effluent treatment sludges and slag. The acceptability of these practices will depend in large part on the characteristics of the wastes and site-specific characteristics of the waste management area. Tailings placement into the voids could be considered as a remedial measure for an existing waste pile; however, this has never been attempted on a large scale.

4.5.6 COST

The costs for co-disposal will depend on many factors including: relative quantity of waste rock, transport distances, ease of co-disposal, the requirements for co-mingling and many site specific factors (i.e., the ease of expanding the tailings basin capacity to accommodate waste rock). The greatest potential cost savings for co-disposal are likely associated with closure, as only 1 area will require reclamation and the properties of the deposit may be greatly enhanced, minimizing the need for sophisticated closure strategies.

4.5.7 MEND AND RELEVANT PUBLICATIONS

MEND 2.32.2 1992. Bassin de co-déposition se stériles st de résidus : Les Mines Selbaie, Joutel, Québec.

MEND 2.32.3a 1994. Injection de résidus miniers dans des stériles miniers comme moyen de réduction des effluents acides. janvier.

Klohn-Crippen Consultants Ltd. 2000. Literature Survey, Co-Disposal of Mine Tailings and Waste Rock, Final Report. Submitted to Porgera Joint Venture. 58 pages. November.

Klohn-Crippen Consultants 2001. Application of Co-Disposed Mine Tailings and Waste Rock to Construct Covers. Submitted to INAP.

4.6 BLENDING AND LAYERING OF WASTE ROCK

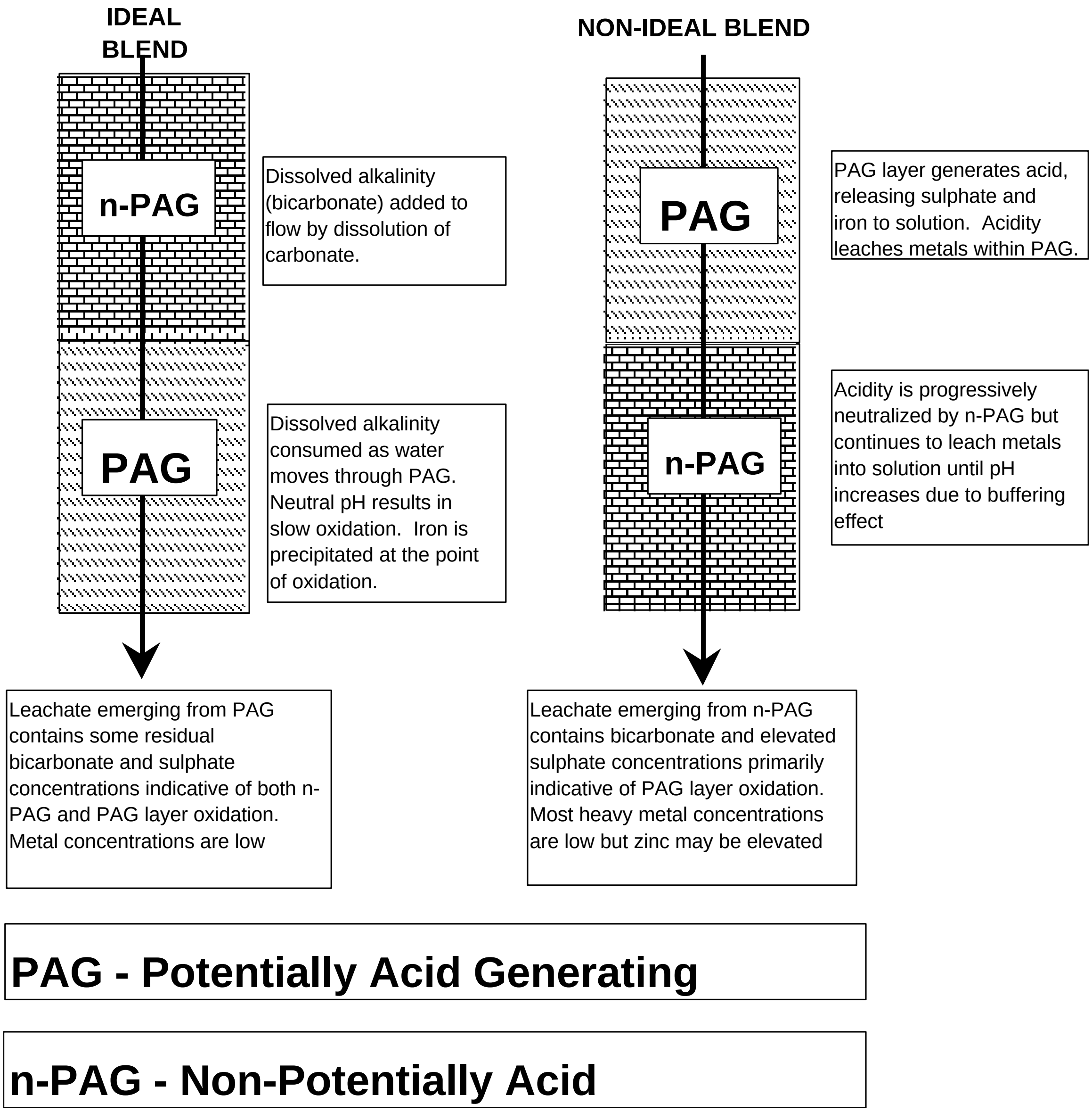
4.6.1 DISCUSSION OF THEORY

Blending can be defined as the mixing of at least two rock waste types with varying acid generation potential, neutralization potential and metal content to produce a pile that has seepage water quality acceptable for discharge without additional measures (Mehling *et al.* 1997). There are varying degrees of blending. An "ideal" blend refers to a mixture in which the resulting leachate is comparable to that produced by grain-by-grain mixing (Day and Hockley 1998). Grain-by-grain mixing in theory may not be required to produce ideal behaviour. A "non-ideal" blend occurs when the leachate is non-acidic but seepage chemistry indicates that rapid heavy element leaching from one or more rock types is influencing drainage quality. Finally, the term "non-blend" is used to describe a waste rock dump in which the materials balance indicates that drainage quality could have been mitigated by blending but the materials were not appropriately placed.

The concept of blending depends primarily on how sources of alkalinity are positioned relative to sources of acidity and heavy metals. In order to develop the concept, the geochemical processes controlling acid and heavy element release are reviewed below.

In mine wastes, acid generation occurs when iron sulphide minerals are oxidized by oxygen in the presence of water. However, the reaction is well known to be catalyzed by various iron and sulphur oxidizing bacteria, which accelerate the oxidation of ferrous iron to ferric iron. These bacteria increase in activity as pH decreases, reaching optimal conditions at pH < 4 depending on the different types of bacteria involved (Nicholson 1994; Gould *et al.* 1994). In wastes containing any calcium or magnesium carbonate minerals, initial pHs are greater than 7 due to active dissolution of the carbonates. Under these conditions, the sulphur and iron oxidizing bacteria are relatively inactive and oxidation proceeds abiotically compared to lower pHs when carbonate minerals are not available. Abiotic oxidation of pyrite occurs at fairly constant rates regardless of pH conditions (Nicholson 1994). The only way for alkaline conditions to be maintained at greater than pH 7 near actively oxidizing iron sulphide minerals is for the alkalinity to be available in dissolved form in adjacent pore waters. Therefore, the likelihood of alkaline conditions being maintained around oxidizing iron sulphide mineral grains depends on the quantity of alkalinity arriving in solution and the amount of oxidation occurring along a flow path until the next source of alkalinity "recharge" is reached. This is illustrated schematically in Figure 4.6-1. As dissolved alkalinity encounters a zone with no inherent alkalinity of its own; the dissolved alkalinity is consumed by contact with actively oxidizing iron sulphide mineral grains. If the water emerges from the non-alkaline material with alkalinity still remaining in solution, then the material has not produced acidic drainage. This process defines the "ideal blend" (Figure 4.6-1).

Figure 4.6-1 **Illustration of Ideal and Non-Ideal Blending Behaviour**



In contrast, if dissolved alkalinity is depleted within the non-alkaline material, the generation of acidic drainage will begin. This allows pH to decrease in the material and results in acceleration of bacterially driven oxidation. The resulting pore water has corrosive oxidizing properties, which allows it to dissolve nearby minerals. If these minerals contain heavy elements, the elements will be released to solution. As the leachate emerges from the non-alkaline material, it contains the evidence of rapid oxidation under acidic conditions (low pH, elevated sulphate, acidity and dissolved light and heavy elements). This water may then contact alkaline materials causing pH to rise, and lowering the concentrations of heavy elements in solution. If sufficient alkalinity is present along the flow path, the pH of the leachate may rise to near 7. Concentrations of elements such as aluminum, manganese, iron and copper can be decreased by several orders-of-magnitude as the solution moves towards neutral pH. However, zinc and elements forming oxyanions (arsenic, antimony, selenium, uranium, molybdenum) can persist in neutral pH solutions. This complete neutralization by contact with solid alkaline materials defines the “non-ideal” blend (Figure 4.6-1).

Finally, if the acidic leachate is not completely neutralized in the alkaline material but then enters a downstream zone of non-alkaline material it promotes accelerated oxidation in that material and adds further dissolved load. This scenario is referred to as a “non-blend” because the mixture contains alkaline material but not in a configuration that allows acidity to be consumed internally. A mitigating factor that is not a function of the rock dump itself is that the acidic leachate may subsequently mix with alkaline water leached from elsewhere in the dump. If this occurs, the results would be comparable to the non-ideal blend.

It is apparent from the above discussion that the transition from ideal to non-ideal to non-blend is controlled by certain distances (Day and Hockley 1998). Ideal behaviour will occur when the acid generating zones have a length (L_{acid}) less than a maximum length (L_i) defined by:

$$L_{acid} < L_i \alpha Q (a_{alkalinity}/R_{FeS\text{-}Ox,alk})$$

where:

Q = the incoming water flow rate;

$a_{alkalinity}$ = the chemical activity of alkalinity in the water flow in the non-alkaline layer (including alkalinity contributed by the nearby alkaline layers and the non-alkaline layer itself); and

$R_{FeS\text{-}Ox,alk}$ = the rate of iron sulphide oxidation under alkaline conditions in the non-alkaline layer.

L_i is proportional to the flow and alkalinity activity (i.e. more flow or alkalinity delivers more alkaline load) but inversely proportional to $R_{FeS\text{-}Ox,alk}$ (i.e. higher rates indicate shorter distances over which alkaline conditions can be maintained for a given alkalinity).

For non-ideal blending, the alkaline zones must have a length ($L_{alkaline}$) greater than a certain minimum length (L_n) defined by:

$$L_{alkaline} > L_n \propto \Sigma A_{FeS\text{-}Ox,acid}/C_{alkalinity}$$

where:

$\Sigma A_{FeS\text{-}Ox,acid}$ = the total acidity produced by oxidation in the non-alkaline layers; and

$C_{alkalinity}$ = the net concentration of available solid alkalinity in the alkaline material, after allowing for acid generation potential in this material.

L_n is proportional to the total acidity produced because a greater flow length is required to neutralize the acidity produced, but inversely proportional to the solid alkalinity concentration because a greater alkalinity concentration decreases the required flow path length for a given rate of oxidation. This indicates that the amount of solid alkalinity along a flow path must exceed all the acidity produced by itself and upstream sources. The available alkalinity may decrease if coating or sealing of the acid consuming waste occurs.

Waste rock dumps with low acidity and metal concentrations in their drainage but containing sulphide material (i.e. potentially acid generating at some scale) illustrate that ideal blending can occur. In these cases, L_i may be greater than grain scale (i.e. L_{acid} is less than several millimetres). Also waste rock dumps containing some acid consuming material produce acidic drainage, probably because L_{acid} is at least several metres. Critical values of L_{acid} therefore probably lie between the scale of millimetres and metres, depending on the site.

4.6.2 DISCUSSION OF RESEARCH

MEND 2.37.1 reviewed the current understanding of blending and layering of waste rock to delay, mitigate or prevent acid generation. It was found that there are very few well-documented examples of waste rock blending. Coal mines in the Appalachian region of the eastern USA have used waste rock blending, however, it was found that other control measures were often applied and these prevented evaluation of the success of blending. Many examples of older hard rock mine waste rock dumps with non-acidic drainage are known, but the source characteristics are not adequately understood. Recently, waste rock dumps have been characterized during construction, but the monitoring records are not yet sufficient to evaluate blending. A significant issue is the question of how long the drainage must be monitored before blending can be judged successful.

It was noted that based on the available information, blending did not reduce sulphide oxidation rates in the potentially acid material unless the neutralizing material was highly reactive, or blending was near ideal. Blending and layering were effective in delaying the onset of acid generation and the tenor of the acidic drainage was reduced. Prevention of acidic drainage did not necessarily prevent dissolved metals from being problematic.

MEND 1.19.1 reviewed column test data on mixing of limestone with potentially acid generating rock. The amounts of limestone added were varied up to the stochiometric amount (6.6%) required to neutralize the acidity produced by oxidation of all the sulphide. Except for the high limestone column, all other columns generated acid. Increasing sulphide oxidation was apparent in the high limestone column before it was shut down.

MEND 2.37.3 compared six column tests and waste rock dump monitoring for the Samatosum project in central British Columbia. The waste rock columns had waste rock layers 0.2 m thick and average NP/AP of 1. Three of the tests operated for more than four years. The waste rock dump contained 6 m layers with an average NP/AP ratio of 3. Five of the columns did not generate acid, but the waste rock dump had a seasonally acidic seep. A control column containing only potentially acid generating rock generated acid after two years. Based on the presence of low paste pHs in the potentially acid generating layers of layered columns, it was concluded that the rates of oxidation in the acid generating layers of the column tests were independent of the presence of the alkaline layers, in other words, the alkaline layers did not reduce the oxidation rates in the acid generating layers.

The presence of an acidic seep from the waste rock dump was believed to be due to physical conditions (i.e. preferential flow through acid generating rock) rather than a failure of geochemical principles.

Ghomshei *et al.* (1997) described construction of the layered waste rock pile at Samatosum and drainage monitoring data. The waste rock contains two 6-m layers of potentially acid rock in carbonate-bearing ultramafic rocks. The total mass of waste rock is 9.6 million tonnes of which 1.5 million is believed to be potentially acid generating (see also MEND 2.37.3 described above). The rock was not characterized during operation to confirm geochemical composition of the rock, nor was rock surface area and preferential segregation of reactive minerals into the fine fraction. Overall drainage from the rock was alkaline to the end of 1996 (with the exception of one acidic seep), but seasonal peaks of sulphate, zinc and manganese, and the mass of sulphate discharged appeared to increase from less than 200 t/a to greater than 400 t/a over five years of monitoring.

Rescan (1992) constructed 20-tonne field test plots with layered potentially acid generating and acid consuming rock. The ratio of acid neutralization potential to acid potential was close to 1 for two tests. Seasonal acidic drainage ($pH < 4$) was observed in the tests. Elevated copper and zinc concentrations were observed even under pH neutral conditions.

Day *et al.* (2000) described research on waste rock blending completed as part of environmental impact studies for the Telkwa Coal project located near Smithers in north central British Columbia. The paper describes recommended design criteria for blended waste rock dumps. The mine planning and permitting process was put on hold in 2000.

4.6.3 APPLICATION AND LIMITATIONS

Blending can be applied in theory to the management of any mine waste. The British Columbia Metal Leaching and Acid Rock Drainage Guidelines (Price and Errington 1998) describe requirements for implementation of waste rock blending. They note that the major constraints, for example, are cost (due to material handling), performance limitations (possibility of elevated metal concentrations at neutral pH), technical uncertainty (prediction of geochemical behaviour) and the need for comprehensive material characterization in advance of construction. For these reasons, blending is an attractive approach, but it is not yet well defined.

Mehling *et al.* (1997) and MEND 2.37.1 described seven factors that should be considered when designing a blended waste rock dump:

- Mineralogy and reaction kinetics;
- Relative proportions of the acid generating and acid consuming rock types, and resulting overall NP/AP ratio;
- Proximity and arrangement of acid generating and acid consuming materials;
- Orebody geometry and mining plan;
- Construction methods;
- Hydrogeology of the waste rock dump; and
- Operational commitment and monitoring.

The second factor is probably one of the most significant factors affecting the degree to which blending can achieve the objective of non-acidic drainage and low concentrations of metals (near ideal blend). The overall geochemical balance is important and is usually expressed as the NP/AP ratio. However, for a given NP/AP, certain combinations of rock types are more likely to result in successful blending. Table 4.6-1 summarizes end-member blend using low carbonate rock (e.g. a carbonate cemented- and altered-rock type), high carbonate rock (e.g. limestone), low sulphur rock (disseminated sulphide) and high sulphur rock (massive sulphide). The combination of low carbonate and low sulphur is probably the most suitable for blending, followed by low carbonate-high sulphur, high carbonate-low sulphur and high carbonate-high sulphur. Based on these considerations, certain mineral deposits are more amenable to blending than other mineral deposits (Table 4.6-1).

Table 4.6-1
Cases of Rock Type Mixing

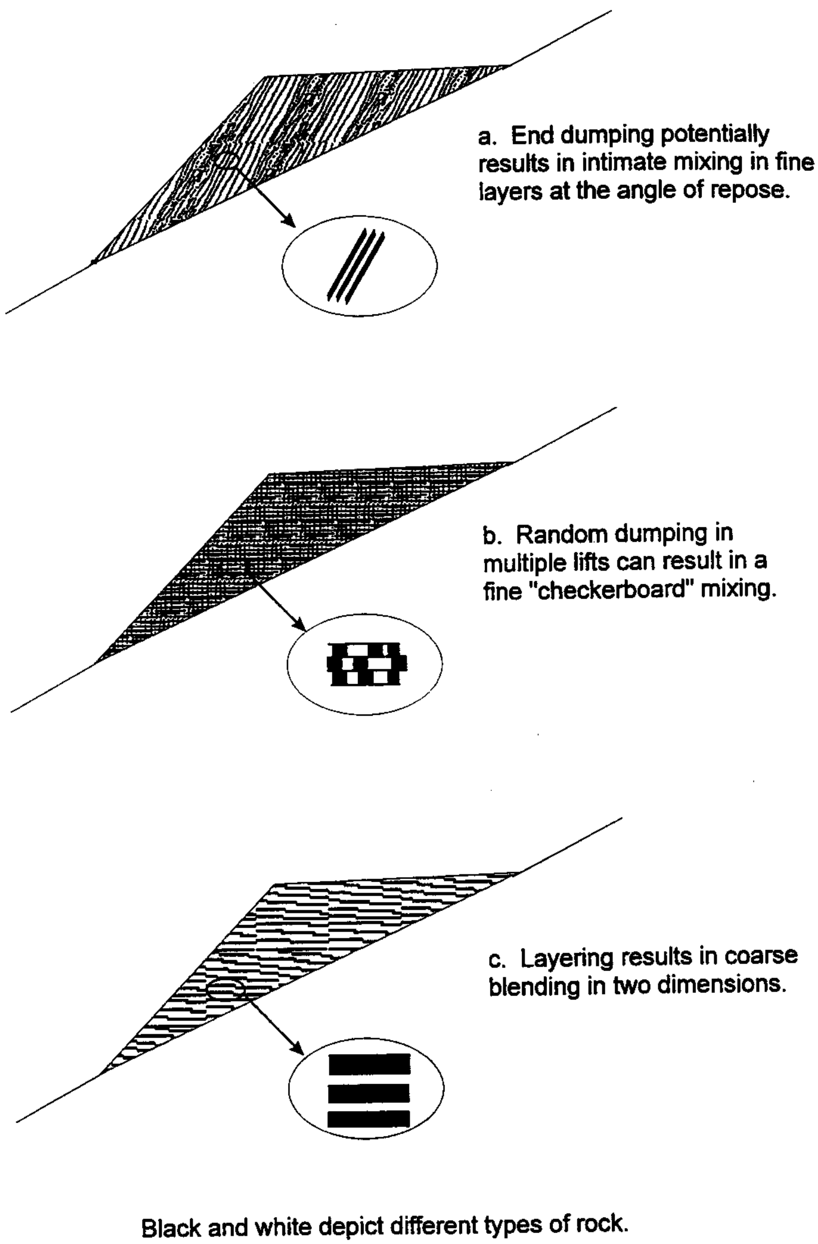
Rock Types Mixed		Acid Generating Rock Type	
		Low Sulphur	**High Sulphur**
Acid Consuming Rock Type	Low Carbonate	Lowest degree of heterogeneity, roughly equal amounts of each rock type, greatest opportunity for mixing	Small amount of sulphidic waste mixed with a large amount of weakly calcareous waste. Potentially effective blending but has some potential for hotspot formation
		Deposit types: Calc-alkalic porphyry, coal, epithermal vein, BIF-hosted gold.	Volcanogenic massive sulphide, magmatic sulphide.
	High Carbonate	Small amounts of calcareous rock mixed with large amount of low sulphur waste. Less effective than high S/low carbonate combination due to small quantities of calcareous rock.	Potential for significant chemical heterogeneities and formation of hotspots.
		Alkalic porphyry, kimberlite skarn, mesothermal veins	Skarn, limestone hosted Pb-Zn, Pb-Zn vein deposits.

Sources: Modified from Mehling *et al.* 1997 and MEND 2.37.1

Orebody geometry and mine plan play an important role in determining the economics of blending. Ideally, blended waste rock piles should be constructed as mining progresses. Stockpiling of acid generating rock is not desirable because it potentially allows the material to become acidic prior to blending. This eliminates the possibility of an ideal blend and increases the requirement for an excess of alkaline material. Therefore blending is applicable if the acid generating and acid consuming rocks are accessed concurrently.

Several different methods can be used to construct waste rock dumps (Figure 4.6-2). End-dumping over high faces potentially results in intimate mixing, though the presence of steeply dipping layers could result in water draining down a single layer and not coming into contact with other materials, or mixing with other waters. Construction of numerous horizontal lifts with random free-dumping could result in fine checkerboard mixing. Deliberate construction of lifts of acid generating rock with lifts of acid consuming rock results in coarse two-dimensional layering. Although the latter is potentially attractive from an operational standpoint, it is least likely to result in a successful blend.

Figure 4.6-2 **Schematic Illustration of the Effect of Different Methods of Waste Rock Construction**



Source: Mehling *et al*. 1997

The hydrogeology of the waste rock pile is important because the conditions required for successful blending must exist along almost every flow path. If preferential flow occurs in a way that favours the acid generating rock, acidic drainage is more likely than might be expected based on the bulk characteristics of the rock. The behaviour of the blend hinges on hydrogeological considerations. For example, at Samatosum the geochemical blend was probably appropriate to prevent acidic drainage, but flow was preferentially channeled through potentially acid generating rock (MEND 2.37.3).

Finally, blending requires commitment at all levels of the operation to ensure that different rock types are managed appropriately. Detailed management systems must be designed, and could include the following components (MEND 2.37.1):

- Pre-mining evaluation of the feasibility of blending using a block model approach to waste characterization;
- In-mine pre-blast characterization of waste blocks to determine destination in the waste dump;
- One or more mine employees dedicated to compilation of waste characterization data and in-mine scheduling of waste management;
- Means of directing dump trucks to specific locations in the dump;
- Post-construction or post-depositional chemical monitoring; and
- Confirmation that blending criteria are being met or exceeded.

Hockley *et al.* (1997) provided an example of a comprehensive field control system for a waste rock pile relocation. The project involved relocation of 65 million m^3 of waste rock into an open pit. Waste rock was classified in the field during excavation using paste pH and NAG pH. The results were used to determine where the waste would be placed with respect to the water table.

4.6.4 BASE METHOD

Construction of blended waste rock piles requires a site-specific approach because numerous chemical, physical and mining factors must be considered, and these tend to be unique for each mining operation. The following example shows an approach that was used to design a blended waste rock pile.

Day *et al.* (2000) described development of design criteria for a blended waste rock pile at the Telkwa Coal project near Smithers, BC. For this project, five factors were identified (Table 4.6-2). For each factor, the objective was determined, followed by design criteria and follow-up operational monitoring. The design criteria described below and presented in Table 4.6-2 was developed using the extensive static and kinetic geochemical databases developed for the project (Day *et al*. 2000). The specific criteria cannot be applied to other sites.

The first factor (overall geochemical balance) addressed the requirement for non-ideal blending behaviour as a minimum. Any vertical column was required to have NP/MPA greater than 1 (note that MPA is based on total sulphur and included some non-reactive sulphur forms). On average, each dump unit was required to have NP/MPA greater than 2. These site specific factors were developed based on kinetic test results, review of the sequencing of mine wastes through the mine life, and comparison with experience at coal mines elsewhere.

The second factor (internal chemistry control) leads to a design criterion that approaches ideal blending. The criterion is that layers of potentially acid generating rock are no thicker (in a vertical direction) than 1 m. This thickness was calculated based on the expected solubility of alkalinity in non-acid generating layers and the rate of oxidation (under alkaline conditions) in potentially acid generating layers. This indicated that PAG waste could not be end-dumped until the dumped rock would form an end-dumped layer thinner than 1 m.

The third criterion (geochemical variability) requires that placement of any material that could generate acid rapidly is avoided. Primarily NP defined this because it controls the likelihood of net acid generation before the current dump face is covered by more waste rock. Coal cleanings were identified as unsuitable and were excluded from the blended waste rock dumps.

The fourth factor was included to address the site-specific issue of slaking of sedimentary rock waste which could preferentially result in pyrite release, possibly resulting in a need to adjust the first two design criteria.

The fifth factor addresses the need to limit oxygen movement into the waste rock. The placement of an engineered soil cover was intended to occur before any of the potentially acid generating rock could have generated acid.

Although not shown in this table, the designs included construction of the dump on low permeability glacial till and drainage collection ditches to ensure that any poor quality drainage could be captured and treated as necessary.

4.6.5 Cost

Waste rock blending can significantly increase the cost of waste handling due to stricter scheduling requirements, the potential requirement to re-handle wastes, greater personnel requirements for monitoring, and analytical costs for blast hole cuttings analysis.

Table 4.6-2

Proposed Design Criteria for Blended Waste Rock Dumps

Factor	**Objective**	**Design Criteria**	**Monitoring**
Overall geochemical balance	No net acid generation (non-ideal blend)	NP/MPA>1.0 for any vertical column (frequency depends on rate of construction of dump) NP/MPA>2 for dump unit	Blast hole with periodic fines monitoring
Internal chemistry control	No internal acid generation (ideal blend)	PAG layers thinner than 1 m (vertical thickness)	Monitor trucks, no PAG dumped until dump face high enough.
Geochemical variability	Prevent PAG units from generating acid on dump faces	NP>15 kg/t. NP/MPA>1 No whole dump truck loads of reactive rock (coal cleanings) to any blended dump. Send to separate dump.	Engineering control. Visual confirmation. Dumping location monitoring.
Physical variability due to slaking	Limit preferential exposure of pyrite due to slaking	See overall geochemical balance (based on fines composition).	Controlled actual slaking tests once mining begins to determine the progress of fines composition.
Oxygen movement control	Limited to diffusion, no convection	Place reclamation till soil before the lowest NP in the oldest part of the dump would be expected to go acid (15 years). If dump not completed, place intermediate soil layer.	Start and continue humidity cells in parallel with mining. Place cover is a low NP cell suggests acid generation. Annual infrared surveys to monitor heat generation Quarterly seepage monitoring. Visual monitoring of particle size segregation.

Source: Modified from Day *et al.* 2000

4.6.6 MEND AND RELEVANT PUBLICATIONS

MEND 1.19.1 1994. Long Term Acid Generation Studies: Cinola Project British Columbia. March.

MEND 2.37.1 1998. Blending Layering Waste Rock to Delay, Mitigate or Prevent Acid Rock Drainage and Metal Leaching: A Case Study Review. April.

MEND 2.37.3 1997. Control of Acidic Drainage in Layered Waste Rock; Laboratory Studies and Field Monitoring. September.

Day, S.J. and Hockley, D. 1998. *Risk Assessment and Management Associated with Blending Waste Rock.* In: Summary Notes of the 5th Annual BC MEM – MEND 2000 Metal Leaching and ARD Workshop (also MEND BC. 01). Vancouver, British Columbia. 9 to 10 December.

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4.7 SEPARATION AND SEGREGATION

4.7.1 TAILINGS SULPHIDE SEPARATION

The possibility of separating sulphides to produce low-sulphur tailings has been investigated with a view to creating materials that can be used for mine site rehabilitation. In this regard, MEND has investigated available mineral separation techniques, the possible disposal of low-sulphur tailings and the use of these tailings in engineered covers, and potential cost benefits.

4.7.1.1 Discussion of Theory

At some sites, it may be possible to use tailings reduced in sulphur content for mine site rehabilitation. This is an area of active research, and is one that is likely to be of greatest interest to mine sites where the local natural borrow materials are not readily available or are uneconomical for use in rehabilitation.

Tailings that have been processed to reduce their sulphur content are referred to as desulphurized tailings or low-sulphur tailings. It is noteworthy that tailings can be expected to have some, albeit low, residual level of sulphides after sulphide separation. The low-sulphur tailings should have enough neutralization potential to safely compensate for the residual levels of sulphides.

Sulphide content can be estimated based on the acid potential (AP). Mineral processing techniques can be used to reduce the sulphur content and more specifically the AP of the processed tailings to produce a low-sulphur tailings with a net neutralization potential (NNP) of greater than 20 kg $CaCO_3$/t – this limit is based on guidelines (Table 4.7-1) proposed by the British Columbia Acid Mine Drainage Task Force (1989). Benzaazoua and Bussière (1998) indicate that the Sobek standard acid-base accounting method provides a good basis for initially assessing the suitability of low-sulphur tailings for mine site remediation. The suitability of low-sulphur tailings for use in rehabilitation can be better defined by also using kinetic testing. When porewater quality may be an issue, leach testing can also be used to characterise the low-sulphur tailings leachate.

Table 4.7-1
Tailings Classification According to the Draft BC AMD Guidelines

NNP (kg $CaCO_3$/t)	Tailings Category
Greater than 20	Non-acid generating
Between 20 and -20	Uncertain
Less than -20	Acid generating

Sources: B.C. AMD 1989; Hanton-Fong *et al.* 1997

The use of low-sulphur tailings as a site rehabilitation material offers the potential to reduce closure costs at some sites. In concept, the cost of processing local sulphide tailings, as opposed to importing materials, could be offset by the reduced cost for material transportation. Some desulphurized tailings have suitable hydraulic and geotechnical properties to allow their use in the capillary layer of engineered covers (MEND 2.22.2a), or as a cover layer over saturated tailings (Catalan *et al.* 1999).

Finch (1998) indicates that future development in minerals processing, such as improved designs in flotation and gravity separation, may improve the economic viability of separating sulphides from other gangue minerals in tailings.

4.7.1.2 Discussion of MEND Research

MEND has played a key role in tailings separation research. MEND 2.45.1a examined the separation of sulphide from mill tailings, and led to continuing research in both tailings desulphurization and the use of desulphurized tailings in engineered covers (MEND 2.22.2; MEND 2.22.3). In addition, a comprehensive field lysimeter study completed under MEND 2.45.2 assessed the potential benefits of low-sulphur tailings disposal, and the possible use of low-sulphur tailings in preventing acid generation in run-of-the-mill tailings.

MEND 2.45.1a Separation of Sulphides from Mill Tailings

MEND 2.45.1a reviews the application of various metallurgical processes for the separation of sulphides from tailings. The report provides good descriptions of the operating principles of each technology, and presents bench scale testing results.

In the study, tailings samples collected from three mills in the Abitibi region of Québec were characterized and then processed to reduce their sulphide content. The mineral separation processes that were investigated included the Falcon concentrator, Knelson concentrator, Reichart tray, 1/8 Wilfrey shaking table, Carpco LC3000 spiral concentrator, and direct flotation with various pre-processing steps and collectors. The gravity techniques were found to be incapable of sufficiently separating the sulphide and non-sulphide fractions. Direct flotation was shown to be effective at reducing the sulphide content to a sufficiently low level.

The low-sulphur tailings samples were then characterized to evaluate the effectiveness of the sulphide separation processes. This characterization work included acid-base accounting tests, humidity cell tests, ICP analysis and mineralogical evaluations. Conceptual flowsheets were also developed and used to estimate sulphide separation costs.

MEND 2.22.3 Tailings Desulphurization and Their Use in Engineered Covers

MEND 2.22.3 and Benazaazoua *et al.* (2000a) reported on a series of tests conducted in Denver cells to study the sulphide flotation kinetics of four different mine tailings samples. Tailings samples P, D, M and G contained 2.9, 3.4, 16,2 and 24.2 wt% sulphur respectively. Table 4.7-2 contains selected data on the four tailings samples including the mineral and chemical composition, and the acid generating and acid neutralization potential. Tailings P, M and G were cyanide free and were floated at a pH of less than 10 using amyl xanthate as the collector. However, Tailings D, from a gold cyanidation by-product, did not yield good sulphide recovery with a xanthate collector due to pyrite depression, and this problem was overcome using amine acetate. This collector allowed easy flotation without a pretreatment stage. In another study, Benzaazoua *et al.* (2000b) reported on the effect of the particle size on the flotation kinetic. In this study froth flotation of fine mine tailings achieved good results with regards to depyritization efficiency with a potassium amyl xanthate collector and a Sasfroth Sc39 frother.

Table 4.7-2
Selected Data for Tailings G, M, D and P

Item	Tailings P	Tailings D	Tailings M	Tailings G
Sulphide minerals	Pyrite, pyrrhotite, chalcopyrite	Pyrite, chalcopyrite, sphalerite	Pyrite, pyrrhotite, chalcopyrite, sphalerite	Pyrite, pyrrhotite, chalcopyrite, sphalerite
Carbonate minerals	Siderite, ankerite, calcite	Calcite, dolomite	Small amount of carbonates	Undetermined carbonates
S (wt%)	2.9	3.4	16.2	24.2
AP (kg $CaCO_3$/t)	90.3	106.3	506.3	756.3
NP (kg $CaCO_3$/t)	130	25	120	105
NNP (total S) (kg $CaCO_3$/t)	39.7	-81.3	-386.3	-651.3
NP/AP	1.4	0.2	0.2	0.1
D_{80} (µm)	105	58	65	56

Sources: Benzaazoua *et al.* 2000a; MEND 2.22.3

As indicated in Table 4.7-3, the operating cost to produce desulphurized tailings at Mine D (cyanide tailings) was estimated to be $0.55/t. For mines G and M (cyanide free tailings) the estimated operating cost was below $0.35/t. The capital cost for a tailings sulphide separation process was estimated to be in the range of one million dollars for each site.

Table 4.7-3
Economic Evaluation of Desulphurization at Mines D, G and M.
(Costs Reported in Canadian Dollars)

Item	**Mine D**	**Mine G**	**Mine M**
Tailings production rate	3300 tpd	2135 tpd	2300 tpd
Environmental sulphur recovery	91.8%	90.9%	84.2%
Type of collector	ARMAC C	KAX	KAX
Cells capital cost	\$576,000	\$528,000.	\$576,000
Total capital cost	\$1,152,000	\$1,056,000	\$1,152,000
Collector cost	\$0.40/t	\$0.18/t	\$0.12/t
Other operating costs including power and maintenance	\$0.15/t	\$0.15/t	\$0.15/t
Total operating cost	\$0.55/t	\$0.33/t	\$0.27/t

Sources: Benzaazoua *et al.* 2000a; MEND 2.22.3

Benzaazoua *et al.* (1998) also examined the benefits of using desulphurized tailings as the moisture-retaining layer in engineered covers through research involving a series of column tests**.** The configuration of the four columns used for the experiments is presented in Figure 4.7-1. The difference between the columns with the multi-layered covers is the concentration of residual sulphur in the desulphurized tailings used as the moisture-retaining layer.

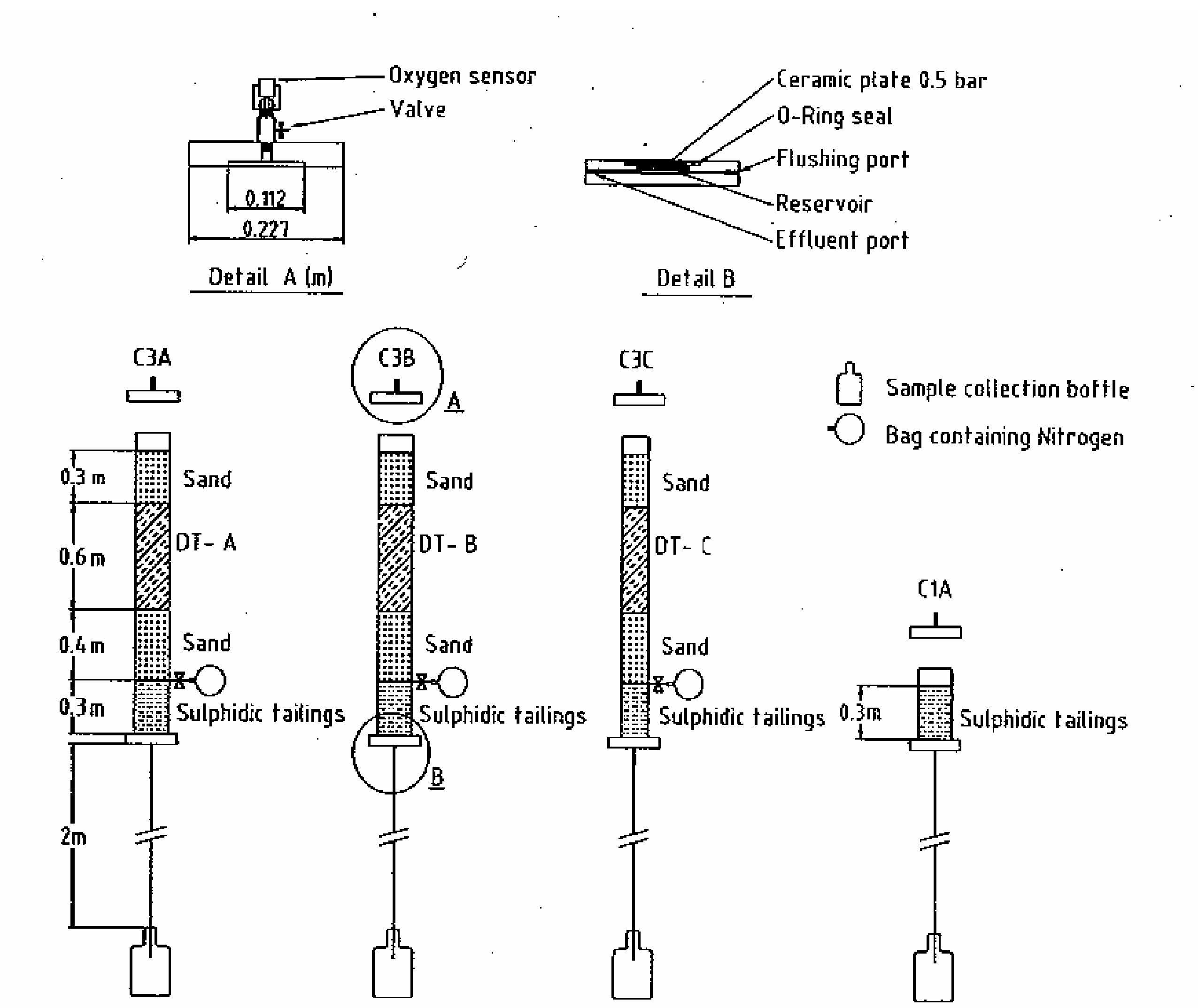
Oxygen consumption measurements (Table 4.7-4) and tailings porewater quality data demonstrated that it is possible to reduce the rate of sulphide tailings oxidation by a factor of 10 to 50 through the use of an engineered capillary break cover with the moisture retaining layer constructed of fine-grained depyritized tailings.

Table 4.7-4
Oxygen Consumption Measurements in Column Tests

Column	**Pyrite Content of Depyritized Tailings in the Moisture Retaining Layer of the Cover**	**Oxygen Consumption (mole $O_2/m^2/yr$)**
Control column (the 49 % FeS_2 tailings were uncovered)		Initially rose to 1800, and then declined to about 550 after 125 days.
Covered tailings column A	0.22%	<20
Covered tailings column B	0.65%	25 to 75
Covered tailings column C	1.17%	

Sources: Benzaazoua *et al.* 1998; Bussière *et al.*1997

Figure 4.7-1 **Description and Stratigraphic Configuration of the Columns**



Source: Benzaazoua *et al.* 1998

The use of a multi-layer desulphurized tailings cover in column tests reduced metal loadings (as measured at the base of the cover) by factors of about 2 to 4 orders of magnitude (Table 4.7.5).

Table 4.7.5

Ratio between the Concentration of Selected Parameters in the Percolated Water of the Covered Columns and the Control Column

Covered Column	Fe	Cu	Zn	SO_4
A	938	2678	17 814	22
B	207	6795	10 434	15
C	346	18 545	23 349	22

Source: MEND 2.22.3

Bussière *et al.* (1997) examined the cost of using depyritized tailings in a capillary break cover over sulphide tailings. The estimated costs (Table 4.7-6) were based on producing low sulphur tailings with an NP:AP ratio of at least 3:1.

Table 4.7-6

Desulphurization to Produce a 3:1 (NP:AP) Ratio or Better

Item	Tailings 1	Tailings 2	Tailings 3
Initial percent sulphur	12.8	21.6	3.71
NP (kg $CaCO_3$/t)	55.8	92.4	136
Acceptable percent sulphur	1.15	2.32	3.3
Required sulphur recovery	92.1%	91.2%	11.1%
Desulphurization cost	\$0.62/t	\$0.72/t	\$0.20/t

Source: Bussière *et al.* 1997

The unit costs for desulphurization shown in Table 4.7-6 are similar in range to those reported by McLaughlin and Stuparyk (1994) and Humber (1995). In some cases where the gold is associated with the pyrite, it is possible to recover the gold in the sulphide concentrate thus reducing the cost of desulphurization (MEND 2.22.3).

Bussière *et al*. (1997) estimated the cost of three rehabilitation options for a hypothetical 60 ha tailings area. Other parameters considered for this case study include:

- Two years of production remain before closure;
- Production rate of 2000 tonnes/day;

- 10% sulphur in the tailings;
- Borrow area located approximately 10 km from the tailings area (sand, gravel and clay);
- Acidity of the effluent at 500 mg/L (without cover); and
- Flow rate of 190 m^3/hr (to treat, without cover).

The three options considered for rehabilitation were: i) the use of depyritized tailings in an engineered capillary break cover; ii) the use of a clay layer in an engineered capillary break cover; and iii) the perpetual collection and treatment of acidic drainage. The estimated costs that are summarized in Table 4.7-7 indicate that the use of low sulphur tailings in a multi-layer cover may reduce site remediation costs.

Table 4.7-7
Closure Cost Comparison

Closure Option	**Equivalent Unit Cost for Tailings Area Rehabilitation Taking Into Account Both Capital and Operating Costs**
Engineered cover with a depyritized tailings layer	$106,715/ha
Engineered cover with a clay layer	$118,200/ha
Perpetual collection and treatment of acid drainage	$169,805/ha

Source: Bussière *et al.* 1997

MEND 2.45.2 Separation of Sulphides from Mill Tailings

MEND 2.45.2 was an integral part of an initiative by INCO Ltd. to investigate potential methods to reduce the reactivity of mill tailings at the Copper Cliff tailings area. MEND 2.45.2 reported that the separation of sulphide minerals and the disposal of low-sulphur tailings is a promising strategy for the prevention of acid drainage.

INCO's research into the desulphurization of tailings and low-sulphur tailings applications included a pilot scale test at the INCO Clarabelle Mill, near Copper Cliff, Ontario. It was demonstrated that a substantial quantity of residual sulphides in the rock tailings could be removed with increased flotation in a slightly acid pulp (Stuparyk *et al.* 1995). This process has the potential to provide a large volume of potentially unreactive low sulphur tailings for use in tailings cover and dam construction.

McLaughlin and Robertson (1994) reported that INCO's plant test, which involved processing over 10,000 tonnes of tailings, confirmed its ability to produce low-sulphur tailings. During the plant evaluation, tailings grades below 0.4% S were achieved at pH 6 to 7 with xanthate addition (at dosages of up to 0.02 lb/ton) and 10 to 15 minute nominal retention time. The feed sulphur assay and retention time were the most important factors influencing the sulphur assay of the test circuit tailings.

The use of froth flotation as a means of removing sulphides from tailings has been examined on other occasions as well. Raicevic (1979) reported that in froth flotation tests of uranium tailings (2 wt% FeS_2) at Elliot Lake, Ontario, some 98% of the pyrite was removed. Benn (1992) reported that the removal of metals and sulphides from lead mill tailings was considered to be feasible from an environmental perspective. At present high cost remains the key barrier to the widespread use of this technology for sulphide separation.

INCO's plant test was based on earlier laboratory experiments using froth flotation (McLaughlin and Stuparyk 1994) where sulphides were separated from a 1 wt% S feed to produce low sulphur tailings (e.g. < 0.5 wt% S) at 94 wt% recovery, as well as a 11 wt% S sulphide concentrate. The reactivity of the low sulphur tailings was assessed through additional laboratory and field experiments (McLaughlin and Stuparyk 1994; McLaughlin and Robertson 1994).

In 1993, three 10 m by 15 m, geomembrane-lined lysimeters were constructed in the Copper Cliff tailings area (Figure 4.7-2). Each lysimeter was equipped with a drain to promote the downward migration of the tailings porewater. The lysimeters were filled with three types of tailings (i.e. low sulphur tailings [LST], main rock tailings [MT], and total tailings [TT]) as indicated in Table 4.7-8. Shaw (1996) reported that the three tailings types had similar mineralogy, but differed in their pyrrhotite content. Pore gas oxygen levels were depleted within the top 20 cm of the MT and TT lysimeters indicating that sulphide oxidation was occurring. Complete pore gas oxygen depletion was not observed in the LST lysimeter.

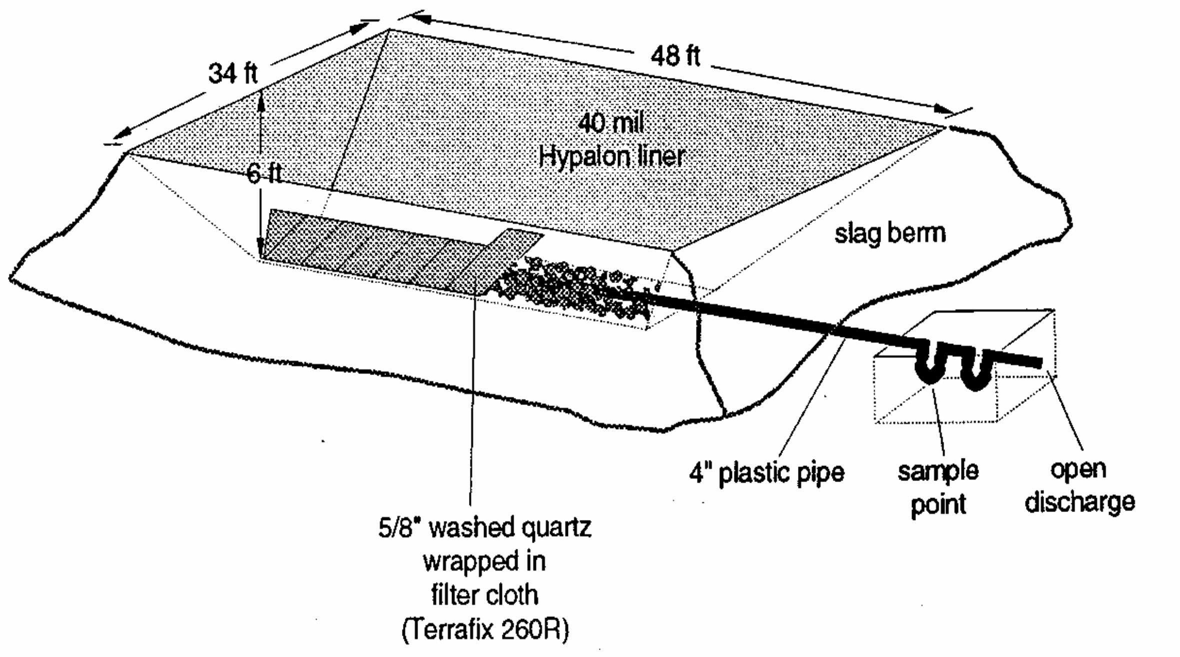
Table 4.7-8

Tailings wt% Sulphur, Lysimeter Oxygen Flux and Sulphide Oxidation Rate

Tailings Type	**Wt% S[1]**	**NNP ($CaCO_3$/t)**	**Oxygen Flux[2] (mol/m^2/day)**	**Sulphide Oxidation Rate[2] (kg/m^2/yr)**
Low sulphur tailings (LST)	0.35%	0 to 10	0.21	0.98
Main rock tailings (MT)	0.98%	-10 to –25	1.70	7.97
Total tailings (TT)	2.3%	-30 to -55	1.01	4.73

Sources: (1) Hanton-Fong *et al.* 1997
(2) Stuparyk *et al.* 1995

Figure 4.7-2 **Field Lysimeter Used in Tailings Area Evaluation**



Source: McLaughlin and Robertson 1994

Oxidation in the MT and TT lysimeters produced acidic conditions with pore water concentrations up to 1,500 mg/L Fe, 1,500 mg/L Ni and > 13,000 mg/L SO_4. In comparison, the pore water concentrations in the LST lysimeter were significantly lower at < 5 mg/L Fe, < 40 mg/L Ni, and < 2,000 mg/L SO_4. The results suggest that tailings pore water acidification and metal loading to the environment could be substantially reduced through the production of low-sulphur tailings.

The lysimeters monitoring program included the collection and testing of pore water, pore gas and tailings solids samples. The resulting geochemical data were assessed using the MINTEQA2 equilibrium speciation mass transfer model (Allison *et al.* 1990) which calculates mineral saturation indices and identifies mineral phases that may affect pore water quality.

In conjunction with the field lysimeter study, column experiments were conducted under moisture saturated flow conditions. Columns packed with LST, MT, and TT were evaluated for porosity and acid neutralization capacity and monitored. The column monitoring data were not yet available when MEND 2.45.2 was completed. However, based on the results of the field lysimeters, it was expected that the tailings porewater could be buffered to a series of pH values as minerals are dissolved and consumed (i.e. carbonate dissolution at pH 6.5 to 7.5, gibbsite dissolution at pH 4 to 4.3, and ferrihydrite dissolution at pH 2.5 to 3.5)

MEND 2.45.2 reported that based on the results obtained to June 1996, and hence after more than three years of field study under natural weathering conditions, the porewater pH in the LST lysimeter remained near neutral. The concentrations of dissolved constituents were similar to those measured at start-up, and were two to three orders of magnitude lower than the concentrations that developed in the MT and TT lysimeters. In the MT and TT lysimeters, the porewater had become acidic with elevated concentrations of SO_4 and metals. The MT lysimeter exhibited the effects of tailings oxidation within four months of the start of the test. Factors such as moisture content and tailings grain size affected the overall rate of oxidation, acidification and the extent of metal loading to the pore water. In particular, a high degree of moisture saturation at the surface of the tailings ponds inhibited oxygen penetration to unoxidized subsurface sulphide minerals.

MEND 2.45.2 concluded that low-sulphur tailings have the potential to be an alternative mining waste. In addition, low-sulphur tailings have the potential to be a useful filling material in waste rock dams, or in capping more reactive sulphide tailings.

In another study, Puro *et al.* (1995) reviewed four options for the long-term care and maintenance of the INCO Ltd. Copper Cliff tailings area. The first option relied on the treatment of acidic drainage and surface run off from the area, while the other three options incorporated acidic drainage prevention. The preferred option involved the use of low-sulphur tailings for dam construction and cover material. Ongoing research at INCO has shown that low-sulphur tailings are non-acid generating and produce seepage with relatively low dissolved constituents. Effluent from the Copper Cliff tailings area may however, continue to be treated over the long term as a result of historical tailings deposition.

4.7.1.3 Cost

Based on research completed to date, the estimated unit cost for depyritization at an operating mill is expected to be in the range of \$0.35/t for non-cyanide tailings, and about \$0.60/t for cyanide tailings.

The quantity of sulphide minerals that would need to be segregated and removed using minerals processing techniques is dependent on the characteristics of the tailings and as such this aspect needs to be assessed on a site-specific basis. Costs are likely to be sensitive and subject to variation due to the specific characteristics of the tailings, the depyritization process, and the low-sulphur tailings application.

4.7.1.4 MEND and Relevant Publications

MEND 2.22.2a 1996. Évaluation en laboratoire de barrières sèches construites à partir de résidus miniers. Mars.

MEND 2.22.3 1998. Valorisation des résidus miniers – une approche intégrée. Avril.

MEND 2.45.1a 1994. Separation of Sulphides from Mill Tailings Phase I. June.

MEND 2.45.2 1997. Separation of Sulphides from Mill Tailings – Field. New Approaches to the Prevention of Acid Mine Drainage. September.

Allison, J.D., Brown, D.S. and Novo-Gradac, K.J. 1990. <u>MINTEQA2/PRODEFA2, A Geochemical Assessment Model for Environmental Model for Environmental Systems: User's Manual.</u>. U.S. Environmental Protection Agency Report EPA/600/3-91/021, 106 p.

Argall, G.O. 1978. *Second International Tailings Symposium*. World Mining. Vol 31. September. pp. 71-75.

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Catalan, L.J.J., Li, M.G., McLaughlin, J., Nesset, J. and St-Arnaud, L. 1999. *Évolution de résidus dépyritisés comme matériau de recouvrement pour limiter le drainage minier acide*. In: Séminaires Mines Écologiques, Congrès APGGQ – Les sciences de la terre : une vision planétaire. Rouyn-Noranda, Québec. 14 April. 10 p.

Finch, J.A. 1998. *And Where Are We Going? Comminution, Flotation and Gravity Separation.* CIM Bulletin. Vol 91, No. 1025. November/December. pp. 68-69.

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McLaughlin, J. and Stuparyk, R. 1994. *Evaluation of Low Sulphur Rock Tailings Production at INCO's Clarabelle Mill.* In: Proceedings of the Innovations in Minerals Processing Conference. Laurentian University. Sudbury, Ontario. 6 to 8 June. pp. 129-145.

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4.7.2 ROCK SEGREGATION

4.7.2.1 Discussion of Theory

In many types of mineral deposits, a small component of waste rock is potentially acid generating or metal leaching, while the majority is either geochemically inert or acid consuming. Some examples of acid generating components include sulphidic coal cleanings above and below coal seams, uneconomical sulphidic coal seams, metasedimentary xenoliths (inclusions) in granite, uneconomical massive sulphide pods in skarn and massive sulphide deposits, and sulphide-rich shoots in porphyry deposits. Improper handling of these components can result in local acid generating zones within rock piles which may be accompanied by acidic seepage and and/or mobilization of metals from the dominantly non-acid generating rock. A small component can potentially contaminate a large volume of rock such that the whole rock mass must be remediated, or all the drainage treated.

The potential benefit of segregation is that the bulk of the waste rock can be disposed without significant drainage quality concerns. The smaller volume of reactive rock can then be disposed in a specially engineered location. A reactive rock disposal facility may, however, produce very severe acidic drainage, thus requiring ongoing collection and treatment and eliminating the opportunity for final reclamation.

The counter-approach to segregation is blending, in which the reactive component is deliberately incorporated into the larger mass of acid consuming rock. Blending makes use of the overall acid neutralizing capacity of the rock and avoids engineering a reactive rock disposal area. The decision to proceed with either segregation or blending depends on the specific features of the project (see Application and Limitations). Segregation may form part of a blending operation to ensure that the potentially acid generating rock is thoroughly mixed with potentially acid consuming rock. Waste rock blending is described in Section 4.6.

The principles of waste segregation are identical to the ore/waste separation methods used for mine grade control. One or more parameters are used to classify different types of rock. In the case of ore/waste separation, economic parameters such as metal content and extraction cost are used, while for waste segregation, parameters that indicate the geochemical reactivity are used. These parameters can include rock type or geological character (determined visually), sulphur content, acid-base accounting, net acid generation (NAG), total metal content and soluble metal content. The appropriate parameter is used to identify the different types of waste materials at the mine face and the segregation occurs during loading.

4.7.2.2 Discussion of MEND Research

Segregation has been used or proposed at a number of mines, though the long-term results of segregation have not been reported in literature. The results of segregation can be gauged by confirming that the separated rock masses are not contaminated by each other (though specifically that benign rock is not contaminated with reactive rock), and that the drainage quality is acceptable for the benign rock.

Potentially acid generating rock above sulphidic coal seams is segregated in the normal course of open pit coal mining and can thus be managed separately to minimize the impact of acid generation. Typically, the potentially acid generating rock lies immediately above the sulphidic coal seam and is removed separately in the course of cleaning the seam prior to extraction. This waste has traditionally been placed on the pit floor and covered with overburden. However, because of the placement on the pit floor, this waste is subject to intermittent flushing and the release of metals as the water table rises and falls. The mismanagement of the segregated rock is known to be a significant source of drainage quality impacts at sulphidic coal mines throughout the world (Erickson 1987).

Recently, sulphidic coal mines in the Appalachian region have shown that segregation and careful management appears to create a relatively benign overburden mixture (MEND 2.37.1). The segregated rock is placed well above the water table and encapsulated by other waste rock (Skousen and Ziemkiewicz 1995). Pennsylvania Department of Environmental Protection (PDEP 1998) indicate that for segregated potentially acid generating rock:

- “High and dry” placement is the most common special handling technique;
- Permanent submergence below the ultimate water table is seldom used in Pennsylvania; and
- The volume of the material to be specially handled should generally be less than 20% of the mine backfill volume because of the need to keep acidic materials away from the surface, water table, highwalls, etc .

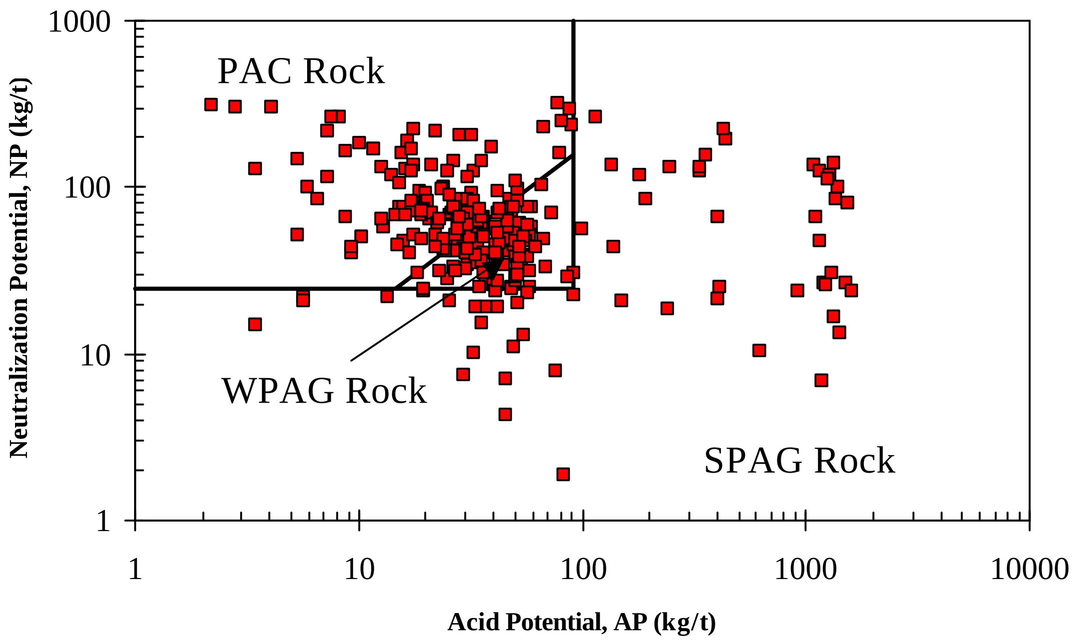
Identification and segregation of acid material is extremely difficult if multiple zones exist in the stratigraphic section unless: blasting is not used to remove the overburden; and the potentially acid generating zones are persistent (laterally and vertically), uniform in thickness, and distinctive in appearance.

The long-term success of the measures described by PDEP (1998) has not been documented.

Day *et al.* (1997) and Hope *et al.* (1997) describe the research used to develop a segregation plan for the proposed semi-massive to massive sulphide copper-lead-zinc Kudz Ze Kayah project in the Yukon Territory. Day *et al.* (1997) describe the development of operational geochemical segregation criteria for non-acid generating rock (PAC), weakly potentially acid generating rock (WPAG), and strongly acid generating rock (SPAG). Figure 4.7-3 illustrates the criteria.

The PAC rock was suitable for permanent subaerial disposal. The WPAG rock was planned for stockpiling during the mine life, followed by back filling into the flooding open pit at closure. The SPAG rock occurring adjacent to the massive sulphide ore zone was planned for immediate subaqueous disposal in the tailings impoundment. The criterion for SPAG rock was based on its neutralization potential and sulphur content. The actual criteria were developed using results of kinetic tests and evaluation of the distribution of the different types of materials, which controlled the practical details of segregation. Hope *et al.* (1997) and the Water Licence Application filed with the Yukon Territory Water Board provides details of the implementation of the segregation and management plan. During mining, cuttings from approximately 10% of waste rock blast holes (~5 per blast) would be sampled and tested for classification purposes. It was planned that sampling would increase as WPAG zones predicted by pre-mining drilling were approached. Pre-mining water quality predictions indicated that the misclassification of SPAG as WPAG would control metal concentrations in seepage from the WPAG stockpiles and the chemistry of water in the final pit lake. The case study illustrates the importance of minimizing classification errors during segregation. Permits were issued to the mining company but mining has not begun.

Figure 4.7-3 **Example of Classification of Waste Rock into Three Categories**



Source: Day *et al.* 1997

Segregation of waste rock based on acid generation potential has occurred at several mines in Western Canada. The Huckleberry Mine extracts copper and molybdenum concentrations from two open pits in a porphyry type deposit (Johnson 1999). The host rocks (andesite and granodiorite) vary from acid generating to acid consuming. Acid generating rock is segregated based on the acid-base accounting (NP/AP) and is placed in locations that will flood when the mine closes. Non-acid generating rock is used for construction of structures (primarily the tailings management facility dams) that will remain permanently exposed. Different NP/AP criteria are used for the different rock types (NP/AP = 2 or 3). Classification occurs by sampling of blast hole cuttings, on-site acid-base accounting, and flagging of dig limits using the same techniques as ore segregation.

The now-closed QR Gold Mine in British Columbia was mined for gold in a skarn deposit from several open pits. Potentially acid generating (PAG) rock was segregated for disposal in the flooded tailings impoundment. Non-acid generating rock was disposed sub-aerially. PAG rock was defined as rock with NP/AP < 2. Basic rock was defined as having NP/AP > 2.5. Rock between these criteria was defined as MPR (i.e. marginal potential ratio). Basic rock was classified as suitable for construction purposes. Initially, the MPR rock was disposed in piles, but is being backfilled into the flooding open pit as part of the closure measures.

The Diavik Diamonds Mines project in the Northwest Territories will manage a small amount of potentially acid generating biotite schist, which occurs as inclusions (xenoliths) in the dominantly granitic kimberlite host rock. Larger inclusions of biotite schist will be segregated and managed to limit water quality impacts. Segregation will occur based on microscopic examination of blast hole cuttings and geochemical analysis for sulphur (Diavik Diamond Mines Inc. 1999).

The advanced exploration-stage Pogo project in central Alaska is currently using arsenic and sulphur concentrations to segregate underground development rock based on potential to leach arsenic. The segregation is determined from analyses of blast hole cuttings.

Segregation criteria in Western Canada tend to use acid-base accounting. In the Asia-Pacific region, the net acid generation (NAG) test is more commonly used (Miller *et al*. 1997). Miller *et al.* (1991) describes the segregation of four types of waste rock using the NAG tests for the Golden Cross Mine, New Zealand. If the NAG test pH was greater than 4, the rock was classified as non-acid generating and suitable for disposal without special measures. Three categories of potentially acid generating rock were based on NAG pH and paste pH. Placards used to identify different materials are described.

4.7.2.3 Applications and Limitations

Segregation is most effectively applied to isolate highly reactive rock from weakly mineralized rock. Blending is potentially appropriate when the two types of rock are not strongly mineralized and are geochemically similar except for differences in neutralization potential. The following factors should be considered:

- Segregation is only practical if a clean separation can be made. If the segregated "benign" rock is significantly "contaminated" with reactive rock, the benefit of segregation is limited; and
- Segregation is probably best applied when a disposal option for the reactive rock is available that prevents or substantially limits the rock from generating acidic drainage, for example, underwater disposal.

The first factor ("the ability to separate cleanly") is very significant. In the Kudz Ze Kayah case example discussed above, the modelling indicated that seepage water chemistry would be very sensitive to small amounts of reactive rock placed with the non-reactive rock. The ability to effectively segregate depends on the nature of the contact between the rock types, the basis for segregation (e.g. visual versus chemical), the presence of inclusions that cannot be segregated, the operating conditions and the management system.

4.7.2.4 Base Method

Incorporation of segregation into a mine plan could consist of the following steps:

1. Initial assessment of the benefits of segregation versus other management approaches (blending, management of rock as potentially acid generating or metal leaching) and preliminary selection of disposal options for segregated rock;
2. Designing a geochemical test program to develop appropriate segregation criteria;
3. Evaluation of achievable segregation using the criteria;
4. Assessment of impacts from rock that cannot be segregated, or as a result of management errors; and
5. Development of a rock management plan and design of facilities.

The rock management plan for an open pit mine could include the following:

- Spatial distribution and frequency of blast hole sampling;
- Analytical methods, including quality control;
- Management system for feedback to mine operators;

- Definition of dig limits;
- Training of shovel operators;
- Placard system for haul trucks;
- Control of dumping locations;
- Follow-up sampling at disposal locations to confirm segregation effectiveness; and
- Follow-up water chemistry monitoring to confirm correlation between geochemical criteria and water chemistry.

4.7.2.5 Cost

Generic costs of segregation cannot be provided. The operational costs are potentially significant due to need for extra personnel to collect and analyze samples, the need for operational systems to monitor rock loading and disposal, and the need for engineered disposal locations for segregated rock.

4.7.2.6 MEND and Relevant Publications

MEND 2.37.1 1998. Blending and Layering Waste Rock to Delay, Mitigate or Prevent Acid Rock Drainage and Metal Leaching: A Case Study Review. April.

Day, S., Hope, G. and Kuit, W., 1997. *Waste Rock Management Planning for the Kudz Ze Kayah Project, Yukon Territory: 1. Predictive Static and Kinetic Test Work.* In: Proceedings of the 4th International Conference on Acid Rock Drainage. Vancouver, British Columbia. 31 May to 6 June. Vol 1, pp. 81-98.

Diavik Diamonds Mine Inc. 1999. <u>Class "A" Water Licence Application - Biotite Schist Managament Plan</u>. Submitted to the Northwest Territories Water Board.

Erickson, P. 1987. *Acid Mine Drainage in US Coal Regions*. In: Proceedings of the Acid Mine Drainage Seminar/Workshop. Halifax, Nova Scotia. 23 to 26 March. pp. 89-100.

Hope, G., Kuit, W. and Day, S. 1997. *Waste Rock Management Planning for the Kudz Ze Kayah Project, Yukon Territory: 2. Waste Management Strategy.* In: Proceedings of the 4th International Conference on Acid Rock Drainage. Vancouver, British Columbia. 31 May to 6 June. Vol 1, pp. 99-114.

Johnson, D. 1999. *Operational Material Characterization and Monitoring of Weathering at Huckleberry*. In: Summary Notes from the 6th Annual B.C. MEM/MEND 2000 Metal Leaching and Acid Rock Drainage Workshop. Vancouver, British Columbia. 1 to 2 December. 14p. (also MEND BC.02).

Miller, S, D., Jeffery, J.J. and Wong, J.W.C. 1991. *In-Pit Identification and Management of Acid Forming Waste Rock at the Golden Cross Gold Mine, New Zealand*. In: Proceedings of the 2nd International Conference on the Abatement of Acidic Drainage. Montréal, Québec. 16 to 18 September. Tome 3, pp. 137-151.

Miller, S., Robertson, A. and Donahue, T. 1997. *Advances in Acid Drainage Prediction Using the Net Acid Generation (NAG) Test.* In: Proceedings of the 4th International Conference on Acid Rock Drainage. Vancouver, British Columbia. 31 May to 6 June. Vol 2, pp. 533-549.

Pennsylvania Department of Environmental Protection, 1998. Coal Mine Drainage Prediction and Pollution Prevention in Pennsylvania. Edited by K.B.C. Brady, M.W. Smith and J. Schueck. October.

Skousen J.G. and Ziemkiewicz, P.F. 1995. Acid Mine Dranage Control and Treatment. West Virginia University and the National Mine Land Reclamation Center, Morgantown, WV. 254 p.

4.8 PERMAFROST AND FREEZING

This section focuses on research into the use of cold climatic conditions and permafrost to inhibit acid generation. As a general comment, this is an area of continuing and evolving research.

MEND has shown that the potential exists to develop unique acidic drainage control strategies in Canadian permafrost regions. While the theoretical and empirical basis already exists for civil engineering projects (e.g. pipelines, foundations and dams) in permafrost regions, the management of sulphide wastes in the complex permafrost environment is still under development. MEND has investigated a number of issues in this area and provided useful findings, but additional research is required. MEND projects have investigated:

- The effects of freezing on the sulphide oxidation process;
- The effects of heat from sulphide oxidation on permafrost and freeze-back;
- The effects of freezing on metal leaching;
- Acid control strategies that could potentially take advantage of permafrost; and
- The use of water covers and dry covers in permafrost regions.

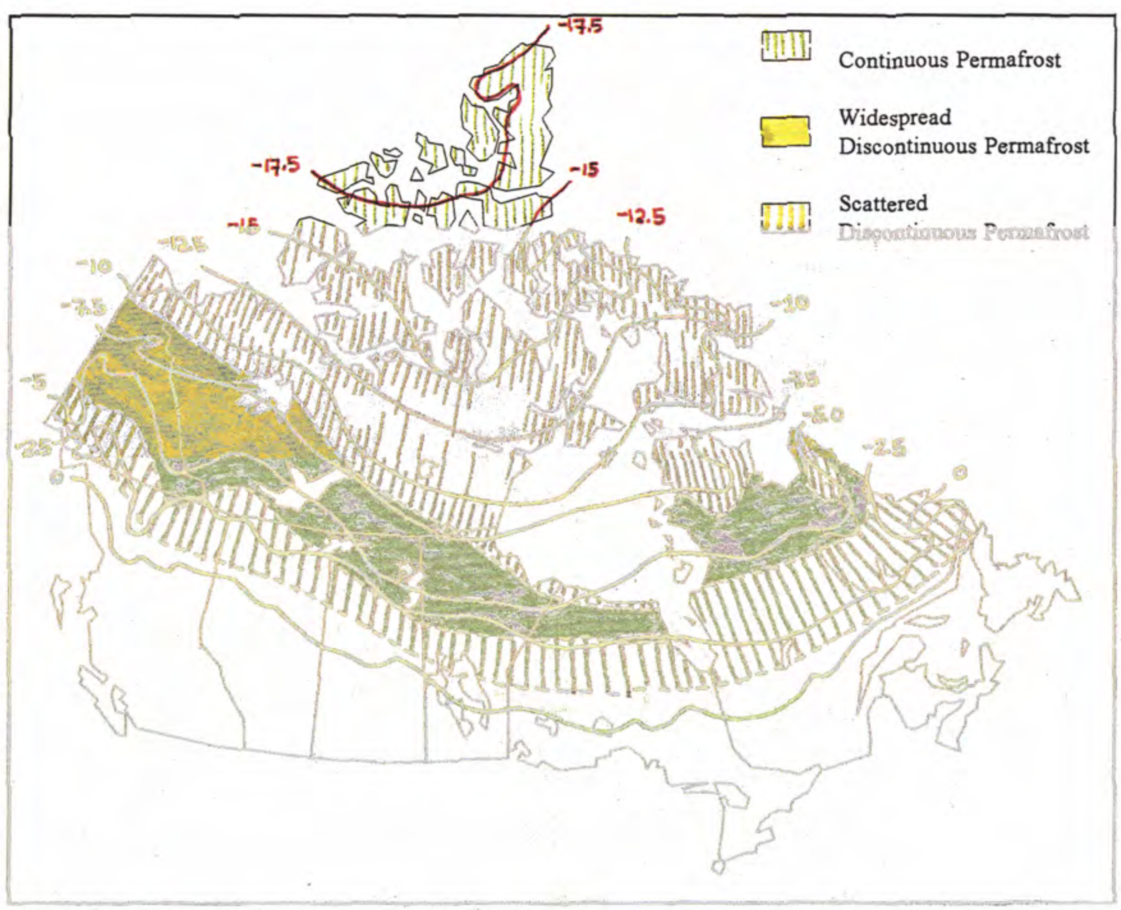
MEND projects related to permafrost and freezing are listed in Table 4.8-1.

Table 4.8-1
Projects on Permafrost and Freezing

MEND Project	Title
6.1	Preventing AMD by Disposing of Reactive Tailings in Permafrost
1.61.1	Roles of Ice, in the Water Cover Option, and Permafrost in Controlling Acid Generation from Sulphide Tailings
1.61.2	Acid Mine Drainage in Permafrost Regions: Issues, Control Strategies and Research Requirements
1.61.3	Column Leaching Characteristics of Cullaton Lake B and Shear(S) - Zones Tailings Phase 2: Cold Temperature Leaching
1.62.2	Acid Mine Drainage Behaviour in Low Temperature Regimes - Thermal Properties of Tailings
W.014	Managing Mine Wastes in Permafrost Zones. Summary Notes MEND Workshop

Permafrost conditions are present across the northern half of Canada and in particular within the Yukon, Northwest Territories, and Nunavut. The discontinuous and continuous permafrost regions of Canada are shown in Figure 4.8-1.

Figure 4.8-1 **Annual Mean Daily Air Temperature Isotherms and Permafrost Regions of Canada**



Source: MEND 1.61.1

MEND has investigated the effects of low temperatures on the rate of sulphide oxidation. The chemically and biologically catalyzed rates of oxidation under unsaturated conditions were shown to slow appreciably, but not stop, as the temperature is lowered and approaches 0º C. As such, the freezing of reactive wastes, as a stand-alone measure, may not be sufficient to prevent or control acid generation.

The possible effects of heat generated by sulphide oxidation on natural permafrost and freezing of tailings and waste rock have been assessed in low temperature laboratory studies. In the low temperature (e.g. 2ºC) leach tests, the onset of acid generation was considerably delayed in comparison to leach tests conducted at 25ºC. MEND has also reviewed the use of water covers and dry covers to control acid generation under cold climate and permafrost conditions. The application of water covers in a cold climate may be influenced by lower temperatures through the increased solubility of dissolved oxygen with decreasing water temperature, the formation and break-up of ice, and the presence or lack of snow cover. Ice scouring of submerged tailings and the resultant disturbance or resuspension of tailings can be a drawback to the application of water covers in the north.

MEND assessed acid generation control options that have the potential to take advantage of permafrost conditions. A method was conceptually developed where the encapsulation of reactive tailings in permafrost could ensure year round *in situ* temperatures at or below 0ºC. Permafrost has been found to have a finite permeability and could limit, but not stop, water mobility. It is noteworthy that permafrost, while a promising factor that influences the rate of sulphide oxidation, cannot by itself be applied for the absolute control of acid generation.

MEND completed a circumpolar review of acidic drainage experiences at mine sites and in the natural environment in permafrost regions, and identified knowledge gaps that required additional research. MEND subsequently filled an important knowledge gap through laboratory research on the thermal properties of unsaturated tailings. More research is required to fill knowledge gaps such as:

- Effects of sub-zero temperatures on sulphide oxidation;
- Hydrogeology and heat transfer within waste rock dumps;
- Large-scale field demonstrations of acid generation control strategies;
- Development of economic designs for insulating covers; and
- Use of heat transfer techniques (e.g. thermosyphons) to augment natural “permafrost” conditions.

In addition, fundamental knowledge concerning acidic drainage prevention/control opportunities needs to be combined with assessments of operational practices that may be beneficial. Such practices could include:

- Waste stripping and placement in winter months only;
- Waste dump design and construction methods;
- Snow removal to enhance heat transfer; and
- Thickened tailings disposal to reduce heat associated with tailings.

While there is a need for additional acidic drainage research in cold climate conditions, it should be noted that the present state of knowledge is sufficient to allow mine waste management facilities to be confidently designed, constructed, operated and decommissioned in permafrost regions. Each site needs to be considered on its own merits, with designs supported through thermal modelling, and due regard paid to potential effects to the active zone and underlying permafrost. Numerous designers are proposing the use of constructed cells for waste disposal; the scheduling of construction activities in cold weather months; and the use of water covers or rock layers over tailings at closure. Geotechnical engineering designs additionally address issues related to ice in dam foundation footprints and in the soils underlying waste management areas; the possible need to incorporate insulation; the use and maintenance of thermosyphons; and other technical factors related to maintaining permafrost conditions. As such, the design of waste management facilities for permafrost regions relies upon technical knowledge and practical experience in many disciplinary areas. Walk-away closure strategies are perferable for many of these sites given their remote locations.

4.8.1. DISCUSSION OF THEORY

The MEND reports listed in Table 4.8-1 provide comprehensive reviews of complex theories related to permafrost, and the effect of low temperatures on the sulphide oxidation process and metal leaching. Readers wishing to obtain a better understanding in these areas are encouraged to review the MEND reports. General information on the following issues is provided below:

- Permafrost/Climate;
- The effect of low temperatures on the rate of sulphide oxidation;
- Heat generation in sulphide oxidation;
- The effects of freezing on metal leaching; and
- The design of tailings management facilities in permafrost.

4.8.1.1 Permafrost/Climate

Permafrost is a thermal condition where the temperature of soil or rock remains below 0ºC for more than two consecutive years.

There is a general relationship between the mean annual air temperature (e.g. climatic conditions) and the extent of discontinuous and continuous permafrost regions. In Canada, the southern limit of discontinuous permafrost roughly coincides with the -1ºC mean annual air isotherm (Figure 4.8-1). The southern limit of continuous permafrost occurs approximately along the -8.5ºC mean annual air isotherm. In the discontinuous permafrost region, permafrost co-exists with areas of unfrozen ground. Along the southern fringe of this region, the permafrost occurs in scattered islands ranging in size from a few square metres to several hectares and is generally confined to peat lands and north facing slopes and areas having little snow cover. Northward, permafrost becomes increasingly widespread and varies in thickness from a few centimetres to 60 to 100 m at the discontinuous/continuous permafrost boundary, north of which the depth of continuous permafrost may increase to as much as 600 m in depth (Johnston *et al*. 1981).

Permafrost occurrence and distribution are also affected by terrain factors which include surface relief, the surface vegetative layer, local hydrology, snow cover depth, glacier ice, soil and rock type, and the loss of vegetation. The surface vegetation, and in particular a ground surface layer of moss and peat, assists in insulating the ground from thawing in the summer.

The surface layer of permafrost that undergoes seasonal thawing is referred to as the active layer (Figure 4.8-2). The depth of the active layer varies from 0.5 to 4 m, depending predominantly on the thickness of the organic layer and water content of the near surface material. On a micro-scale, freeze/thaw cycles may impact the surface of minerals within the active zone. The distribution and surrounding of the minerals may be altered on the macro-scale. The thickness of the active layer impacts upon the position of the permafrost table and may vary annually. Seasonal freeze/thaw cycles are generally limited to the active layer, where moisture movement occurs during the freezing of porous media as a result of the expulsion of water due to the 9% volume increase in the phase change from water to ice, and the attraction of water due to capillary suction at the ice/water interface (Fawcett and Anderson 1994; MEND 1.61.2). The growth of ice lenses at the freezing front is a main cause of frost heaving.

The removal or disturbance of a natural surface vegetative layer can result in permafrost degradation and may deepen the active layer. Snow cover acts as an insulator to both frost penetration and spring thaw. Permafrost is also influenced by the infiltration of surface water and the movement of subsurface water. As examples, poorly drained discontinuous permafrost zones inhibit permafrost, and thaw basins are present under lakes and rivers that do not freeze to the bottom in winter. Damage to the natural surface organic layer in permafrost regions will have varying effects on both the thermal and moisture regimes of the affected area.

The depth of zero amplitude is the vertical distance below the ground surface at which there is no fluctuation in the ground temperature (Figure 4.8-3). The temperature at the depth of zero amplitude ranges from a few tenths of a degree below 0ºC at the southern limit of the discontinuous permafrost zone, to approximately -5ºC along the discontinuous/continuous permafrost boundary, to -15ºC in the high arctic.

Groundwater within permafrost migrates generally to the active zone, which develops during the summer months. Free water may occur in the ground above permafrost (suprapermafrost), in an unfrozen zone (talik) within permafrost, or below the permafrost base (subpermafrost). Permafrost should be considered to be a practically impermeable zone.

Not all pore water freezes when fine-grained soils, such as tailings, are subjected to freezing temperatures. The temperature that is required to freeze pore water varies and is dependent on the capillary and adsorption forces, and the concentrations of dissolved impurities. Factors that determine the unfrozen water content in saturated frozen soils include the temperature, specific surface area of the solids phases, pressure, the chemical and mineralogical composition of the soil, and the solute content.

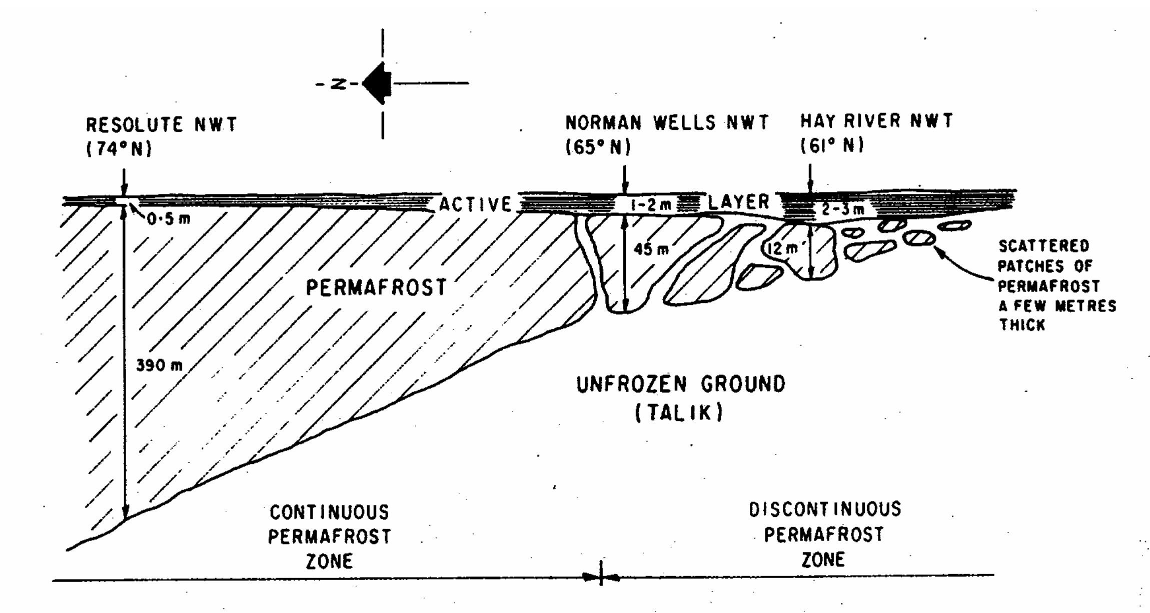
Researchers have postulated that global warming could have a profound effect on permafrost zones. Nakayama *et al*. (1993) reported on the possible effect of global warming and proposed an active layer index as an indicator of the degree of change due to a warming trend.

4.8.1.2 The Effect of Low Temperatures on the Rate of Sulphide Oxidation

MEND 1.61.1 reports that the sulphide oxidation process is temperature dependent as both chemically and biologically catalyzed sulphide oxidation rates slow with decreasing temperature. Ahonen and Tuovinen (1989; 1991) reported that microbial activity occurred at temperatures as low as 4ºC, where the growth rate of *Thiobacillus ferrooxidans* was reduced from its optimum by a factor of 10. Knapp (1987) reported that near 0ºC, the non-catalyzed rate of sulphide oxidation was less than 15% of its value at 25ºC. MEND 1.61.3 reported delayed onsets of acidic drainage, and reduced rates of sulphide oxidation at 2ºC versus those observed at 25ºC.

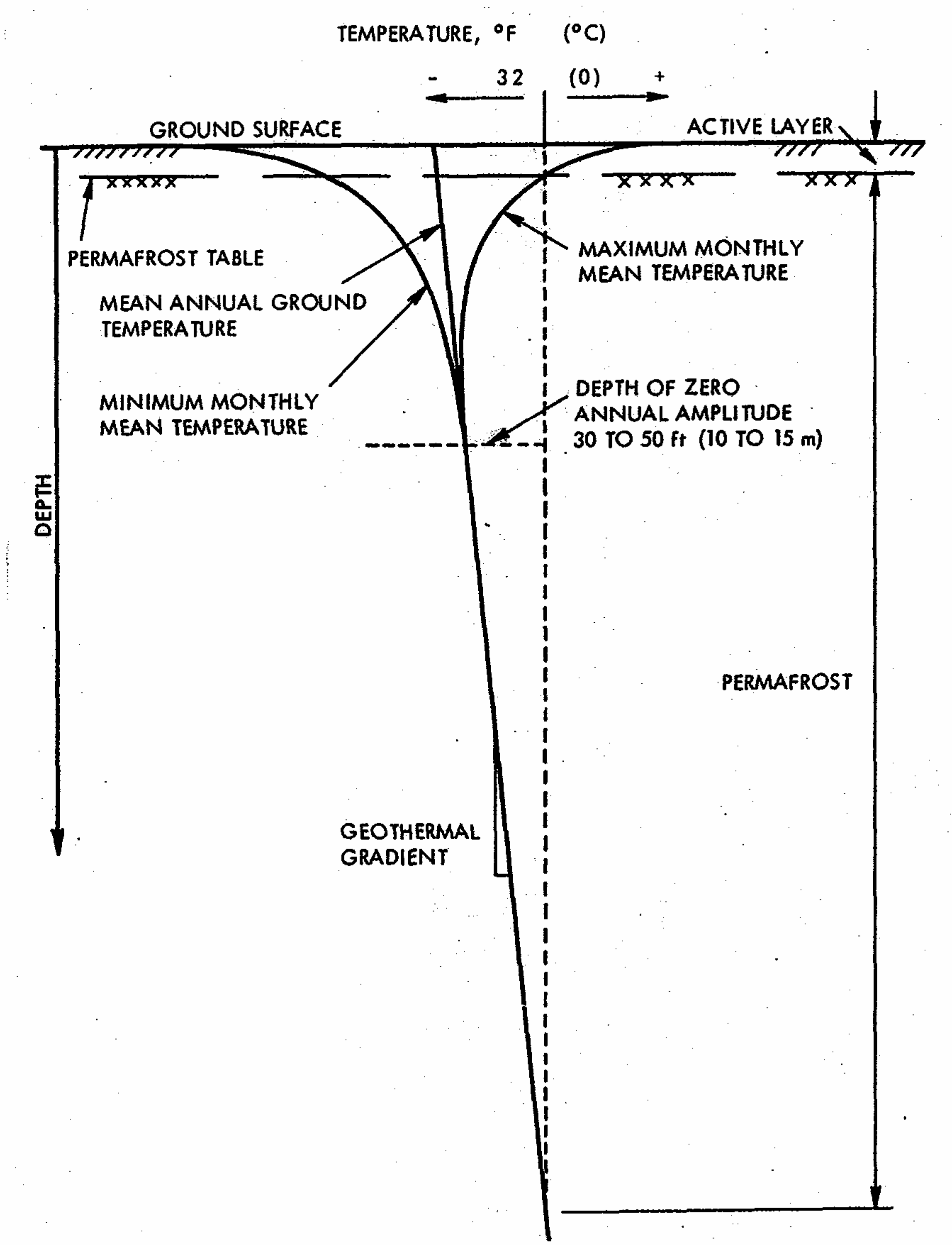
Elberling *et al*. (2000) investigated the microbial and chemical sulphide oxidation activity and oxygen consumption in the active layer of tailings at the Nanisivik Mine, located on Baffin Island in the high arctic of Canada. The tailings are comprised of 75 to 95% pyrite, with the remainder consisting of dolomite and residual sphalerite (ZnS) and galena (PbS). The tailings are usually deposited under water in a tailings impoundment. Tailings samples were collected from the top 60 cm of exposed tailings in August 1998. The researchers reported that litho- and

Figure 4.8-2 **Active Layer in Regions of Continuous and Discontinuous Permafrost**



Source: MEND 1.61.1 (after Johnston 1981)

Figure 4.8-3 **Depth of Zero Amplitude in Typical Permafrost Regime**



Source: MEND 1.61.1 (after Johnston 1981) organotrophic bacteria were present in the exposed tailings, and that calorimetric measurements indicated that bacterial activity was contributing to the oxidation of the exposed sulphides.

4.8.1.3 Heat Generation in Sulphide Oxidation

MEND reviewed the effects of heat release from pyrite oxidation on the natural freezing process. Pyrite oxidation is an exothermic reaction (Equation 1); one mole of pyrite produces around 1.4 x 10^6 J/(mole FeS_2)[1], excluding energy from other reactions (e.g. complexation, secondary mineral dissolution and precipitation).

$$FeS_2 + H_2O + \frac{7}{2}O_2 \rightarrow Fe^{2+} + 2\,SO_4^{2-} + 2\,H^+ \qquad (1)$$

The heat generated by complete oxidation of pyrite (Equation 1) is equivalent to 1.2 x 10^4 kJ per kilogram of pyrite[2]. Given the heat generation from sulphide oxidation, the temperature of oxidizing sulphide minerals can be higher than the bulk temperature of the surrounding materials even if only on a microscale. A principal issue in this regard is the potential for thermal warming of permafrost due to the heat released by sulphide oxidation. As pyrite oxidation is exothermic, temperatures at the mineral surface may remain above freezing even if the mineral is encapsulated in permafrost. The flowchart of the coupled relationship between pyrite oxidation and heat transfer is shown in Figure 4.8-4.

4.8.1.4 The Effects of Freezing on Metal Leaching

Laboratory leaching tests of reactive tailings undertaken in MEND 1.61.3 demonstrated the effect of temperature on the onset of acidic drainage and the rate of acid generation. At 2ºC tailings with 2.31 wt% S and a neutralization potential of 45.36 kg $CaCO_3$/tonne had a delayed occurrence of acidic drainage, and the acid generation rate was low such that a reasonable degree of acid neutralization was achieved. While acidic drainage was eventually generated under the 2ºC condition, the overall impact in terms of effluent metal and total acidity loading was lower than that at 25ºC.

4.8.1.5 The Design of Tailings Management Facilities in Permafrost

MEND 6.1 indicates that the temperature of the ground within a depth of some 10-m can fluctuate over the year in a time-lagged response to changes in the ambient air temperature. The

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[1] 7x 10^{-7} moles pyrite generates 1 joule of heat (MEND 1.11.1, after Harris and Ritchie 1985)
Therefore, 1 mole pyrite generates 1428.57 kJ

[2] 1 kg pyrite = 8.3333 moles

time lag in the temperature response at a depth of 2 to 8 m below the ground surface may be 1 to 4 months respectively. This suggests that a near-surface ground temperature measurement taken only at one time is likely to be insufficient for design purposes. For ground temperature measurements to be useful, the measurements should be taken on a monthly or bimonthly basis over a year or more. In addition, the temperature profile should extend down to about the zero amplitude depth. In the absence of this type of data, the mean annual permafrost temperature can be obtained by either a one time temperature measurement at a depth of at least 15 m, or estimated based on published climatic normals for the nearest Atmospheric Environment Service station – the latter method is suitable for preliminary design purposes. In general, the zero amplitude ground temperature of permafrost within the discontinuous permafrost zone ranges from about 0 to -4ºC and from about -4 to -12ºC in the continuous permafrost zone.

4.8.2. DISCUSSION OF MEND RESEARCH

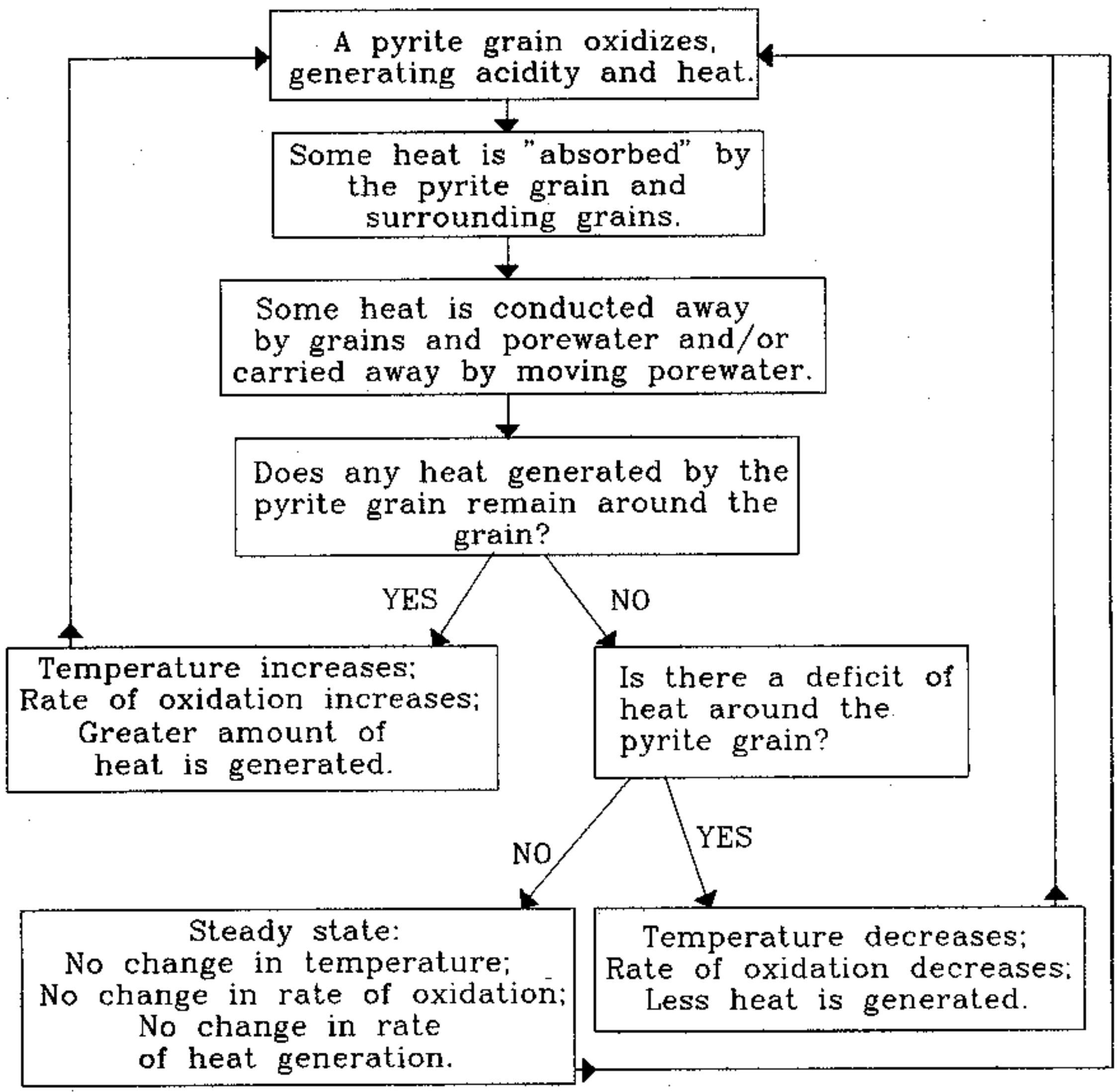
MEND 6.1 Preventing AMD by Disposing of Reactive Tailings in Permafrost

MEND 6.1 investigated potential methods to prevent acidic drainage from reactive tailings through disposal in permafrost. The project reviewed northern physiology and climatic factors, the occurrence and distribution of permafrost, and acidic drainage occurrence in cold climatic conditions. It reviewed case histories for the Lupin and Giant mines in the Northwest Territories, the Faro mine in the Yukon, as well as for uranium mines in northern Saskatchewan. The report addressed design considerations with regard to the location of tailings management facilities, hydrology, permafrost and potential concepts for permafrost enhancement. Selected aspects from the case studies for the Lupin, Giant Yellowknife and the Faro mines are provided below. The locations of the sites are shown in Figure 4.8-5.

Lupin Mine: The Lupin gold mine is situated on Contwoyto Lake approximately 380 km north-east of Yellowknife in a continuous permafrost region where the annual mean air temperature is -12ºC. The cyclic thawing of the natural ground stratigraphy consisting of 0.3 m organic layer underlain by silty-sand till is as follows: i) the ground begins to thaw about June 1 and the maximum depth of thaw (1 m) occurs at about the end of August; and ii) the frost table begins to rise in September and by mid-October the active layer has completely refrozen.

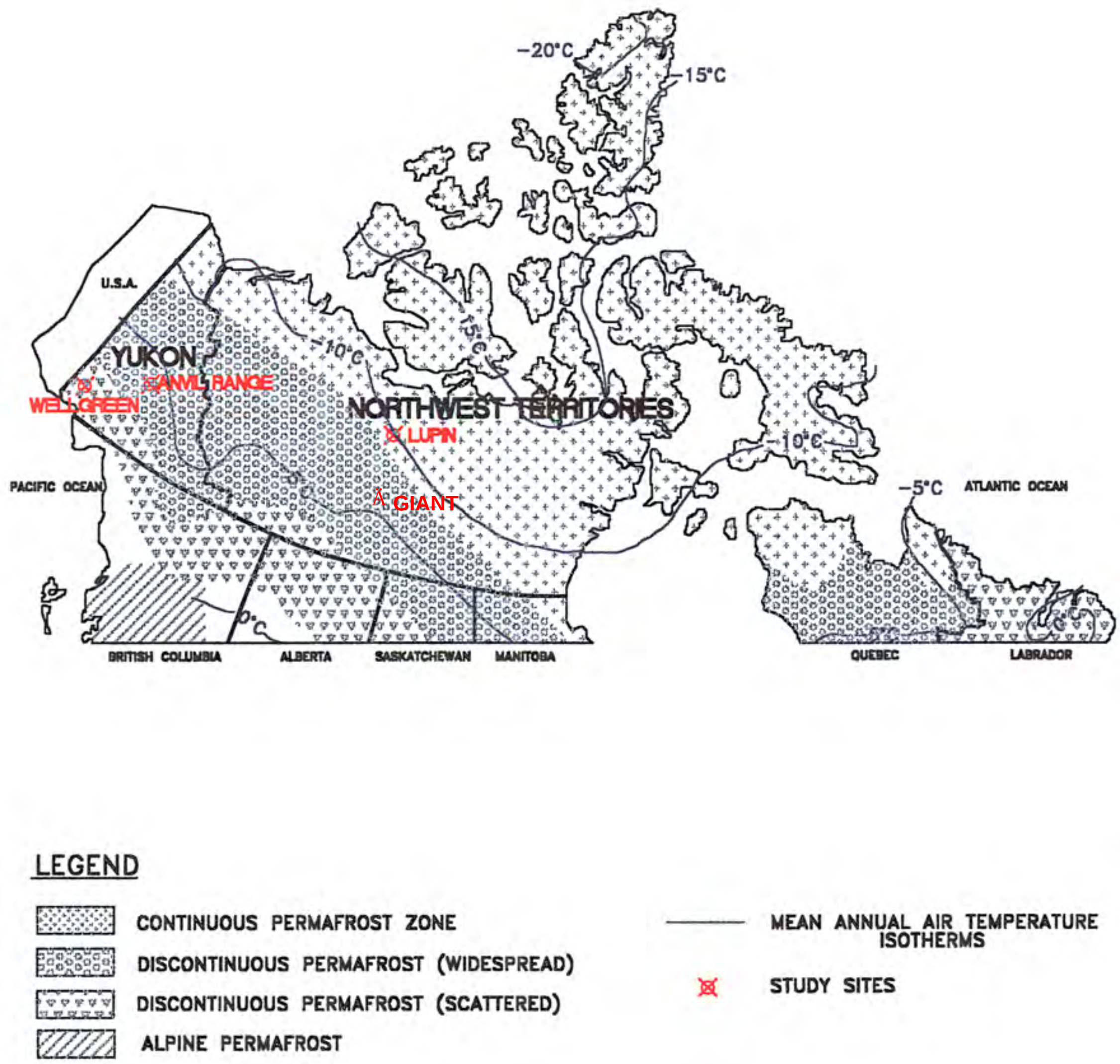
The Lupin mine tailings impoundment area was constructed in 1982. The 750 ha impoundment area is subdivided into tailings cells and polishing ponds as shown in Figure 4.8-6. Frozen core dams were constructed to prevent seepage. Extensive monitoring between 1983 and 1990 confirmed the integrity of the frozen dam core during water pond impoundment, and the development of permafrost in the tailings.

Figure 4.8-4 **Flowchart of Coupled Relationship Between Pyrite Oxidation and Heat Transfer**



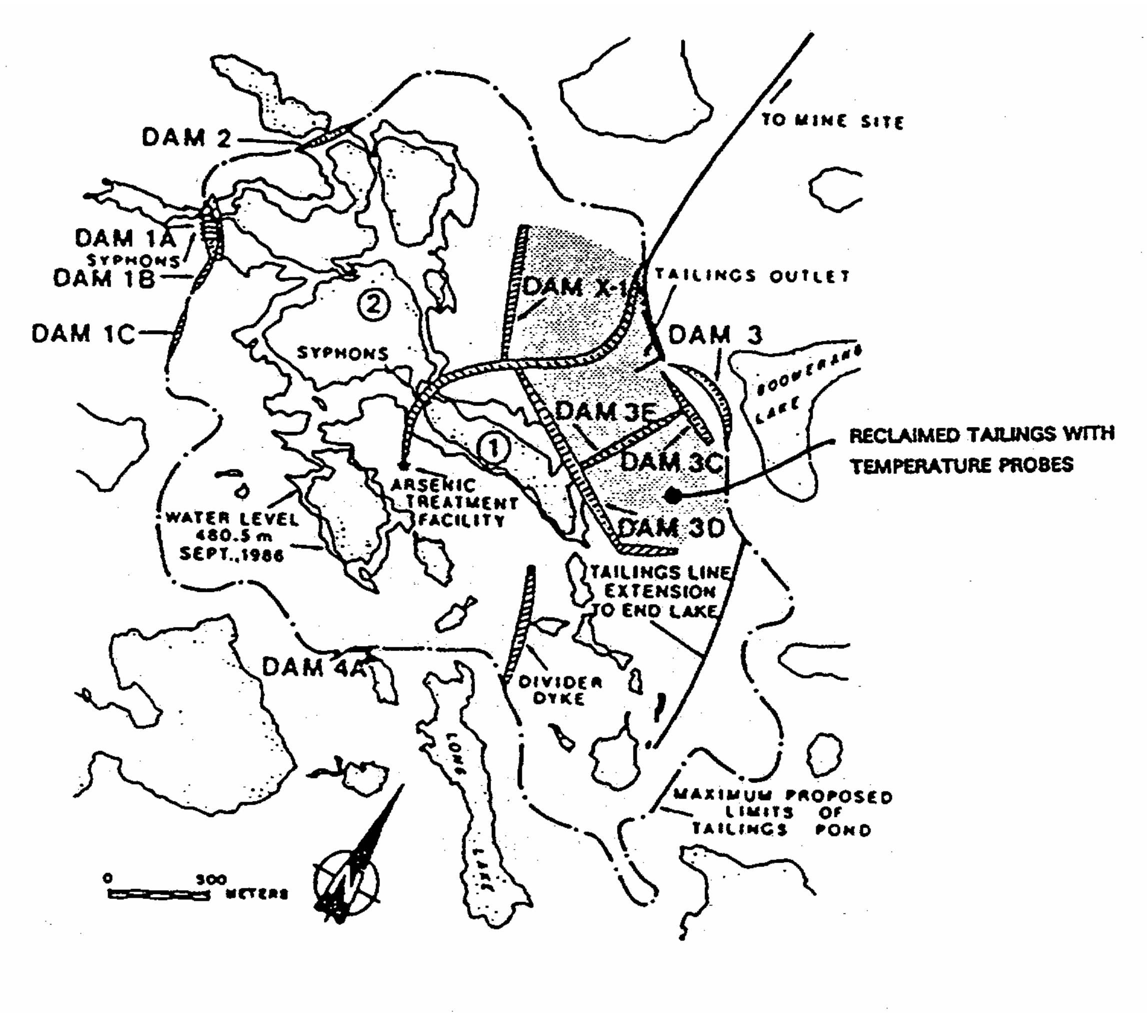
Source: MEND 1.61.2

Figure 4.8-5 **Locations of Case Study Sites**



Source: MEND 1.62.2

Figure 4.8-6 **Lupin Mine Tailings Impoundment**



Source: MEND 6.1 (from Wilson 1989)

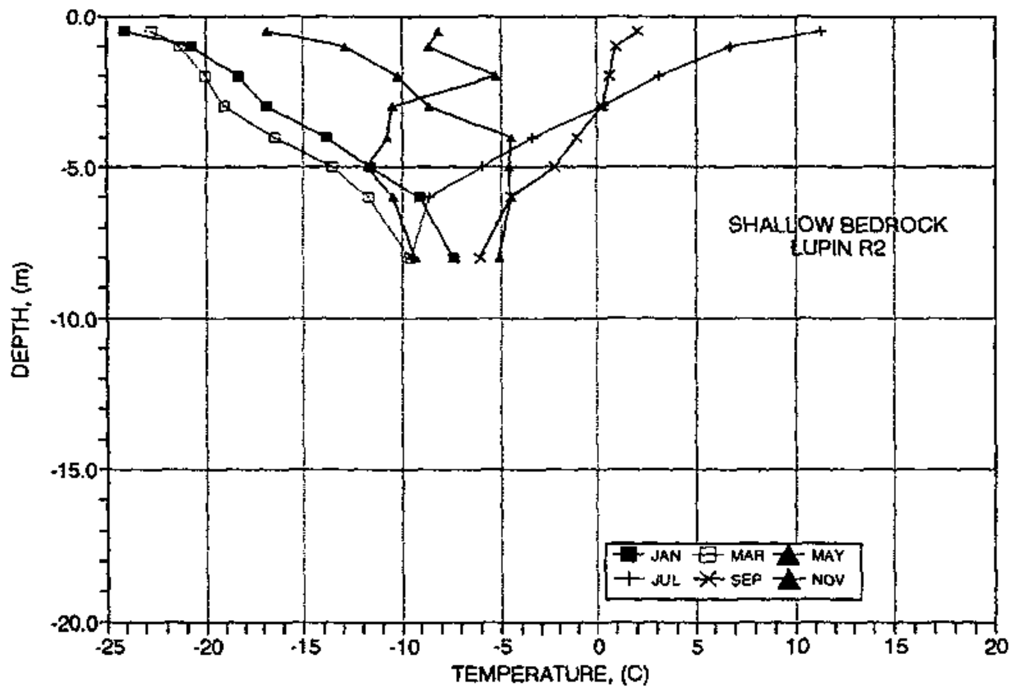
Bimonthly temperature profiles in a reference thermistor string in shallow bedrock are shown in Figure 4.8-7. The profiles show that the active zone in the shallow bedrock is about 3.5 m deep (MEND 6.1; Holubec 1990). Zero amplitude is estimated to occur at a depth of greater than 15 m where the temperature is about -8ºC. Figure 4.8-8 shows temperature profiles through the silty-sand dam fill and into the natural silty-sand and bedrock. The profiles indicate that the temperature at zero amplitude is about -4ºC. The zero amplitude temperature under the dam crest is warmer than the estimated zero amplitude ambient temperature of -8ºC, likely as a result of the warming effect of adjacent ponded water.

Figure 4.8-9 shows temperature fluctuation profiles measured in a reclaimed tailings cell where 4.9 m of tailings had been covered with a 0.6 m thick layer of sandy gravel. The cell is situated just south of Dam 3 (Figure 4.8-6). The temperature profiles show that the active layer is about 2.5 m deep and that the warmest temperature at the bottom of the tailings is about -2.5ºC. MEND 6.1 indicated that a sand/gravel cover with a thickness of about 3 m would be required to keep the tailings permanently frozen and to maintain the active zone above the cover/tailings interface. Due to a lack of available granular materials at this site, a 3 m thick cover would be cost prohibitive to construct. Alternative cover options identified in MEND 6.1 are:

- The construction of a cover using non-acid generating waste rock. In concept, the coarse rock could decrease the cover thickness requirement by allowing heat to escape from the tailings by convection during the winter, and reduce warm weather thawing through the insulating effects of the air trapped in the rock voids.
- To contain surface water within the tailings management area. Total containment could, in concept, be achieved through the use of frozen core perimeter dykes. The development of a total containment facility with frozen core dykes could be developed artificially through the use of thermosyphons to extract heat from the ground during cold weather, and polystyrene board insulation to insulate the ground during warm weather. Total water containment may be feasible at sites where there is excess evaporation in comparison to total precipitation. A total containment facility may also include a tailings surface wind erosion barrier.

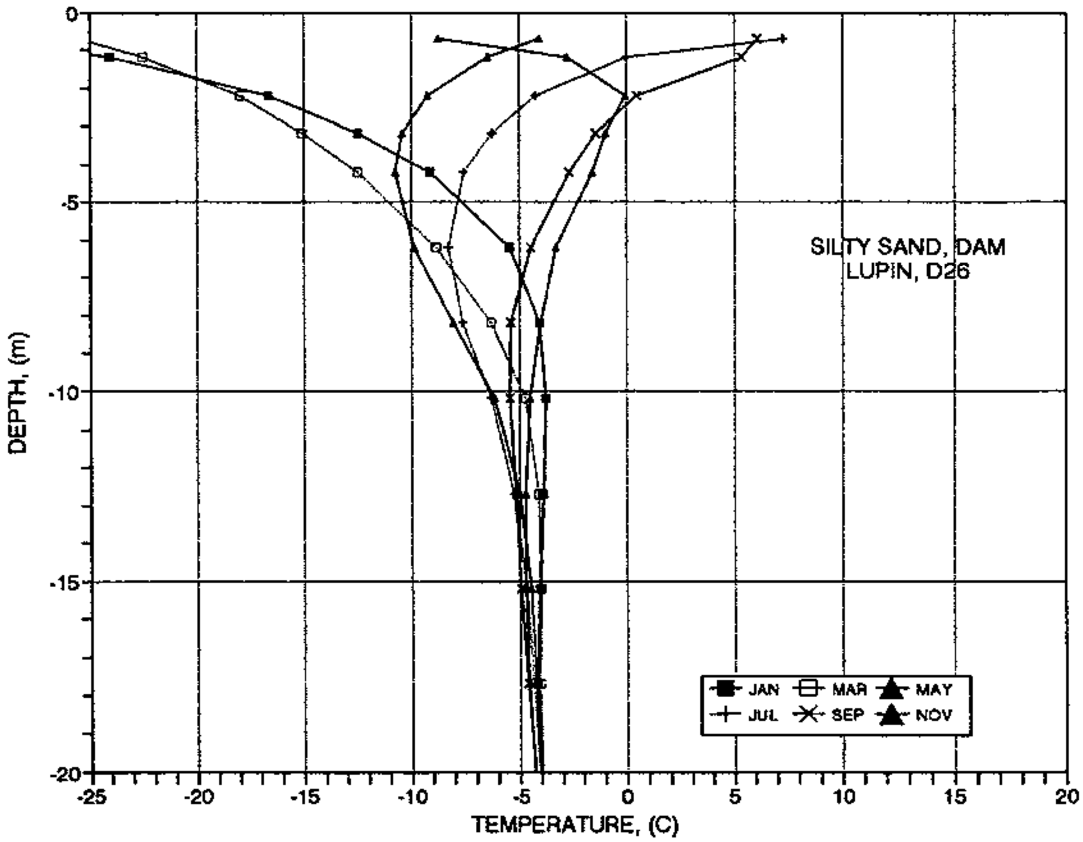
MEND 6.1 reports that it is unlikely that an insulation and sand/gravel cover would be sufficient to develop permafrost conditions in tailings deposited in a discontinuous permafrost region. Insulation is beneficial in that it decreases the rate of heat exchange but it does not change the long-term thermal condition. As such, a heat extraction system would need to be used in combination with insulation to enhance permafrost development. Natural convection devices such as thermosyphons (Figure 4.8-10) are widely used in civil engineering applications to remove heat from the ground. Thermosyphons are more effective in heat transfer than either convection tubes or air convection piles.

Figure 4.8-7 **Reference Thermistor String in Bedrock**



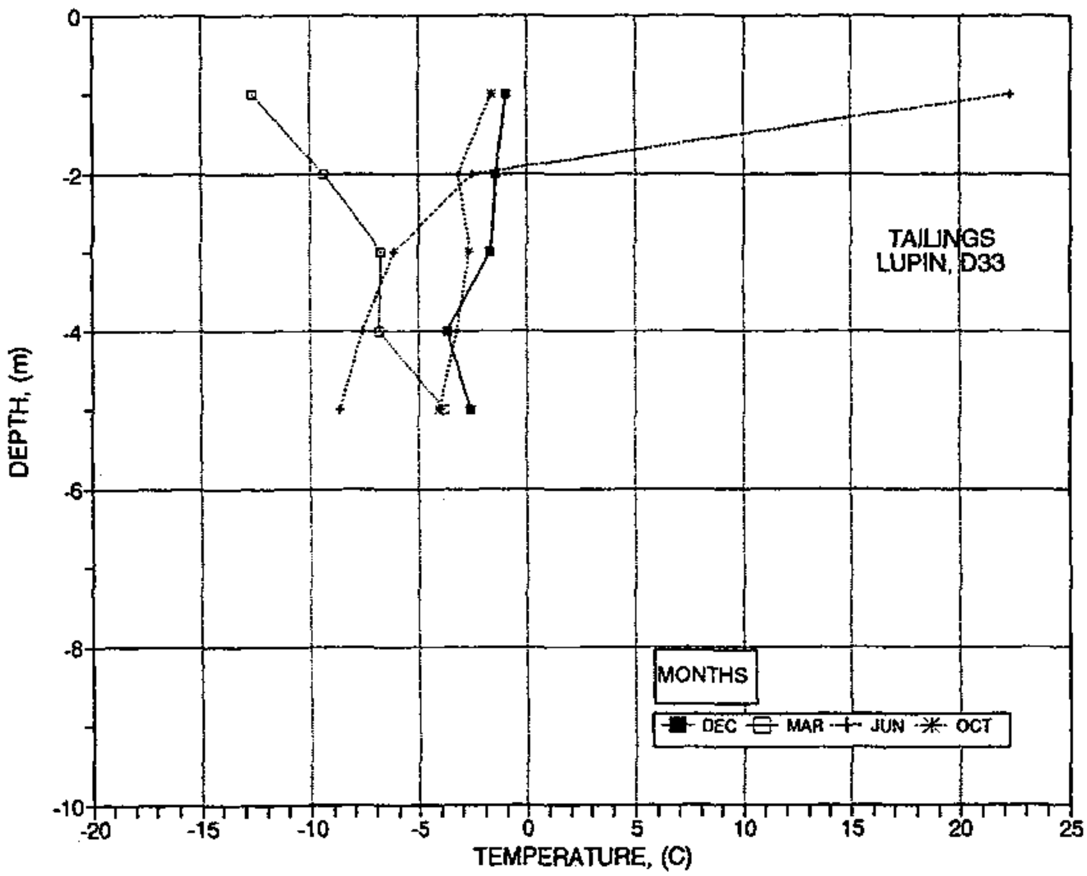
Source: MEND 6.1 (from Holubec 1990)

Figure 4.8-8 **Temperature Profile Fluctuations under Crest of Tailings Dam**



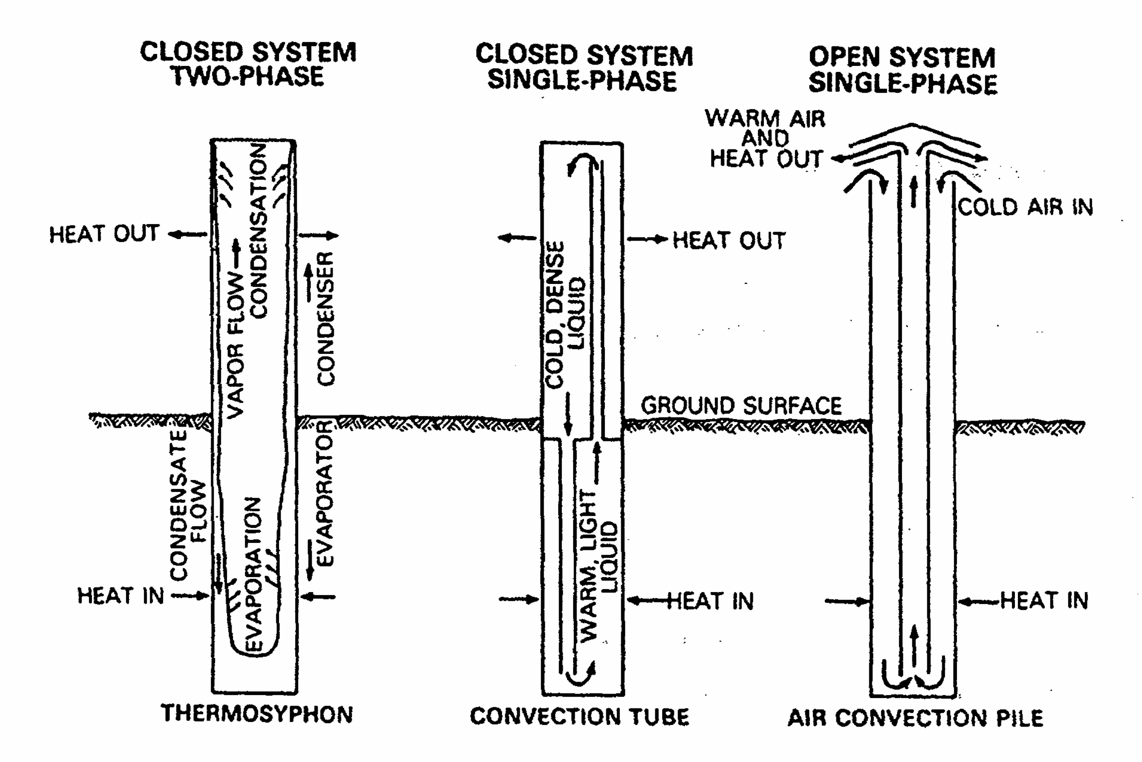
Source: MEND 6.1 (from Holubec 1990)

Figure 4.8-9 **Temperature Profile Fluctuations in Reclaimed Tailings**



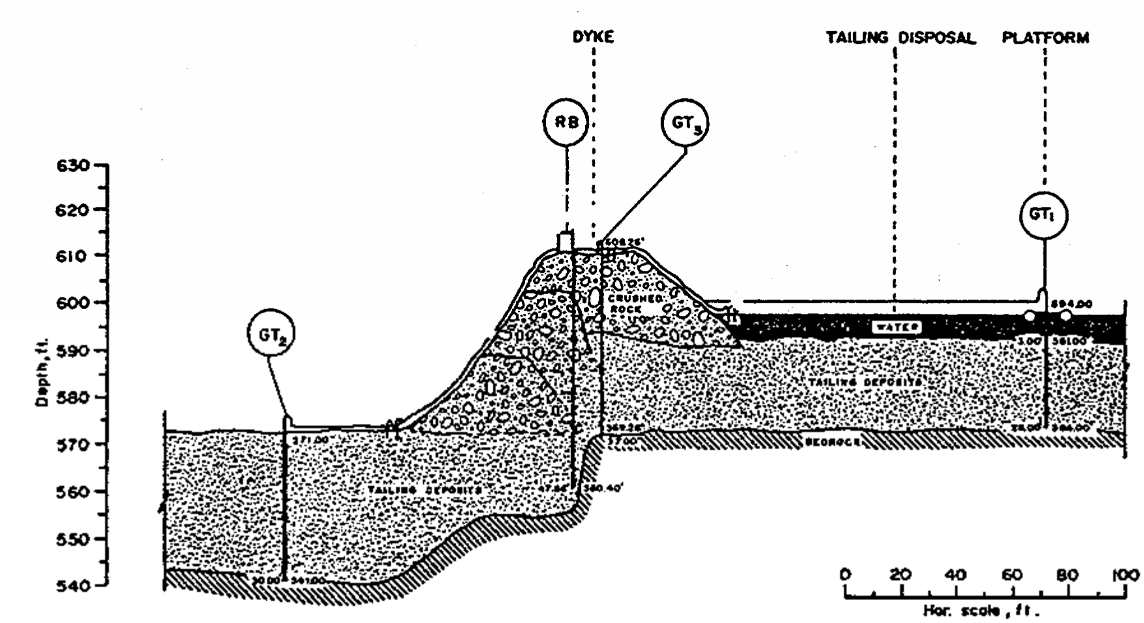
Source: MEND 6.1 (from Holubec 1990)

Figure 4.8-10 **Natural Convection Devices**



Source: MEND 6.1 (from Heuer *et al.* 1985)

Figure 4.8-11 **Giant Mine, Dyke No. 3, Cross Section and Instrumentation**



Source: MEND 6.1 (from Roy *et al*. 1973)

Giant Mine: The Giant gold mine at Yellowknife is situated in the South Boreal discontinuous permafrost region. As part of a 1971 Federal Department of Indian and Northern Affairs study of the stability of two dyke embankments in Yellowknife, thermocouple strings were installed upstream and downstream of the Giant mine tailings containment structure (Figure 4.8-11).

Thermocouple string GT_1 was installed upstream of the dyke in the active tailings disposal area where there was a 1.8 m deep water cover. The distribution of permafrost with temperatures recorded in May and October of 1971 is shown in Figure 4.8-12. The ponded water at GT_1 remained frozen to the bottom from the end of November to the end of May. The temperature monitoring results (Figure 4.8-13) indicate that the top 4 m of the submerged tailings froze to a maximum depth of about 4 m, and that the tailings below a depth of 4 m remained unfrozen throughout the year.

Thermocouple string GT_2 was installed downstream of the dyke in an old tailings area that had a 0.3 m deep water cover. The temperature profile at GT_2 is shown in Figure 4.8-14. The profile indicates that top 6.8 m of tailings thaws and freezes from the ground surface. The top 6.8 m thick layer of tailings was thawed from June to September. The tailings below the 6.8 m depth remained frozen throughout the year at a temperature of between 0 to -1ºC.

Faro Mine: Tailings cover test plots were constructed at the Faro zinc mine in the Yukon Territory. A key program objective was to evaluate the effect of various covers on the tailings temperature. The three test plots included:

- Two composite, convection rock cover test plots consisting of 2 m of tailings overlain by 0.5 m of tailings slimes, 0.5 m of till, and a 0.5 m top layer of waste rock. One of the plots was moisture unsaturated, and the other was saturated; and
- A tailings control plot.

The temperature profiles for the above plots are shown in Figure 4.8-15. The results show that the convective rock cover led to a decrease in the temperature within the tailings. Based on the data for the warmest months (e.g. July and August):

- In the unsaturated test plot temperature profile, the temperature at the tailings slime surface was 6ºC lower than the temperature of the exposed tailings surface Figure 4.8-15b); and
- The saturated test plot temperature profile also had lower tailings temperatures at depth, but the reduction in temperature was not as great as in the unsaturated test plot (Figure 4.8-15c).

A key finding was that the convective rock cover acted as a cold air trap. Cold air entering the cover in the winter cools the tailings, and the cover acts as an insulator during warm weather periods. The limited depth of temperature measurements did not allow a conclusion to be reached concerning the enhancement of permafrost through the use of a rock cover.

The temperature profile in the old Faro mine tailings area, located outside of the convective cover test plot area, was also monitored. The temperature profiles (Figure 4.8-16) show that the tailings remain frozen below a depth of 4 m.

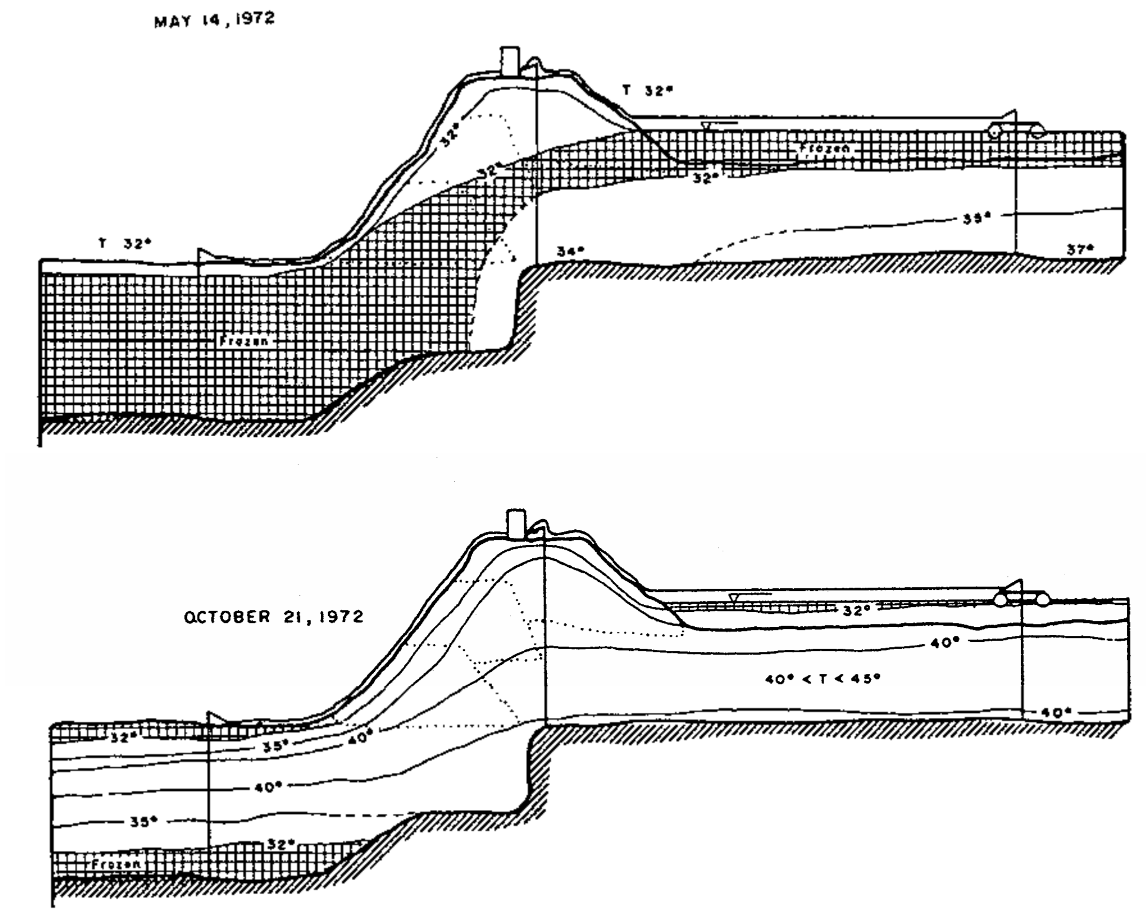
As indicated above, a potential method to develop and maintain permafrost in discontinuous permafrost regions is the use of a convective rock cover. In such an application, heat is extracted from the ground during the winter by air convection, and the air in the rock voids serve to insulate the tailings during warm weather periods. There are documented case histories from Russia and Canada that support this principle (MEND 6.1). However, MEND 6.1 also indicates that the development of permafrost conditions in tailings in the Canadian discontinuous permafrost regions (Boreal and Cordillera permafrost regions) would be highly challenging and maybe practically impossible. Section 7.3 of MEND 6.1 provides a cost comparison of potential tailings cover designs for the Boreal and Cordillera discontinuous permafrost regions. The options considered were: the use of an insulated perimeter dyke and a vertical single line of thermosyphons; the use of an insulated dyke and a horizontal double line of thermosyphons; and the use of a composite convective rock cover. The composite convection cover was marginally more costly than the other options and although it was appealing because of its simplicity, it was a distant choice because of its unproven technology.

Hull *et al*. (1999) reported on the proposed use of paste tailings at the Julietta Gold Project located in the Magadan Province of Eastern Russia. About 55% of the tailings would be disposed on surface as paste tailings, with the balance of the tailings used for underground backfill. It is expected that the surface disposal of paste tailings will reduce the disturbed land area, and that the low moisture content of the paste will allow the tailings to freeze as placement proceeds. The tailings basin would be lined with a geotechnical liner to minimize seepage. The tailings basin is located in a permafrost area with subzero ground temperatures extending to a depth of at least 30 m. Ground temperature profiles for this site suggest that the seasonal thaw layer extends to a depth of 3 to 4 metres.

MEND 1.61.1 Roles of Ice, in the Water Cover Option, and Permafrost in Controlling Acid Generation from Sulphide Tailings

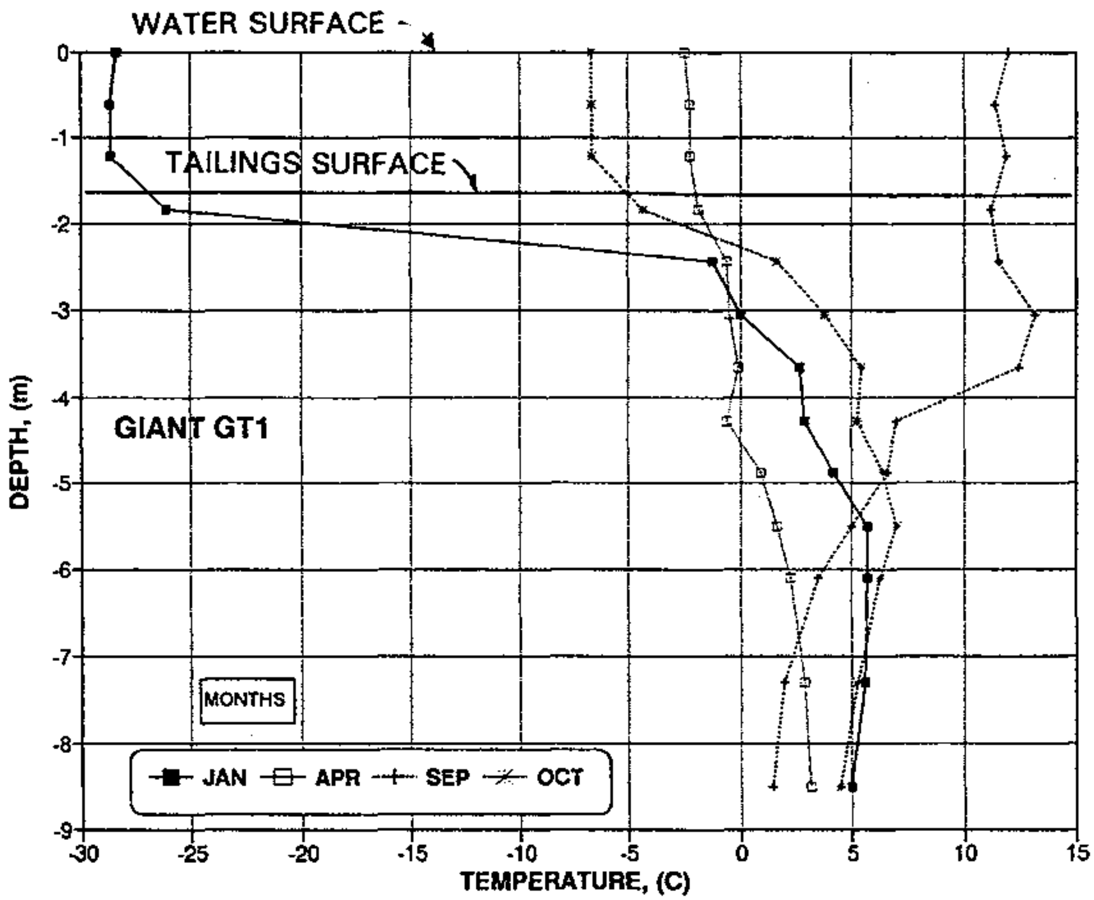
MEND 1.61.1 investigated the effect of ice on a water cover and the potential use of permafrost to control acid generation in tailings. Water covers and engineered dry covers are considered two of the most advanced acidic drainage control technologies available at present. MEND re-assessed the application of these technologies in the north, and determined that the potential exists to use permafrost to control acidic drainage. In particular, the encapsulation of tailings in permafrost, and below the active layer, may ensure *in situ* temperatures at or below 0ºC and reduce the rate of chemical oxidation and bacterially catalyzed sulphide oxidation.

Figure 4.8-12 **Temperature Sections During May and October 1972, Giant Mine**



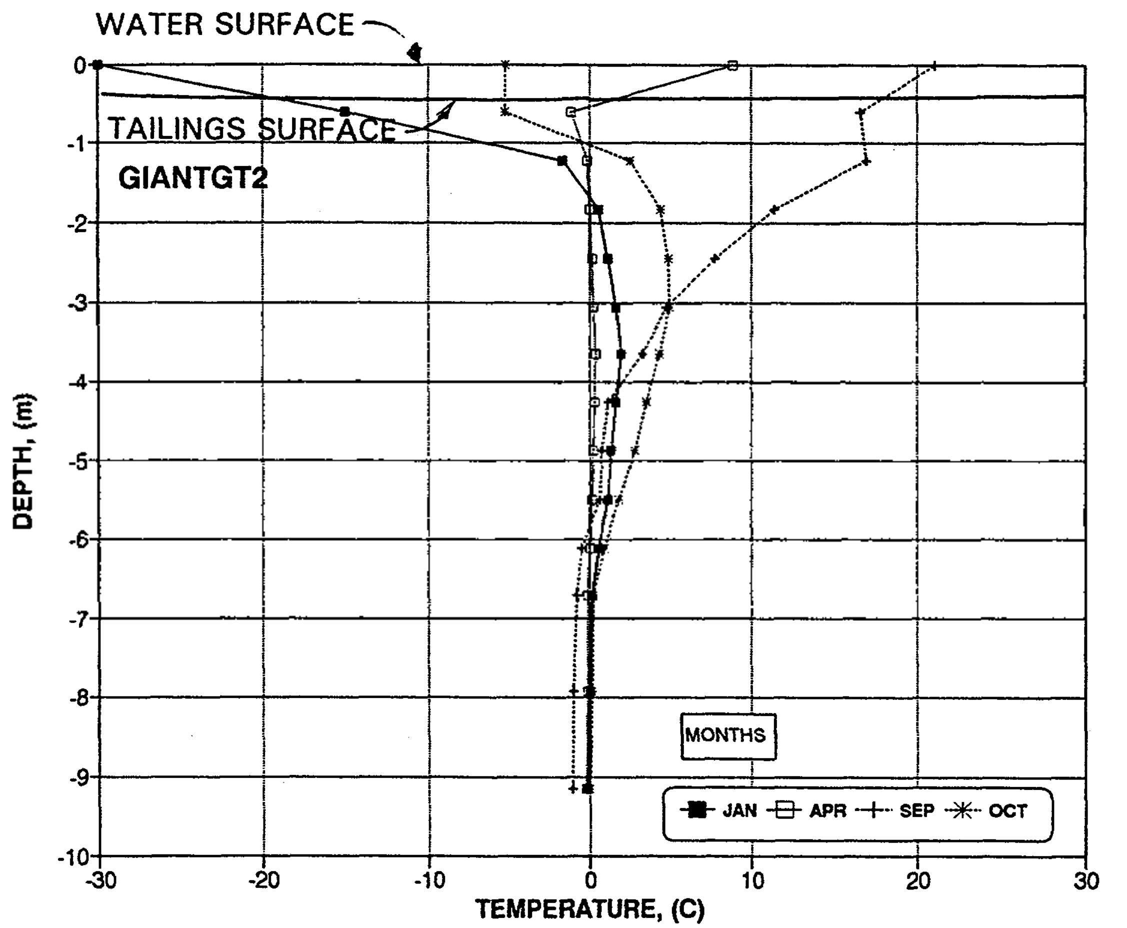
<u>Note:</u> **Temperatures are in degrees Fahrenheit**
Source: MEND 6.1 (from Roy *et al*. 1973)

Figure 4.8-13 **Temperature Profiles at Tailings Disposal Area with 1.8 m Water Cover, Giant Mine**



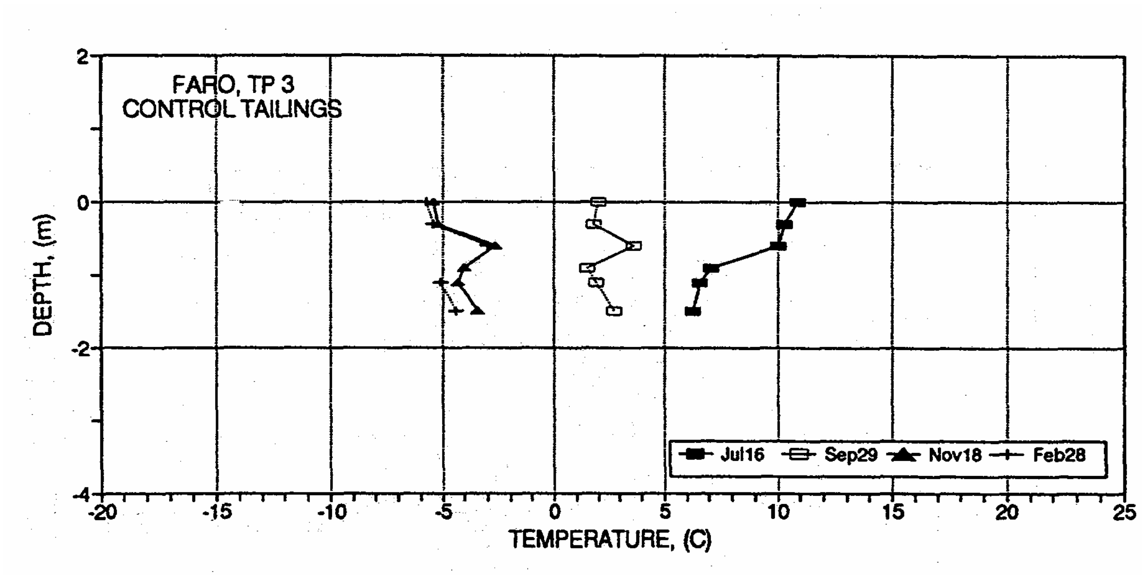
Source: MEND 6.1 (from Roy *et al*. 1973)

Figure 4.8-14 **Temperature Profiles at Old Tailings Area with 0.3 m Water Cover, Giant Mine**



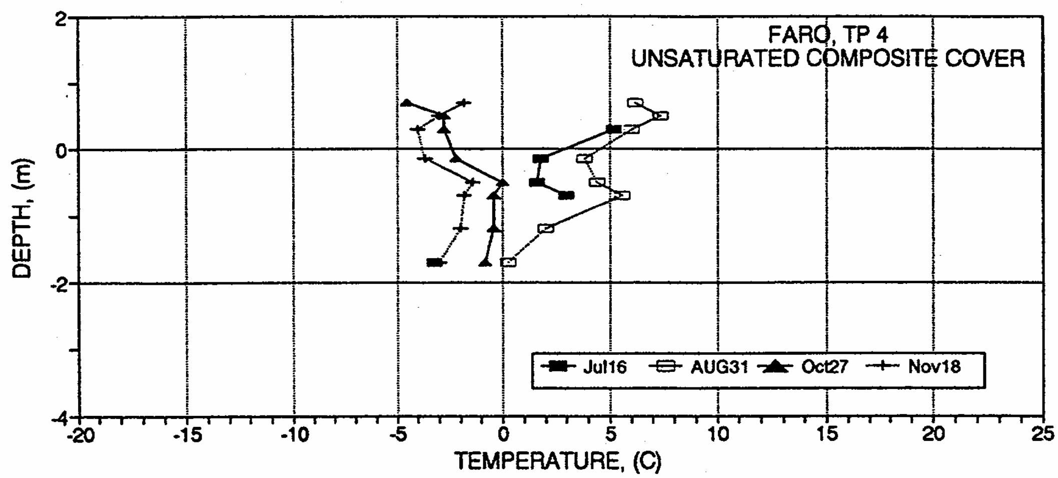
Source: MEND 6.1 (from Roy *et al*. 1973)

Figure 4.8-15 **Temperature Profiles at Tailings Plot, Faro Mine A) Tailings Control Cell B) Composite Cover, Unsaturated C) Composite Cover, Saturated**

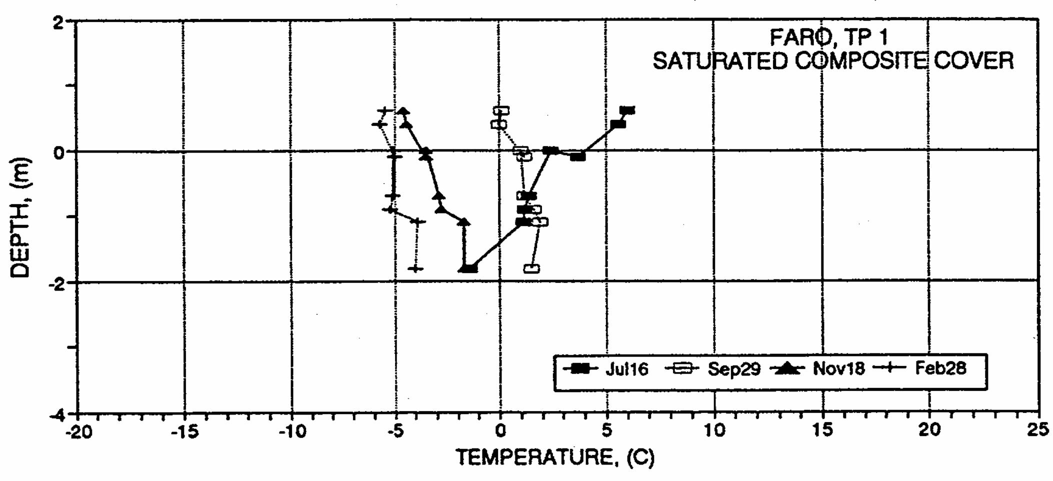


a) Tailings Control Cell

Figure 4.8-15 **Temperature Profiles at Tailings Plot, Faro Mine A) Tailings Control Cell B) Composite Cover, Unsaturated C) Composite Cover, Saturated (continued)**



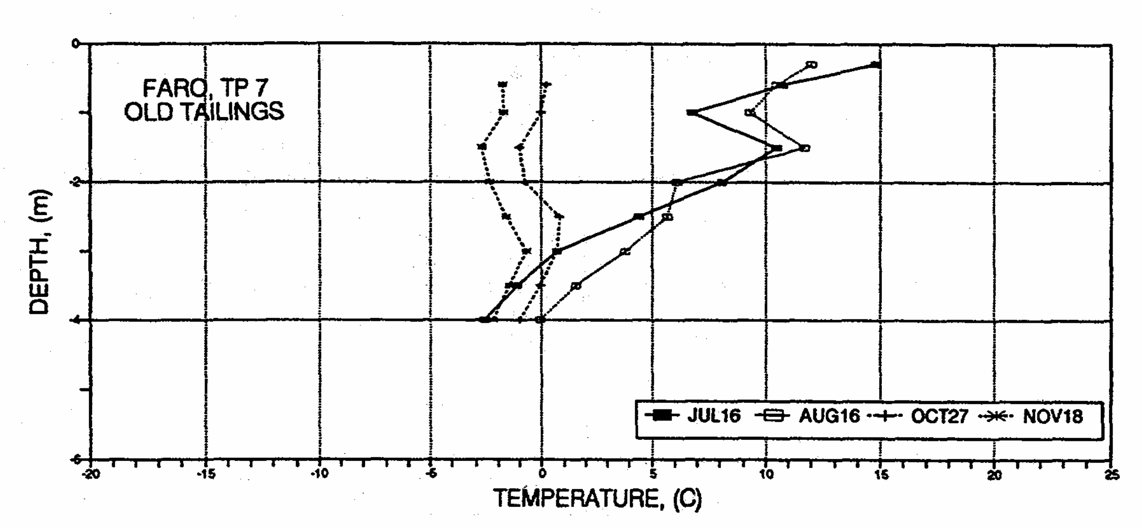
b) Composite Cover, Unsaturated



c) Composite Cover, Saturated

Source: MEND 6.1 (from Robertson and Barton-Bridges 1990)

Figure 4.8-16 **Temperature Profiles at Old Tailings Area, Faro Mine**



Source: MEND 6.1 (from Robertson and Barton-Bridges 1990)

<u>Potential Effect of Cold Climate and Permafrost on Subaqueous Disposal</u>: The key objective of using a water cover in northern Canada is the same as in other parts of the country - to effectively inhibit acid generation over the long term. However, cold climate and permafrost have the potential to influence a water cover as indicated below.

- Oxygen solubility in water increases with decreasing water temperature. The dissolved oxygen concentration ranges from about 8.6 mg/L at 25ºC to a maximum of 14.4 mg/L at 0ºC. MEND 1.61.1 indicates that the increase in the dissolved oxygen content is offset by a reduced rate of sulphide oxidation at low temperatures.
- Water cover ponds may stratify to varying extents as a result of temperature and density contrasts. The surface layer, referred to as the epilimnion, is directly affected by air temperature, wind and solar radiation, and tends to be turbulent and may be highly spatially and temporally variable. The underlying water layer, the hypolimnion, is often only weakly stratified, and can become dissolved oxygen depleted.
- A water cover may turn over prior to freeze-up and after ice-off due in large part to temperature driven, vertical mixing. MEND 1.61.1 indicates that the existence of a strong density stratification or chemocline near the bottom of the water cover may preclude water column mixing.
- A layer of ice over a water cover restricts direct contact between the water cover and the atmosphere. The ice cover may restrict the oxidation reactions during winter months depending upon the volume of the underlying water and the rate of oxygen consumption within the water column. MEND 1.61.1 indicates that the formation of an ice cover over a water cover may lead to oxygen deficient and stagnant water column conditions that can further reduce oxygen influx.
- The presence of ice over a water cover would be expected to reduce disturbances to the water column and the submerged sulphide solids by attenuating wind-induced wave action. Snow cover on an ice layer would be expected to delay the warming of the water column and the melting of the ice cover.

The disturbance of submerged tailings by resuspension and ice scouring can impact the rate of oxidation of the submerged sulphide solids. Aspects which affect the impact of ice scouring on submerged tailings are ice movement, ice strength, the depth of the water under the ice, and the physical properties of the submerged waste or the water cover/tailings interface layer (if present). MEND 1.61.1 reviews factors that can be applied to reduce the impact of ice scouring on subaqueously disposed mine wastes. These factors include minimizing ice thickness and strength, selecting appropriate water cover depths and slopes in shallow water areas, reducing the effect of fetch and wind, and the use of a subaqueous capping layer at the water/tailings interface.

<u>Potential Effect of Cold Climate and Permafrost on Dry Covers:</u> Excluding or reducing water infiltration for acid generation control purposes requires the use of impermeable barriers such as synthetic membranes or composite covers that generally consist of a saturated clay layer over a capillary break. The application of these covers, even when appropriate material is available, tends to be expensive and their long-term rates of degradation due to heaving, subsidence and root penetration is unknown. However, a dry cover is likely one of the best solutions for dealing with acidic drainage and metal leaching in the arctic.

In parts of the Canadian north where the annual evapo-transpiration exceeds the annual precipitation, a negative net influx of water from a cover is not in itself sufficient to control acid generation. As such, the exclusion of oxygen is probably the most effective long-term generation control technique. As indicated in the discussion regarding the effect of low temperatures on water covers, potentially available dissolved oxygen concentrations increase with decreasing temperature and reach a maximum near 0ºC. The diffusive transport of dissolved oxygen can lead to sulphide oxidation, however, its availability may not be rate limiting. Tributsh and Gerischer (1976) concluded that sulphide oxidation can proceed in a series of chemical steps or can include electrochemical steps. The electrochemical oxidation of sulphides in a permafrost environment may only require an inert wick that electrically connects sulphides at depth to an oxygen richer near-surface environment (Cameron 1979). In a permafrost environment, sulphide minerals may act as conductors with unfrozen water present as interfacial films in the permafrost and in the active layer when thawed. At present, no known comparison of the significance of the electrochemical, chemical and biochemical oxidation rates of sulphides has been completed.

The present technical knowledge of permafrost is sufficient for the design and construction of tailings management areas (TMA) over continuous permafrost. These facilities typically involve the use of frozen core containment structures that are effective in preventing seepage losses. Tailings placed in TMAs will be subject to freezing during cold weather and active layer thawing during the summer. Available data suggest that the *in situ* freezing of tailings within an engineered TMA is beneficial because at 0ºC: i) the chemical oxidation rate is reduced by about an order of magnitude in comparison to the oxidation rate at 25ºC; and ii) the biological sulphide oxidation process is inhibited.

Acid generation would be expected to occur in the near-surface zone of tailings due to oxygen diffusion. As such, the active layer of tailings deposited over permafrost conditions would likely represent the acid generation capacity of the contained tailings.

Ideally, the bulk of stored tailings should remain frozen over time and measures should be taken to prevent the cyclical thawing of surface tailings. MEND research has led to the development of promising potential methods that may allow best use of permafrost conditions to inhibit acid generation over the long term. There have, however, been few detailed field investigations into the natural freezing of acid generating wastes. The need to improve upon the present database is compounded by the range of conditions encountered in continuous and discontinuous permafrost regions. The large-scale and in-field complexities of permafrost migration into acid producing materials is one area where additional research is required.

Effects of Permafrost on Encapsulated Tailings: Permafrost offers generally suitable conditions for the encapsulation of tailings in frozen ground, where the *in situ* temperatures are 0ºC or less. In such cases, disposed mine wastes could be covered by a sufficient depth of material to allow the raising of the permafrost table to above the waste/cover interface, and maintain the macro temperature of the waste at or below freezing. This approach is however, not universally applicable and is dependent on site-specific permafrost, hydrologic, hydrogeologic, waste and climatic conditions.

As indicated in MEND 1.61.1, a small but significant percentage of pore water below the permafrost table remains unfrozen due to capillary and adsorption forces and a depressed freezing point due to dissolved contaminant concentrations. Permafrost has a finite permeability and reduces, but does not entirely prevent pore water mobility. The migration of oxidation products would be expected to be limited at cold climate sites with low or net negative annual precipitation. It has been postulated that at some sites, waste encapsulation may not be sufficient as a stand alone acidic drainage control measure, and could require the additional use of an oxygen diffusion barrier. The design of a cover layer, whether for oxygen diffusion or as an insulating thermal layer would need to consider the effects of frost heaving and the growth of ice lenses and other features causing frost heaving.

The encapsulation of wastes and the use of permafrost are viewed as promising technologies for the control of sulphide oxidation in the northern environment, but not an absolute control for acidic drainage generation. Proposed strategies for tailings disposal in permafrost regions are listed in Table 4.8-2.

MEND 1.61.2 Acid Mine Drainage in Permafrost Regions: Issues, Control Strategies and Research Requirements

MEND 1.61.2 investigated acidic drainage generation in a permafrost environment, and reviewed the conceptual aspects of engineered control strategies that could potentially be applied to take advantage of cold temperatures and frozen ground conditions. The study involved the review of computer databases and MEND studies, and discussions with personnel from a number of northern mines. The following discussion of freeze control and climate control strategies is reiterated from MEND 1.61.2.

Table 4.8-2
Proposed Strategies for Tailings Disposal Options in Permafrost

Strategy	Options of Variations	Issues for Consideration
Continuous Permafrost		
Encapsulation of tailings in perennially frozen ground by the addition of sufficient non-acid generating material as a cover in order to raise the permafrost table to the tailings/cover interface	• Sandy gravel covers • Use of alternate cover materials such as insulation and organic matter • Use of a capillary break in a composite cover to minimize frost heave potential • Deposit and freeze thin layers of tailings within perimeter dykes • Timing of the placement of cover material over tailings	• Impact of electrochemical, chemical and biological oxidation processes • Depth and variability of the active layer and effect on permafrost table position and local hydrology • Availability of cover materials • Impact of snow cover • Total freezing layer thickness in comparison to the thickness of thin layers of tailings • Minimization of frost heaving and its impact on the cover and the tailings
Containment of tailings (and reaction products) through the use of impermeable frozen core perimeter dams	• Maximize the percent solids of the tailings • Use of a wind erosion cover • Sequence for the construction of perimeter dykes and tailings deposition • Use of insulating material or thermosyphons to raise the permafrost table in dykes • Minimize snow deposition in the early stages of construction and deposition to enhance freeze back	• Impact of frost weathering on mineral grains • Site water balance and impact on dam stability and erosion • Position of frozen core in dykes relative to the water level in the contained wastes • Dyke height and stability • Rate of oxidation • Long-term impact of contaminated, contained pore water on freezing point depressions • Depression of permafrost table where the contained water does not freeze to depth in winter
Discontinuous Permafrost		
Containment of tailings using artificially frozen core perimeter dams	• Use of insulated dykes and thermosyphons • Use of a convective rock cover	• Sensitivity of permafrost regime to variations in climate or surroundings • Long-term maintenance of artificially developed permafrost

Sources: MEND 1.61.1 (with references to MEND 6.1; MEND 1.61.2)

1. Freeze Control for Tailings

Freezing immobilizes the pore fluids that control reactions and the migration of contaminants. The successful application of freeze control strategies appear to be a viable but unproven strategy that takes advantage of permafrost conditions. Two potential strategies are examined below: total freezing and perimeter freezing. Table 4.8-3 provides a summary of each strategy for discontinuous and continuous permafrost regions.

Table 4.8-3
Tailings Freeze-Controlled Acidic Drainage Strategies

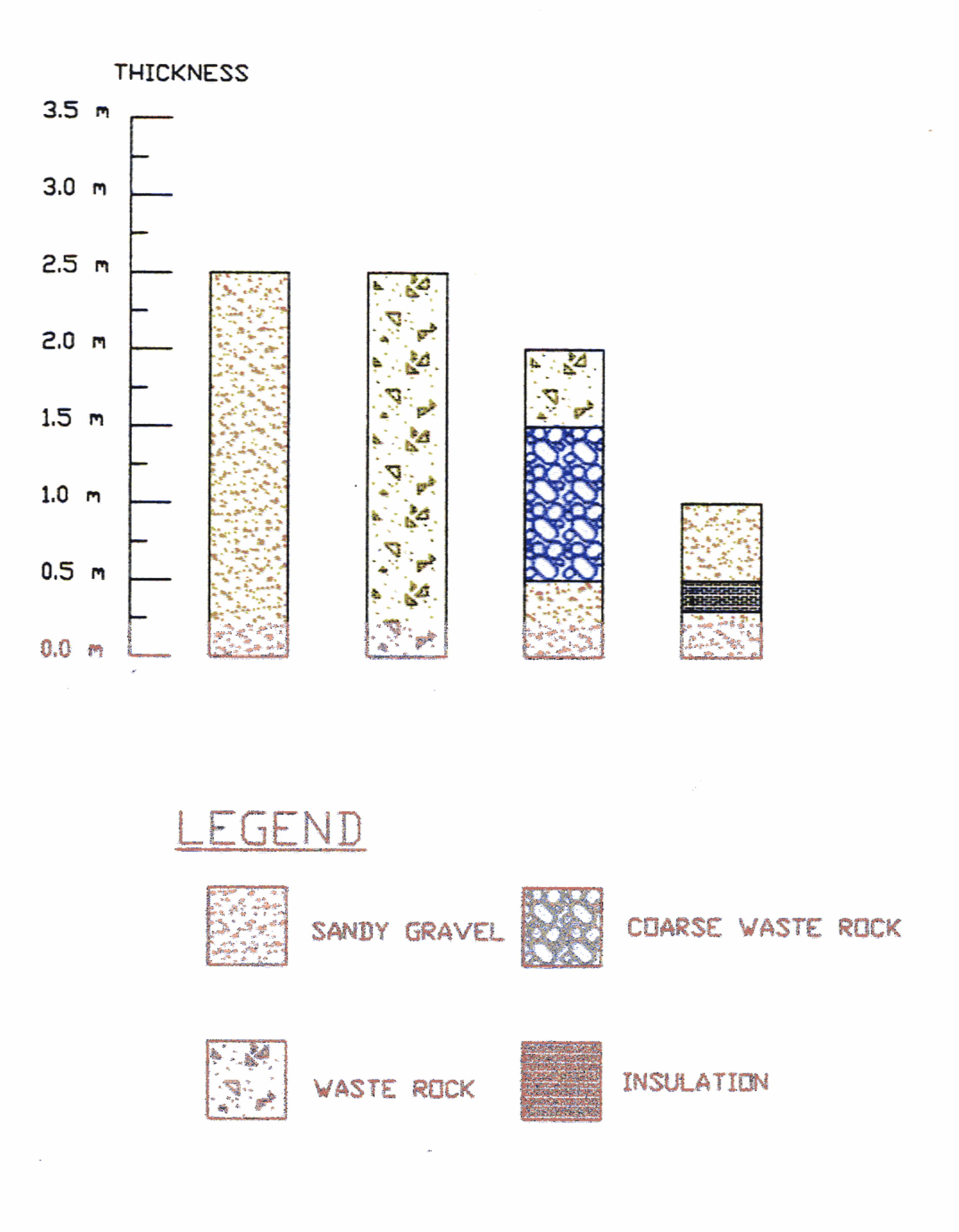
Freeze Control Strategy	Discontinuous Permafrost Regions	Continuous Permafrost Regions
Total freezing	It is difficult to develop freezing conditions in discontinuous permafrost regions.	The use of thin-layered freezing may reduce containment design.
Perimeter freezing	Thick covers or enhanced freezing techniques (e.g. thermosyphons) are required.	The technology for the design and construction of frozen dams is established.

Source: Modified from MEND 1.61.2

Total Freezing: Total freezing entails freezing the tailings mass to reduce or virtually eliminate the rate at which acidic drainage is generated and to reduce the seepage fluxes of contaminants. Total freezing is not considered a viable strategy in discontinuous permafrost regions as ground temperatures are not cold enough to develop permanent frozen conditions. In continuous permafrost, total freezing could be developed after the containment area is filled or during filling by freezing thin layers of tailings.

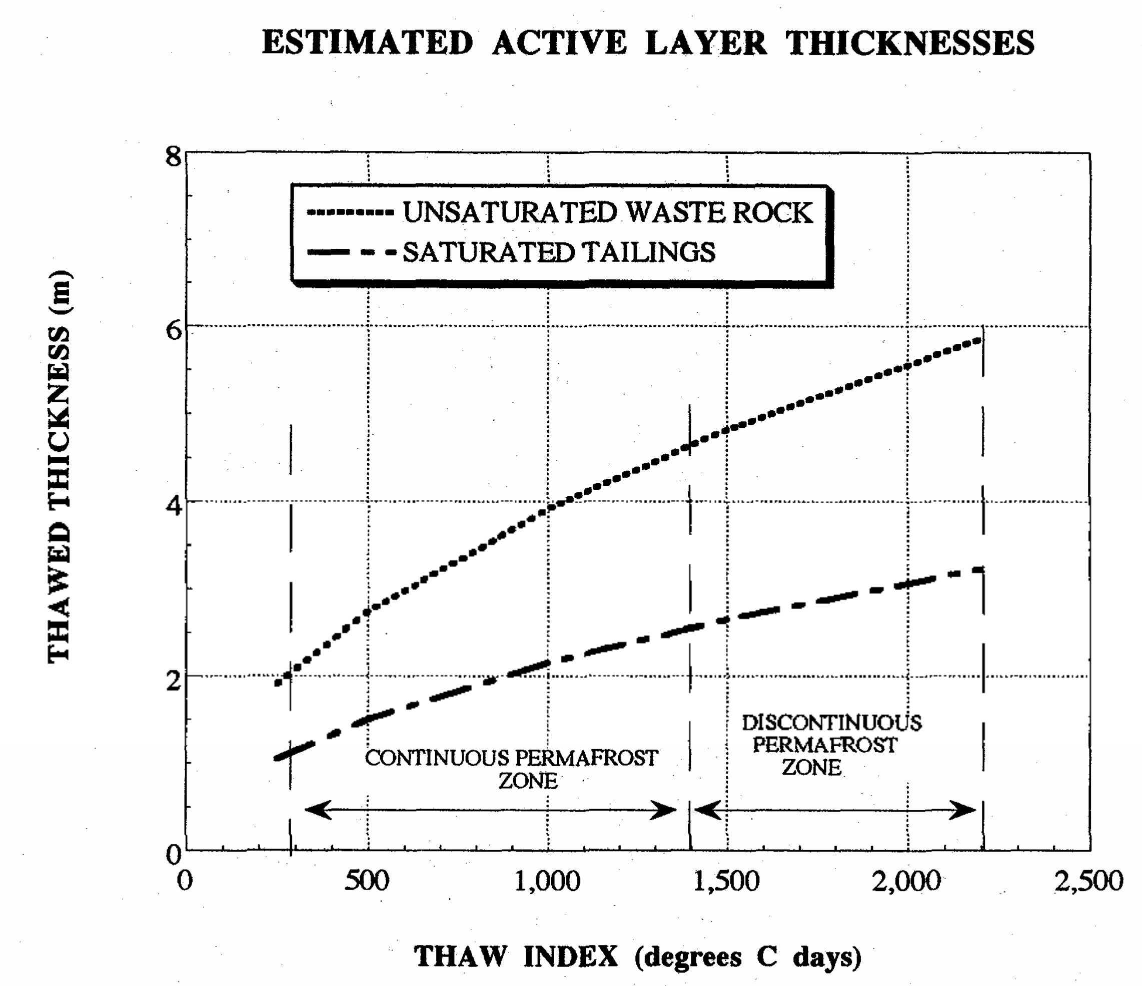
Figure 4.8-17 shows cover options to keep the active zone above frozen tailings. The thickness of a cover to keep tailings frozen would generally be expected to be in the 2 to 4 m range depending on cover saturation and climate (Figure 4.8-18). A potential benefit of using coarse rock as an insulating medium is that the coarse mine rock may serve to remove heat from the tailings by convection during the winter, and reduce the depth of thawing during the summer through the insulation of the trapped air. Historic information on the use of coarse rock as an insulating medium is lacking although it is noted that coarse rockfill and wooden shields have been used as insulation for protecting frozen dams in Russia from summer thaw. At the Portovy Creek dam in the Western Russian Arctic (mean annual air temperature of -10.5ºC), rockfill assists in maintaining frozen conditions (MEND 1.61.2).

Figure 4.8-17 **Cover Options for Total Freezing in Continuous Permafrost Regions**



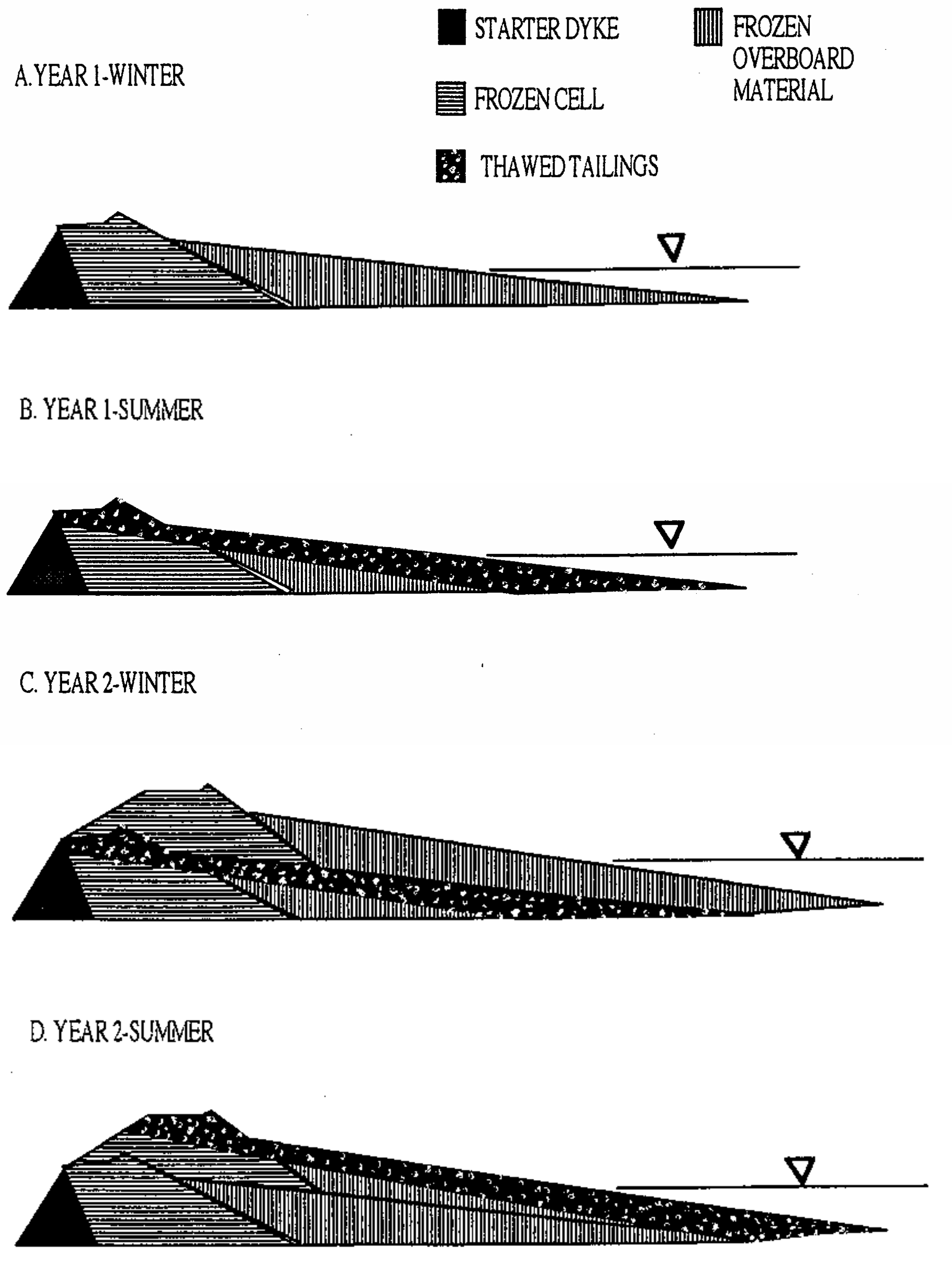
Source: MEND 1.61.2

Figure 4.8-18 **Seasonal Thawing Thickness for Mine Waste in Permafrost Regions**



Source: MEND 1.61.2

Figure 4.8-19 **Tailings Thin Layered Freezing Concept**



Source: MEND 1.61.2

A tailing management area designed for frozen containment could take advantage of freezing many thin layers rather than one thick layer to maximize the total frozen thickness (Figure 4.8-19). Experience suggests that there will be a maximum thickness that can be frozen in a season. This will be dependent upon conditions during placement and pore water content.

Perimeter Freezing: Perimeter freezing involves the construction of a frozen core perimeter dam to contain tailings and prevent lateral seepage. Experience suggests that perimeter freezing is best applied in continuous permafrost regions. In discontinuous permafrost regions freezing may likely need to be enhanced through the use of thermosyphons or insulation.

2. Freeze Control for Waste Rock

Presently there is limited information on the acidic drainage development and strategies employed for waste rock with acid generation potential that are stored in permafrost regions. Research has shown that the oxidation process decreases considerably as the waste rock temperature approaches 0°C and most likely any oxidation below 0°C may not be significant enough to impact seepage water (Holubec 2000).

The analyses and design of closure approach for waste rock dumps in the 'warm' permafrost regions (mean annual air temperature warmer than about -7°C) and 'cold' permafrost regions (mean annual air temperature colder than about -7°C) are quite different.

In 'warm' permafrost, it can be anticipated that the air temperature will reduce the oxidation and therefore the leaching rates. Otherwise, the closure design approach is similar to southern areas.

In 'cold" permafrost, the closure design can consider that permafrost will develop within the waste rock unless the oxidation rate is so high that it will produce great heat which cannot be compensated by the cold air temperatures. This could exist in waste rocks with high sulphur content. Presently there are no research or field observations available that could provide guidance.

However in both 'warm' and 'cold' permafrost, the surface layer of the waste rock, that is likely between 3 to 4 m thick, will warm above freezing during the summer months. This layer will oxidize and leach metals during the summer unless closure measures are taken.

The Diavik Diamonds Project involves some waste rock that contains biotite schist with sulphur. The initial closure plan submitted to the regulatory authorities proposes to cover the waste rock dump with a frozen cover. The cover will consist of 3 m thick clean rock layer that is underlain by 1.5 m of a silty sand layer. The concept is to use the clean rock to absorb the major depth of the active zone and have the majority of the sandy silt remaining permanently frozen. It is predicted that the active zone will not penetrate more than 0.5 m into the water/ice saturated silty sand and thereby leaving a 1 m thick frozen silty sand zone. This 1 m thick frozen zone will shed water and provide an oxygen barrier (Holubec 2000).

Extensive thermal modelling of the waste rock dump indicates that the central portion of the waste rock will be at or just below zero degrees. At this low temperature and the low sulphur content of the waste rock, it is unlikely that rock oxidation will have any influence on the rock pile internal temperature. Temperature and seepage monitoring during the construction of the piles and the construction of large-scale test piles will verify this.

It is noted that the mean annual air temperature at the Diavik Diamond Project site is about -11°C, which is about average for many of the mines that operate or are being developed in the cold permafrost region south of the Arctic Sea. It is not certain that a similar design approach can be taken for mines located in the warm regions, such as Yellowknife, NWT.

Freeze-control strategies identified in MEND 1.61.2 for waste rock disposal in discontinuous and continuous permafrost regions are summarized in Table 4.8-4.

Table 4.8-4

Waste Rock Freeze-Controlled Acidic Drainage Strategies

Freeze Control Strategy	Discontinuous Permafrost	Continuous Permafrost
Freeze Control	<u>Layered freezing</u> Requires special consideration for the insulation of the underlying permafrost	<u>Layered freezing</u> Requires non-acid waste rock insulating cover
Climate Control	<u>Segregate fine and coarse waste rock</u> Main impediment is lack of understanding of mass transfer and geochemical processes within waste piles	

Source: MEND 1.61.2

In discontinuous permafrost, a frozen perimeter cannot be practically constructed without resorting to artificial freezing (e.g. thermosyphons) with insulation. Thermosyphons may not be suitable for long-term conditions due to maintenance requirements.

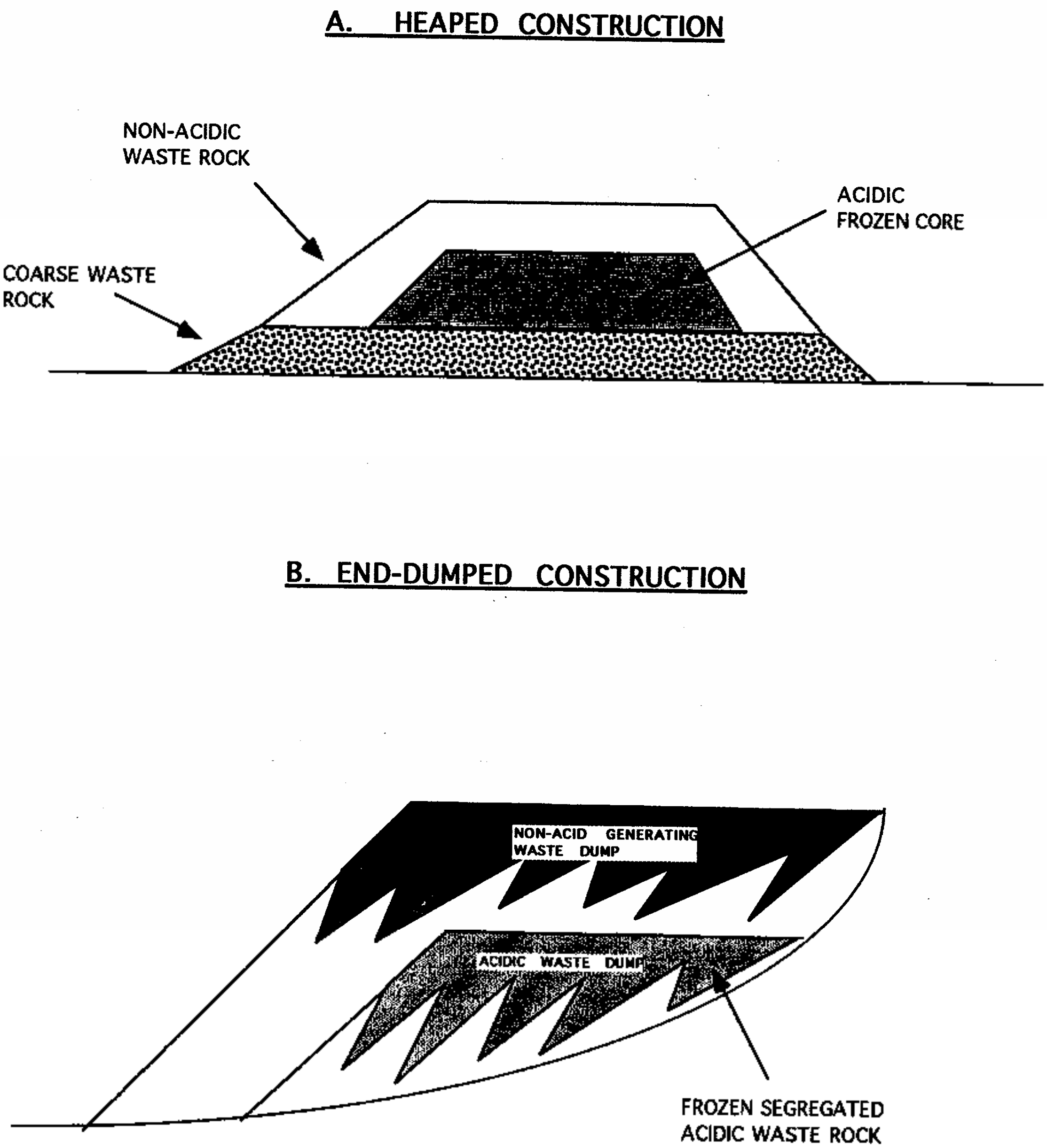
Figure 4.8-20 shows possible concepts for the freeze control of waste rock. In heaped dump construction (Figure 4.8-20a), where the dump is raised in lifts, acidic frozen material could be encapsulated in non-acidic material for insulation protection. In discontinuous permafrost regions, a coarse convective insulating blanket could be placed at the base of the pile to protect the underlying foundation from thaw degradation. A similar approach could be applied for end-dumped construction. Figure 4.8-20b shows acidic waste, frozen in layers parallel to the dump face and encapsulated by non-acidic material.

The potential exists to develop freeze controlled and climate controlled strategies for waste rock. Factors such as water flow, airflow, geochemical processes within a waste rock dump, and the influence on the underlying permafrost would need to be assessed on a site-specific basis.

While reliable control strategies cannot be advanced based on average climatic factors, there may be site-specific opportunities to use the permafrost climate to mitigate acid generation. As examples, consider the potential strategies outlined below:

- Finer material placed over-top of a dump could serve to significantly reduce convective airflow. The permeability of air in a sandy gravel material at the field capacity moisture content could, in some cases, reduce oxygen by several orders of magnitude compared to unsaturated rock-like material commonly found at the base of waste rock piles. If the layer is thick enough (e.g. > 2 m) most of the evaporation would be expected to occur in the upper portion of the fine-grained material.
- Zoning of coarse material in a waste rock dump could be used to advantage. Coarse material could be used for drainage control where high seepages are expected. At low moisture fluxes, coarse material could be used as a capillary barrier.

Figure 4.8-20 **Freeze Controlled ARD Control Concepts for Waste Rock.**
A) Heaped Construction B) End-Dumped Construction



Source: MEND 1.61.2

MEND 1.61.3 Column Leaching Characteristics of Cullaton Lake B and Shear (S) – Zones Tailings Phase 2: Cold Temperature Leaching

In MEND 1.61.3, cold temperature column leaching tests were conducted at 2ºC and 10ºC on tailings samples from the Cullaton Lake B and Shear (S) Zones to evaluate oxidation and leaching characteristics at lower ambient temperatures. The testing was an extension of a leaching study at 25ºC completed by Davé (1991). The Cullaton Lake gold mine is located in the Keewatin sub-district of Nunavut, and 416 km north of Churchill, Manitoba. The site is at the tree line and in the zone of continuous permafrost.

The low temperature studies were designed to simulate conditions similar to those expected in the Cullaton Lake tailings after the completion of decommissioning works including the capping of the tailings with approximately 1.5 m of waste rock and overburden to promote freezing and permafrost conditions within the tailings. It was expected that during the frost-free period, some of the tailings could thaw to temperatures of 0ºC to 10ºC. Selected results of the cold temperature column testing program are presented below.

- Oxidation and acid generation occurred in both the B and S Zone tailings at the low (2ºC) and intermediate (10ºC) temperatures. The rate of acidic generation was low and the occurrence of acidic drainage was delayed at these temperatures in comparison to the column test completed under 25ºC temperature conditions.
- The B-Zone tailings (2.31 wt% S) have a neutralization potential of 45.36 kg $CaCO_3$/t and a net neutralization potential of –26.84 kg $CaCO_3$/t. The acid generation rate of these tailings was low at 2ºC. A degree of acid neutralization was achieved and acidic drainage was prevented over much of the leaching test period. Towards the end of the two-year test period, the overall impact in terms of total acidity and effluent metal loading was low. In the 10ºC leach test, the concentrations of acidity, iron and several metals peaked after only two months of leaching. Selected data are presented in Table 4.8-5.
- The S-Zone tailings (0.4 wt% S) have a neutralization potential of 2.0 kg $CaCO_3$/t and a net neutralization potential of -10.5 kg $CaCO_3$/t. In the 2ºC leach test, the acid generation rate of these tailings was initially low but rose as the available alkalinity was consumed. Similarly for the 10ºC test for the B-Zone tailings, the effluent concentration peaks occurred within a few months of the start of the test. Selected data for the drainage from the S-Zone test are presented in Table 4.8-6. MEND 1.61.3 reports that in contrast to what was observed in the B-Zone tailings test, no significant reduction in acid generation was observed during the leaching of the S-Zone tailings at 2 and 10ºC, reflecting the low available alkalinity.

Table 4.8-5
Cold Temperature Leach Test Data
Cullaton B-Zone Tailings Drainage Characteristics

Drainage Parameter	Drainage Effluent Data Obtained Near the End of a Two Year Leach Test at 2ºC	Drainage Effluent Data Obtained Two Months After Start of Leach Test at 10ºC
pH	~6.6	~5.0
Acidity	~ 300 mg/L $CaCO_3$/L	~850 mg/L $CaCO_3$
SO_4	~ 2000 mg/L $CaCO_3$/L	~3,500 mg/L $CaCO_3$
Fe	~150 mg/L	~450 mg/L
Ca	~500 mg/L	~500 mg/L
Mg	~100 mg/L	~100 mg/L
Al	~0.15 mg/L	~0.2 mg/L
Mn	~20 mg/L	~22 mg/L
Cu	Below Detection Limit	Below Detection Limit

Table 4.8-6
Cold Temperature Leach Test Data
Cullaton S-Zone Tailings Drainage Characteristics

Drainage Parameter	Drainage Effluent Data Obtained at Middle of a Two Year Leach Test at 2ºC	Drainage Effluent Data Obtained About Two Months After Start of Test at 10ºC
pH	~3.0	~3.0
Acidity	~ 600 mg/L $CaCO_3$/L	~650 mg/L $CaCO_3$
SO_4	~ 700 mg/L $CaCO_3$/L	~1,500 mg/L $CaCO_3$
Fe	~175 mg/L	~225 mg/L
Ca	30 mg/L	~500 mg/L
Mg	10 mg/L	~100 mg/L
Al	~25 mg/L	~25 mg/L
Mn	~30 mg/L	~30 mg/L
Cu	~1.2 mg/L	~1.2 mg/L

MEND 1.62.2 Acid Mine Drainage Behaviour in Low Temperature Regimes – Thermal Properties of Tailings

MEND 1.62.2 involved laboratory research on the thermal properties of saturated tailings. The testing focused on determining the unfrozen water content and the thermal conductivity of saturated tailings samples. The unfrozen hydraulic conductivity and the amount of unfrozen water provided an indication of the order of magnitude decrease in hydraulic conductivity after freezing.

The tailings samples were obtained from two sources: the inactive Wellgreen nickel mine located in a discontinuous permafrost zone in the Yukon, and the Lupin gold mine located in continuous permafrost in the Northwest Territories. For pore water, one sample of tailings water was obtained from the Lupin mine, and two samples (designated "A" and "B") were obtained from the Anvil lead-zinc mine in the Yukon. The test program involved the testing of the four solids listed in Table 4.8-7.

Table 4.8-7
Characteristics of Solids Tested

Tailings Solids Sample	**Specific Gravity**	**Paste pH (pH units)**	**Total Sulphur (%)**	**Acid:Base Accounting Data**	
				NNP	**NP/AP**
Wellgreen slab [(1)]	3.16	3.2	14.5	-450	-0.02
Wellgreen crumbs [(1)]	3.13	2.9	14.35	-459	-0.02
Lupin tailings	3.17	7.4	2.82	79	0.10
Control sample[(2)]	2.72				

Notes: [(1)] The slab sample consisted of large chunks of crusted tailings, and crumbs were gravel-sized pieces that broke off the slabs during sampling
[(2)] Quartz silty-sand
Source: MEND 1.62.2

Unfrozen Water Content

Saturated frozen soils comprise solids, ice and unfrozen water. The unfrozen water content present below 0ºC depends upon the temperature, applied pressure, specific surface area, and the pore water chemistry. Detailed information on how frozen water is associated with soil particles is presented in MEND 1.61.2 and Farouki (1986).

The Time Domain Reflectivity (TDR) technique was used in MEND 1.62.2 to measure the unfrozen water content in the tailings samples. TDR measures the travel time of a megahertz pulse through a soil sample from which an apparent dielectric constant is obtained. Topp *et al.* (1980) developed an empirical relationship between the apparent dielectric constant and the volumetric water content that is nearly independent of the soil type, density, temperature and salinity.

The volumetric water content is the ratio of the volume of unfrozen water to the total volume. Smith and Trice (1988) calibrated the relationship between the apparent dielectric constant and the volumetric unfrozen water content that was used in MEND 1.62.2. The data from the tests are plotted in Figure 4.8-21 as the volumetric water content versus temperature. Table 4.8-8 presents interpolated unfrozen water content data for a range of temperatures.

Table 4.8-8
Summary of Unfrozen Water Content - Test Results

Sample	Volumetric Unfrozen Water Content (%)				
	At > OºC	At –1ºC	At -5ºC	At -10ºC	At -15ºC
Wellgreen slab A	48	10	6.2	5.5	5.0
Wellgreen slab B	48	7	5.2	4.8	4.2
Wellgreen crumbs A	48	10	5.0	4.0	4.0
Wellgreen crumbs B	48	9	6.3	4.8	4.2
Lupin tailings	51	7	5.2	4.8	4.2
Control sample	51	0	0	0	0

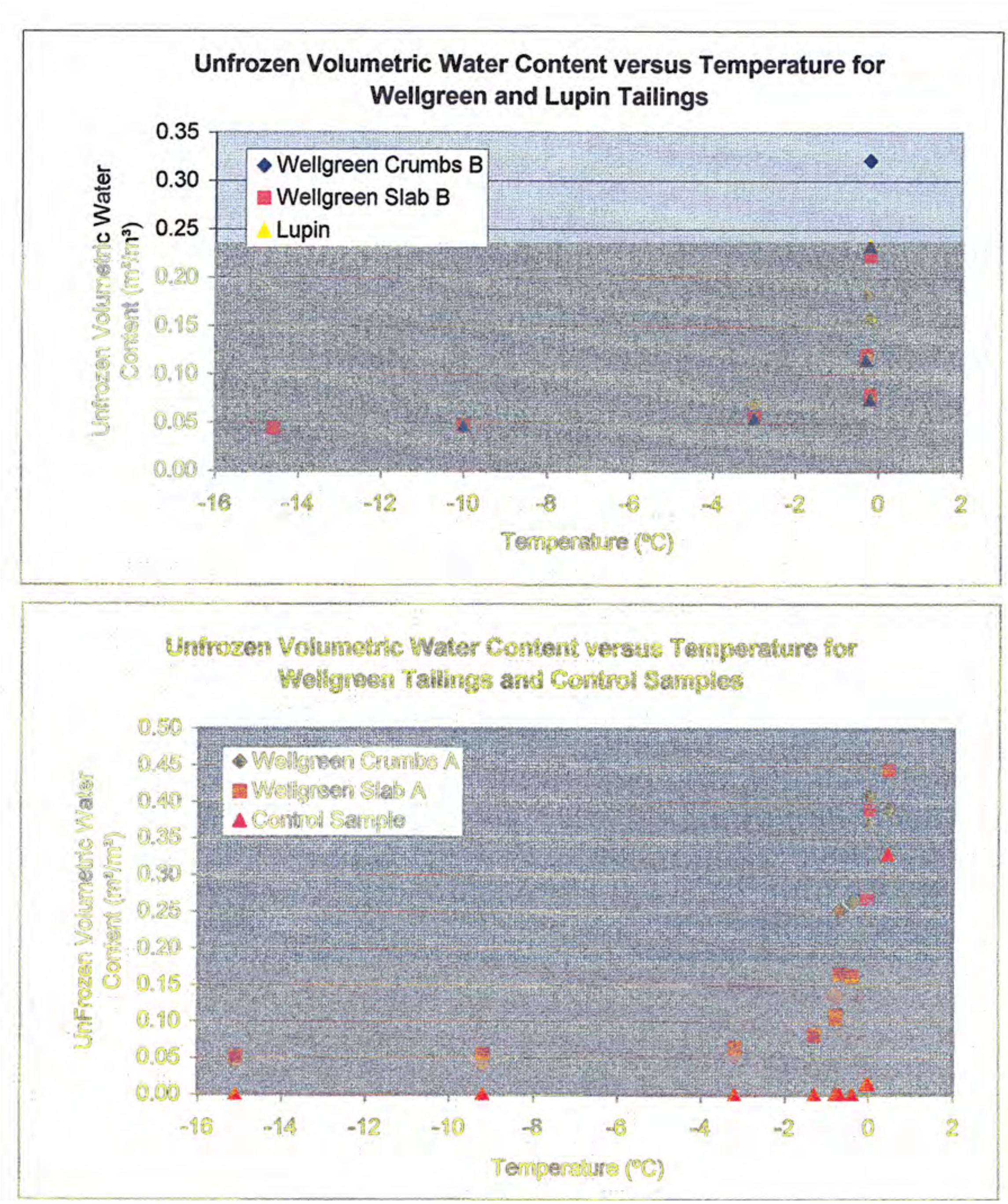
Notes: Wellgreen slab A: consists of Wellgreen slab with Anvil "A" mine water.
Wellgreen slab B: consists of Wellgreen slab with Anvil "B" mine water.
Wellgreen crumbs A: consists of Wellgreen crumbs with Anvil "A" mine water.
Wellgreen crumbs B: consists of Wellgreen crumbs with Anvil "B" mine water.
Lupin: consists of Lupin tailings with Lupin mine water.
Control sample: consists of quartz silty-sand with distilled water
Source: MEND 1.62.2

The data in Table 4.8-8 indicates that:

- At temperatures in the range of -1ºC, the unfrozen water contents are in the range of 7 to 10%. The highest value of 10% corresponded to a sample having the highest pore fluid conductivity; and
- At temperatures from about -5 and -15ºC, the unfrozen water content for the tailings samples is about 5%.

The control sample did not have any unfrozen water below 0ºC. The sand (negligible absorption properties) with distilled pore water (no freezing point depression) helped to confirm the validity of the testing procedure and the empirical relationship between the volumetric unfrozen water content and the dielectric constant. Measurements of unfrozen water content for Wellgreen slab B, Wellgreen crumbs B and Lupin tailings samples at -80ºC show that the unfrozen water contents approaches 2%.

Figure 4.8-21 **Unfrozen Water Content versus Temperature for Tailings Samples**



Source: MEND 1.62.2

<u>Thermal Conductivity Testing</u>

MEND 1.62.2 reports that heat transfer through soils occur mostly by conduction. The thermal conductivity in soils can be measured with steady state and transient methods – the latter was used in the experiment. Additional information on thermal conductivity measuring and estimating methods can be obtained in Farouki (1986). The following information is reiterated from MEND 1.62.2.

Typical thermal conductivity values for soil constituents are shown in Table 4.8-9.

Table 4.8-9
Typical Thermal Conductivity Values for Soil Constituents at 20ºC

Soil Constituent	Thermal Conductivity[(1)] (W/mK)
Water	0.57
Ice	2.2
Quartz	7.7
<u>Other Common Rock Forming Minerals</u>	
Feldspar	1.6 to 2.9
Pyroxene	4.4
Amphibole	3.5

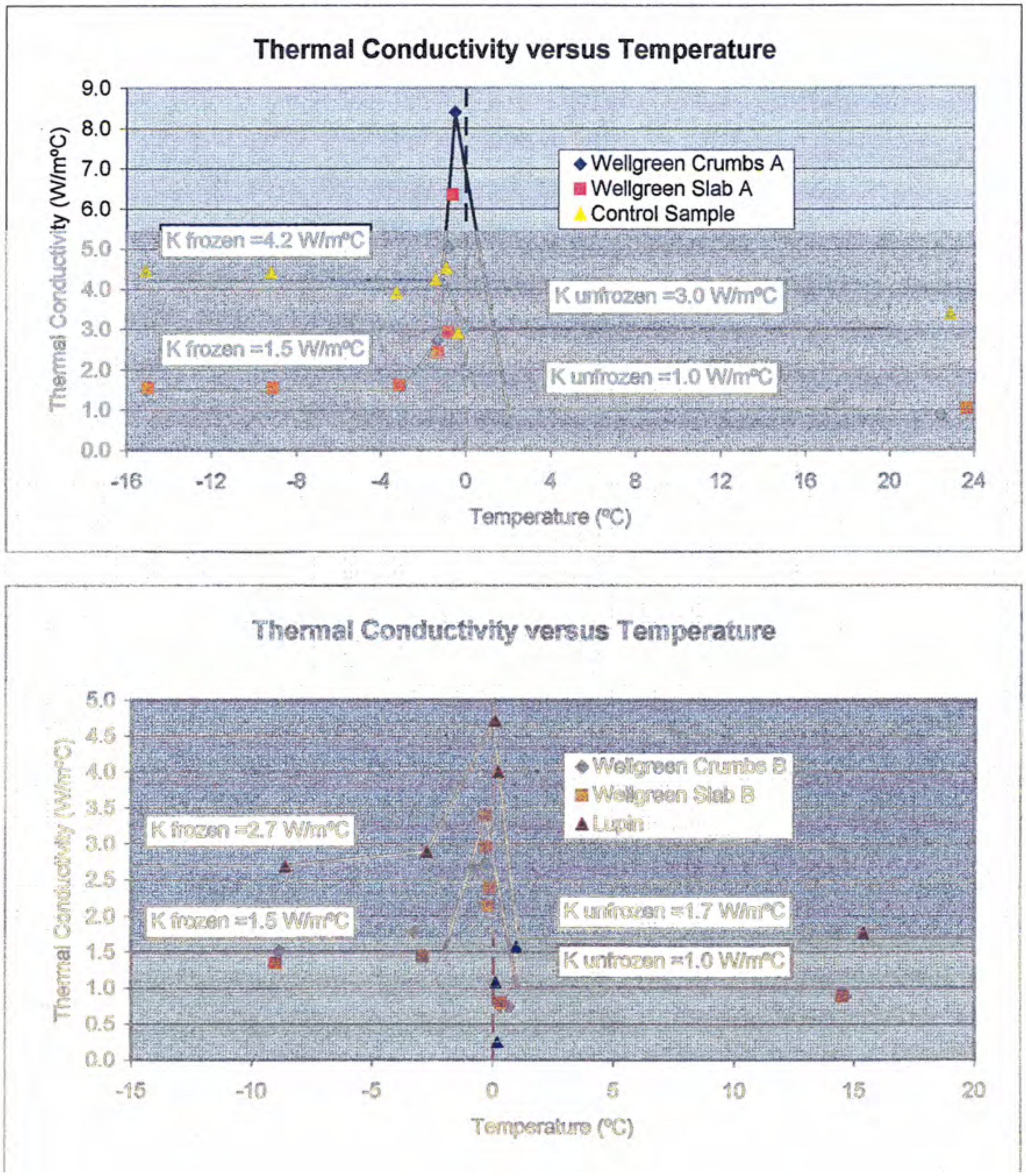
<u>Note</u>: (1) It is common practice to consider thermal conductivity as a single parameter for either frozen or unfrozen conditions (MEND 1.62.2)

Source: MEND 1.62.2

The thermal conductivity of ice is about four times that of water, and as a result, soils are better thermal conductors when frozen. The results of the TDR testing completed under MEND 1.62.2 are shown in Figure 4.8-22. The thermal conductivities of each of the tailings samples peaked between their frozen and unfrozen values. It was postulated that the peaks may be due to the onset of solute crystallization at the freezing temperature of the solutions. Estimated thermal conductivity values for the tested materials are summarized in Table 4.8-10.

MEND 1.62.2 reports that the thermal conductivity of saturated natural soil can be estimated empirically by relating the thermal conductivity of the soil constituents to the porosity and the degree of ice/water saturation. Farouki (1986) reports that estimates can be obtained within 25% of measured values.

Figure 4.8-22 **Thermal Conductivity versus Temperature for Tailings Samples**



Source: MEND 1.62.2

Table 4.8-10
Estimated Frozen and Unfrozen Thermal Conductivity Values

Sample	Porosity (%)	Thermal Conductivity (W/mk)	
		Unfrozen	Frozen
Wellgreen	48	1.0	1.5
Lupin tailings	51	1.7	2.7
Control sample	51	3.0	4.5

Source: MEND 1.62.2

<u>Frozen Hydraulic Conductivity</u>

The advective transport of water within a soil at temperatures below 0ºC is related to the amount and the distribution of unfrozen water. Horiguchi and Miller (1983) tested silts, clays and zeolites and determined that the hydraulic conductivity decreased sharply by three to four orders of magnitude (below 10^{-8} cm/s) as unfrozen water content was reduced to 50 to 70% of the total water content when the temperature reached the -0.2 to -0.4ºC range.

In MEND 1.62.2, estimates of the frozen hydraulic conductivity were made using measured unfrozen hydraulic conductivity values and a semi-empirical approach for estimating frozen hydraulic conductivity. The unfrozen conductivity was calculated from the rate of the volume change of soil in a standard (ASTM 0-2345) uniaxial test method to measure soil compressibility. The unfrozen hydraulic conductivity values for the Wellgreen slab and crumbs tailings ranged from 10^{-3} to 10^{-4} cm/s, while hydraulic conductivity for the Lupin tailings sample (which has a higher silt content than the Wellgreen tailings samples) was in the range of 10^{-4} to 10^{-5} cm/s.

A general finding of MEND 1.62.2 is that the freeze-back of most types of mine tailings in a continuous permafrost environment is a viable long-term strategy. Freeze-back should occur at similar rates, or slightly lower for weathered materials, than would be expected for naturally occurring materials. Thermal protection of the active layer would ensure that seasonal thawing does not lead to acid generation.

4.8.3 APPLICATIONS AND LIMITATIONS

Technologies involving the use of permafrost conditions to prevent acidic drainage from tailings are under development. The development of a mine closure strategy based on the presently available concepts would necessitate site-specific research including but not limited to temperature monitoring, permafrost and materials characterization, and a geochemical and thermokinetic assessment of the long-term performance of the overall permafrost/frozen tailings system.

The oxygen diffusion flux at the tailings interface was predicted for the following cases based on a scenario involving a 100 ha tailings area containing 10 Mt of 20% pyrite tailings, and *in situ* temperatures of 0, 4 and 25ºC (MEND 1.61.1). The predictions are summarized in Table 4.8-11.

Table 4.8-11
Steady State Oxygen Diffusion Flux at the Tailings Interface

Scenario	**Oxygen Flux ($g/m^2/yr$)**	**Estimated Time to Oxidize Pyrite (years)**
Case 1: "Control". The waste is not covered, and is exposed and unsaturated	11,000 at 25º C 6,000 at 4º C 4,000 at 0º C	180 at 25º C 360 at 4º C 470 at 0º C
Case 2: The waste is water saturated, with a zero depth of water over the waste	3.0 at 25º C 2.5 at 4º C 2.0 at 0º C	570,000 at 25º C 760,000 at 4º C 870,000 at 0º C
Case 3: The waste is covered by a 1 m deep and stagnant water cover	0.5 at 25ºC 0.6 at 4º C 0.7 at 0º C	4,200,000 at 25º C 3,200,000 at 4º C 3,000,000 at 0º C
Case 4: The waste is covered by a 2 m deep and stagnant water cover	0.25 at 25º C 0.35 at 4º C 0.38 at 0º C	8,000,000 at 25º C 5,700,000 at 4º C 5,200,000 at 0º C
Case 5: The waste is covered by a 1 m deep water cover that is well mixed	3.5 at 25º C 2.6 at 4º C 2.3 at 0º C	570,000 at 25º C 760,000 at 4º C 870,000 at 0º C

Note: Figures have been rounded.
Source: MEND 1.61.1

MEND 1.6.1.1 (also Davé and Blanchette 1999) report that the effects of water cover depth in attenuating oxygen diffusive flux are realized in unmixed or stagnant water columns where oxygen influx decreases with increased water column depth (Table 4.8-11). However, only a minimal incremental gain in oxygen flux attenuation is achieved by doubling the water cover depth from 1 to 2 metres.

Raglan Mine

The Raglan Mine located at the northern tip of the New Québec Peninsula, about 2000 km north of Montréal, Québec operates in an environment where the underground rock temperature is -5.5ºC and the expected limit of the continuous permafrost is 425 m below surface (Pronovost 1999). Delarosbil (1999) reports that the Raglan site contains in excess of 20 million tonnes of proven and probable reserves grading 3.17% Ni, 0.88% Cu with precious and rare metals. Experience at the Raglan mine site has been that permafrost conditions can be established in tailings and used to the advantage in reclamation (Swarbrick 2000). For waste rock, similar advantages can be obtained through planned waste rock placement with a focus towards the possible use of an engineered cover at closure. The focus of current evaluations at the mine site are to develop dry cover placement techniques that can provide suitable thermal/hydrogeological conditions over the long term.

Research Needs

Limitations of the present state of knowledge of acid generation control in permafrost regions relate to the need for additional research. Research needs have been identified in a number of areas, as follows:

- The effects of sub-zero temperatures on sulphide oxidation;
- The hydrogeology of waste rock dumps, and the heat transfer within rock dumps;
- Field demonstrations of acid generation control strategies;
- Assessment of biotic oxidation under cold climate conditions including those where bacteria adapt to low temperatures;
- The effect of electrochemical sulphide oxidation; and
- The development of affordable insulating-cover designs.

4.8.4 VARIATIONS

Continuous and discontinuous permafrost conditions present different challenges for acid generation control. The application of permafrost technologies to inhibit acidic drainage would be expected to be a greater challenge in a discontinuous permafrost region in comparison to a continuous permafrost region.

4.8.5 COST

Acidic drainage control methods that take advantage of permafrost and cold temperatures are under development. The cost for the application of the various methods will be subject to site-specific conditions.

4.8.6 MEND AND RELEVANT PUBLICATIONS

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MEND 1.61.2 1996. Acid Mine Drainage in Permafrost Regions: Issues, Control Strategies and Research Requirements. July.

MEND 1.61.3 1997. Column Leaching Characteristics of Cullaton Lake B and Shear (S) - Zones Tailings Phase 2: Cold Temperature Leaching. June.

MEND 1.62.2 1998. Acid Mine Drainage Behaviour in Low Temperature Regimes - Thermal Properties of Tailings. July.

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4.9 BACKFILLING

The backfilling of mine openings, both open pits and underground workings, has been practised extensively internationally. It is only recently, however, that pit backfilling programs have been designed for the disposal of sulphide tailings and waste rock with mine closure in mind. As a result, some mine openings that may have previously been considered a legacy may be beneficially used for mine waste disposal.

MEND completed a focused investigation of open pit backfilling practices at both the national and international levels and determined that some pits provided suitable physical and geochemical environments for mine waste disposal (MEND 2.36.1). MEND also determined that in-pit disposal has been widely applied, and that a key deficiency is a general lack of scientific monitoring data for backfilled pits. This section focuses on the in-pit disposal of sulphide wastes as a means of preventing and controlling acid generation.

MEND has also assessed the use of technologies such as paste backfill, which can offer superior performance in terms of underground support, reduced fill cycle time, reduced overall cost, and practical operating considerations in comparison to conventional hydraulic backfill. MEND 2.17.1 has investigated the use of these technologies in surface applications to create elevated water table conditions in tailings deposits (Section 4.3 Saturation). As indicated in the following subsections, tailings backfill technologies could conceivably be applied to an in-pit disposal program under suitable conditions.

4.9.1 DISCUSSION OF THEORY

An open pit mine that has ceased production may be suitable for the planned disposal of waste rock and tailings that are, or have the potential to become, acid generating. The wastes may be excavated and promptly disposed into a mined-out pit, or relocated to the pit from waste rock dumps.

The key objective in placing sulphide wastes in a mined-out pit is to provide a suitable physical and geochemical environment to prevent acid generation and thereby reduce adverse impacts to receiving groundwater and surface water resources. The level of engineered controls that are required to prevent acid generation can be determined to a large extent by predicting the future quality of the pore water within the wastes. The degree of oxidation of the wastes to be disposed in a pit will determine the suitability of an in-pit disposal program, and significantly influence the design of the in-pit disposal program.

The quality of the pore water in the disposed wastes may vary over time. In some cases, the initial rates of contaminant release will be high, and then reduce as the leachable fraction of the waste declines, possibility to levels where the potential for environmental impacts is negligible. Conventional testing (i.e. pore water sampling, sequential leaching tests, humidity cells) can be used to predict pore water characteristics, leachable fractions, and ultimate concentrations of contaminants. MEND 2.36.3 examined available laboratory methods and their applicability to assessing metal release in a subaqueous environment.

Tailings placed in a pit can have a lower permeability than the fractured pit walls. Under this type of condition, groundwater movement may tend to be through the fractured rock zones, and hence around a pit, rather than through the waste. This concept which would be expected to reduce contaminant release, has been applied based on the “pervious surround” or “porous envelope” approach, whereby a permeable zone is constructed around the waste placed in a pit. Other phenomena that may occur within backfilled tailings are:

- Permeability may be reduced due to secondary mineral precipitation;
- Permeability may be reduced due to the initial state of compaction and the potential for wastes to consolidate over time. Planning needs to address the potential for differential settlement and effects on barriers placed over, under, or around the reactive wastes placed in the pit;
- Permeability may increase due to dissolution of secondary minerals;
- Permeability may decrease as a result of waste weathering and/or alteration; and
- Freezing may result in the formation of ice lenses in tailings.

4.9.1.1 Acid Generation Control Strategies

The following acid generation control strategies are available for use in an in-pit disposal program.

1. Underwater disposal. Mined-out pits that become flooded provide potential opportunities for the disposal of mine wastes in the pit lakes. Water covers have been demonstrated to be effective in controlling acid generation as described in Section 4.2.

2. Buffering. Alkali materials can be blended with the mine wastes. Waste rock is commonly blended with limestone. A drawback with limestone as the sole neutralization reagent is its inefficiency in precipitating metals that are not effectively precipitated at pH levels below 6.0. Hydrated lime can be used to raise the pH to higher levels. The blending and layering of waste rock is reviewed in Section 4.6.

The site-specific requirements for acid buffering depend on the objective. As an example, buffering may be used to neutralize existing acidity and/or acidity that will be generated in the future. In the first case, it may be possible to place acidified waste into a pit and then flood or cap the waste. The second case could apply to waste rock that will be disposed above the water table in a pit. Under this condition, engineered controls (e.g. an engineered cover) would be required to prevent or control acid generation, and an alkali material(s) could be blended with the acid waste.

3. Moisture Infiltration. Engineered covers can be constructed over the waste placed in a pit to inhibit moisture infiltration and the transport of oxidation products. Engineered covers are reviewed in Section 4.4. A dry cover can also be designed to serve as an oxygen barrier. Oxygen barriers include saturated barriers that act to control diffusive oxygen flux and barriers that consume infiltrating oxygen.

4. Sulphate Reduction. Sulphate reducing conditions (e.g. a reactive wall) can be constructed to intercept contaminated groundwater flow and passively remove dissolved metals from solutions. The biological reduction of sulphate to hydrogen sulphide or bisulphide occurs under anaerobic conditions and requires a source of decomposable organic carbon. The hydrogen sulphide can in turn react with dissolved metals to precipitate metal sulphides. Sulphate reduction is reviewed in Volume 5.

4.9.2 DISCUSSION OF MEND RESEARCH

MEND 2.36.1 investigated the use of in-pit disposal as a means of permanently disposing of acid generating, or potentially acid generating mine wastes. The project reviewed in-pit disposal practices, and presented twelve applications as case studies. The key findings were:

- In-pit disposal is common practice;
- The use of in-pit disposal is not universally applicable, as site-specific conditions determine which pits are suitable for backfilling with acid generating, or potentially acid generating mine waste; and
- In-pit disposal practices are well developed. A deficiency exists in the area of monitoring, as there is a general lack of scientific monitoring data for pits used for mine waste disposal.

MEND 2.36.1 also identified four options for the placement of wastes in pits.

Option 1: Underwater Disposal

Option 2: Elevated Water Tables

Option 3: Dry Disposal

Option 4: Perched Water Tables

4.9.2.1 Option 1 – Underwater Disposal

In this option, a pit lake or wetland would exist upon the completion of backfilling. The four suboptions (Figure 4.9-1) for the placement of the wastes in the pit are:

Option 1a – Simple Underwater Disposal

Option 1b – Underwater Disposal with a Surface Barrier

Option 1c – Underwater Disposal with Groundwater Barriers

Option 1d – Underwater Disposal with Surface and Groundwater Barriers.

Option 1a - Simple Underwater Disposal

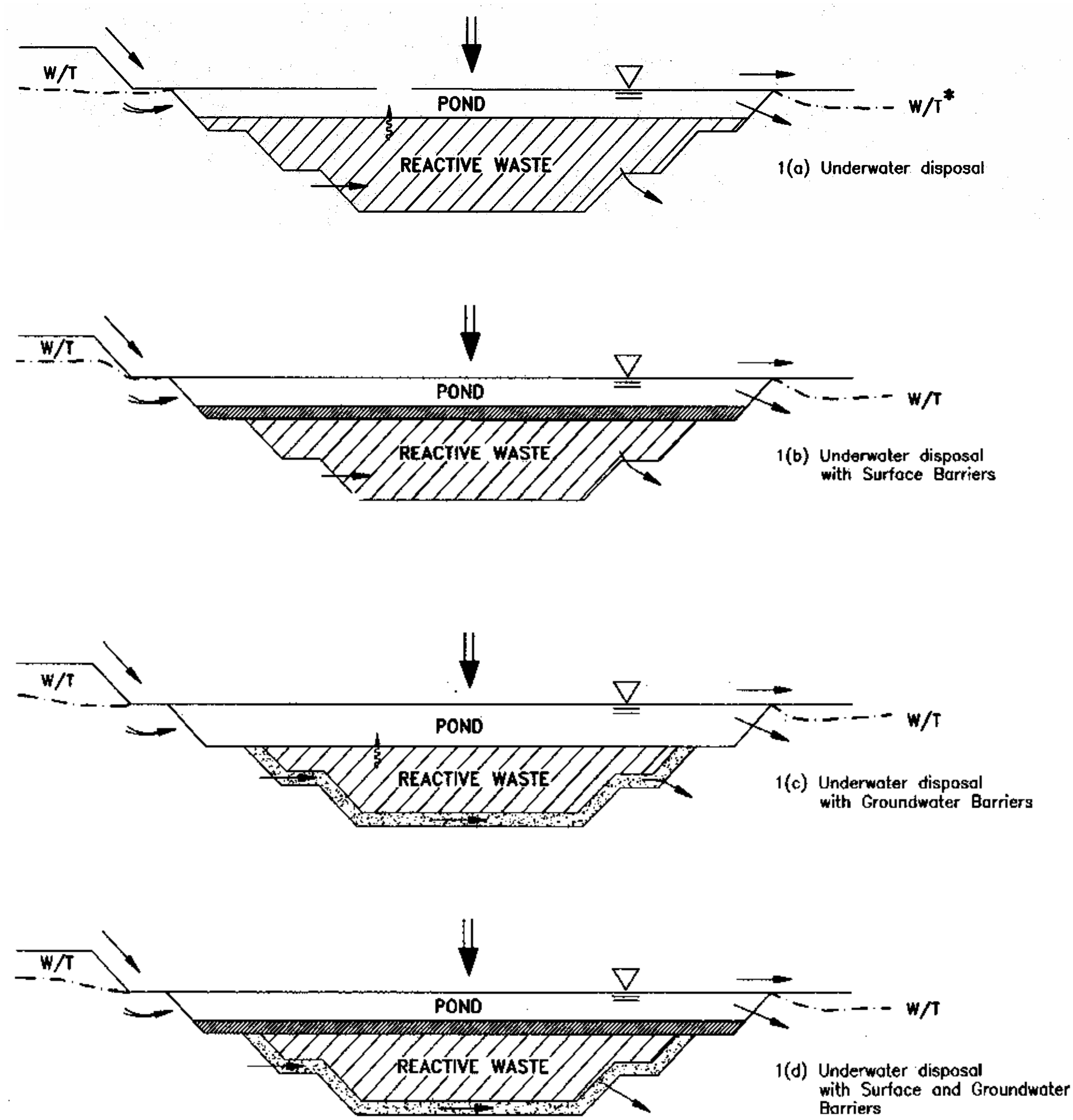
In this option, the waste is placed at the bottom of the pit, which is then flooded. In an ideal pit, convective groundwater transport would be minimal and the prime mechanism for release would be through mass transfer (diffusion) from the surface of the waste into the pit lake water.

The quality of the water in a pit lake also needs to be taken into consideration. Shevenell (2000) reports that open pits in Nevada (Figure 4.9-2) that have been developed to below the water table will slowly fill once the mining and pumping operations cease. Pit dewatering and filling concerns include regional effects on the water table during and after pumping, the rate of pit lake filling, the ultimate pit lake water quality, limnology and impacts on wildlife.

For deep flooded pits, additional factors include development of meromixis (chemical stratification); thermal stratification; or anaerobic conditions. Stratification refers to a lack of mixing between layers. Some of these layers are stable (e.g. meromixis), while others may break down (e.g. seasonal turnover events). The development of anaerobic or anoxic conditions may occur in deep pits as minimal to no oxygen can penetrate to the bottom, especially in pits where meromixis persists. The chemical effects of anoxic conditions are not likely to be significant unless decomposable organics are present to support biological sulphate reduction and metal sulphide precipitation. From an environmental perspective, meromixis and the anoxic conditions could make bottom sediments in deep pits generally unsuitable for most aquatic species (i.e. fish, benthos).

A variation of underwater disposal was used at BHP's Island Copper open pit mine located near the northern end of Vancouver Island in British Columbia. The mine closed in 1995 after nearly 25 years of operation and production of metal concentrates containing over 1.3 Mt of copper, 31,000 t of molybdenum, 31.7 t of gold, 336 t of silver and 27 t of rhenium. At closure, options for the use of the open pit included: its use for municipal waste disposal; connecting the pit to the sea to make a new inlet; and the flooding of the pit with seawater to create a stable meromictic lake for the passive treatment of acidic drainage from waste rock dumps on the mine site. In the

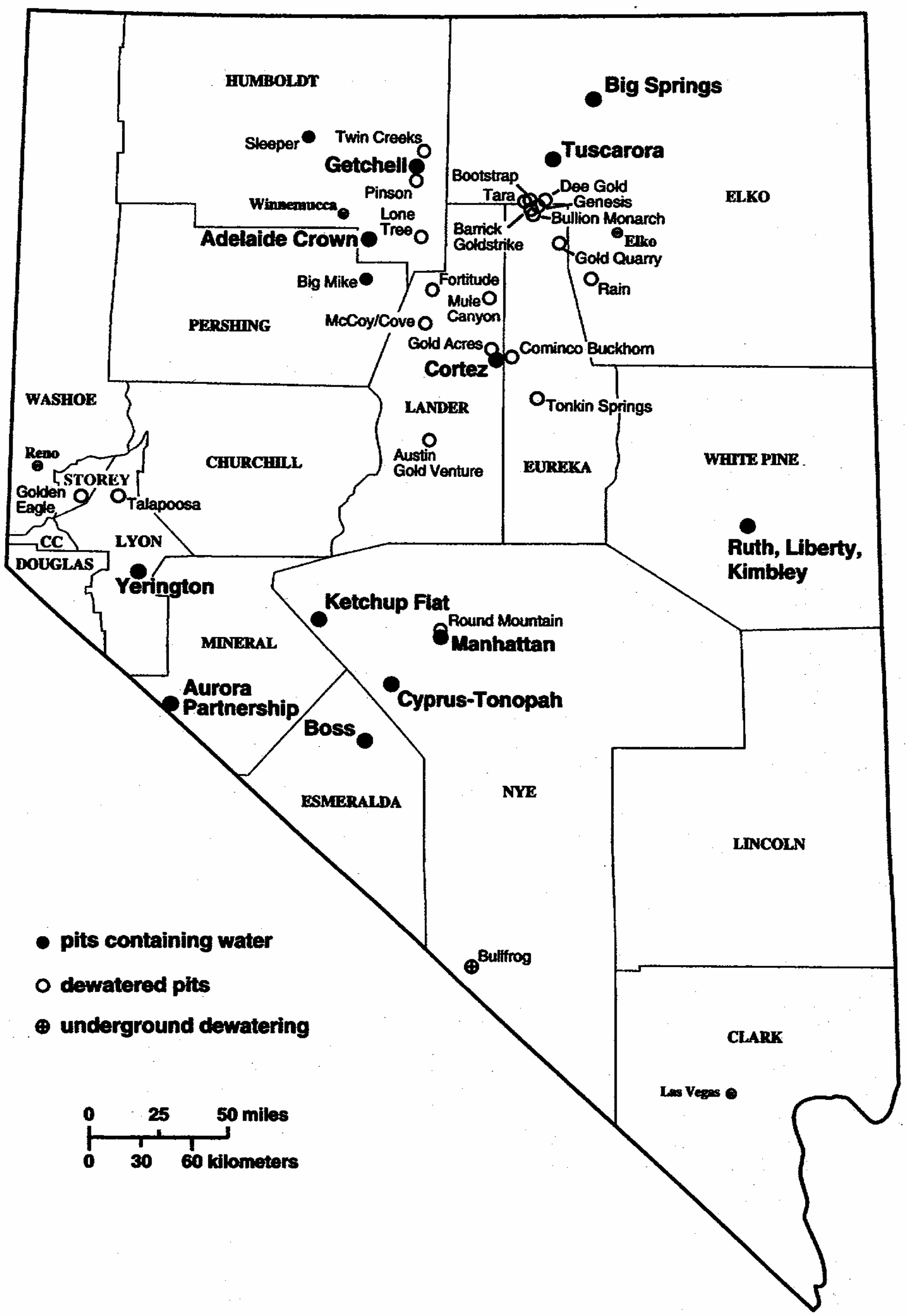
Figure 4.9-1 **Pit Disposal Concepts – Underwater Disposal**



* W/T = Water Table

Source: MEND 2.36.1

Figure 4.9-2 **Map of Nevada Showing Locations of Current and Future Open Pit Mines that will Contain Water Upon Cessation of Mining Activities**



<u>Note:</u> ? Pits that currently contain water or have contained water
? Pits that will contain water in the future

Source: Shevenell 2000

third option, the mainly sea water lake was expected to become anoxic below about a 30 m depth due to bacterial action and the near absence of mixing of oxygenated water at the surface of the pit lake. Sulphate reducing conditions in the pit would allow dissolved metal ions to be precipitated as insoluble metal sulphide particles, which would then settle out as sediments. Acidic drainage, collected in two drainage ditch systems around the waste rock piles, would be discharged via two polyethylene pipelines at depths of 200 m at both the north and south sides of the pit.

The pit flooding option was selected and the Island Copper pit was flooded with seawater in 1996. Poling (1998) reports that pit lake monitoring shows that the pit lake developed a three-layer meromictic system. The top layer of fresh and brackish water extends down 12 m, and the quality of the surface-most layer of water that eventually flows into Rupert Inlet meets discharge criteria. The centre saline layer, located at a depth of 12 to 200 m, appears to be relatively well mixed by the upwelling plumes of the low density acidic drainage that is piped into the pit and released some 200 m below the pit lake surface (Figure 4.9-3). The bottom saline layer, below 200 m, appears to be quiescent and progressing towards anoxia faster than the centre layer.

Empirical and mechanistic models can be used to predict short-term and long-term contaminant concentrations in a pit lake. A sampling of technical papers that describe pit lake water quality monitoring is provided.

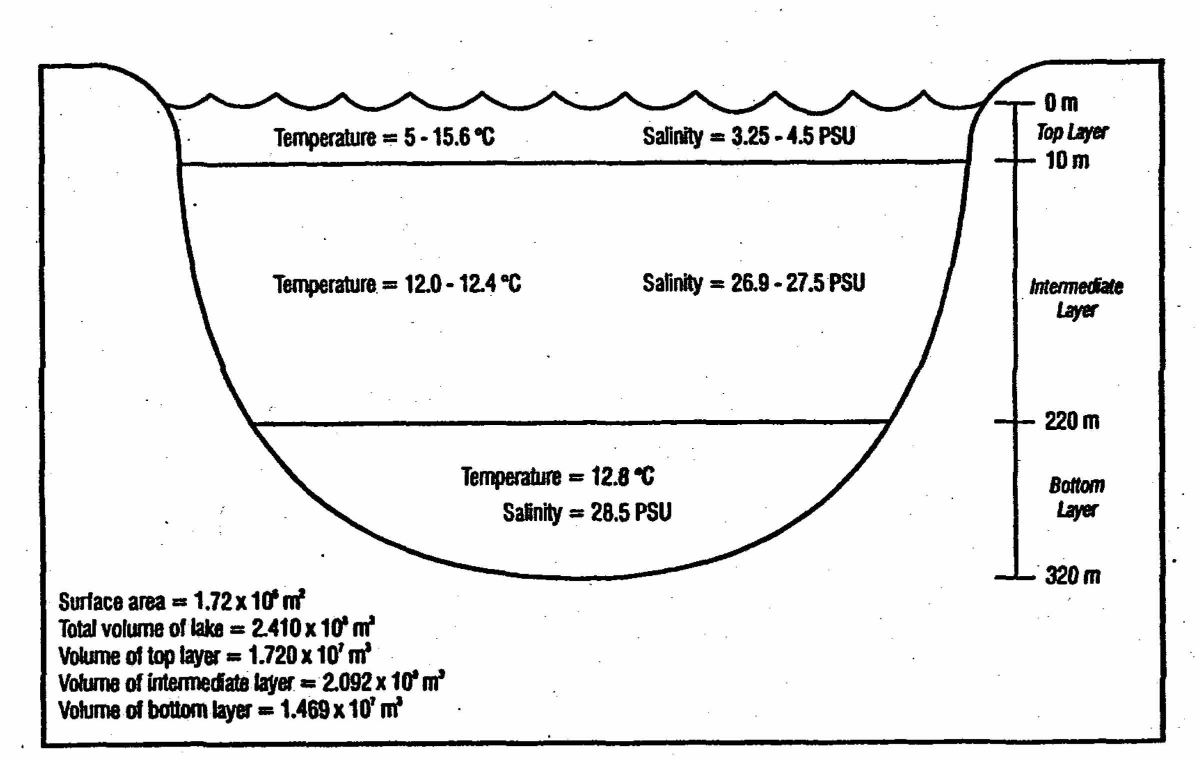
- Morin (1990, 1994) developed a computer program (MINEWALL) which considers the relevant geochemical aspects of unit-rock-surface reaction rates (measured experimentally) and the total amount of reactive rock surface. The key factor in predicting mine water chemistry is the estimation of percentages of reactive surface that are regularly flushed (prior to flooding).

- Kempton *et al*. (1994) describe the combined use of several models to predict source contributions and resulting water quality in future pit lakes at the Robinson Project in White Pine County, Nevada. Monte Carlo simulations were used to address uncertainty related to chemical release from pit walls, net neutralization potential in wall rock, the sulphide content of wall rock, and the wall rock porosity.

- Bird *et al.* (1994) examined the suitability of five hydrogeochemical computer modelling software packages (BALANCE, MINTEQA2, PHREEQE, WATEQF, and WATEQ4F) to predict the current pit water geochemistry at the Cortez open pit. They concluded that each computer code could be used for a subset of the overall pit water modelling process (e.g. models should be used in combination).

- Stevens *et al.* (1994) described the modelling of the thermal stratification of water-filled pits.

Figure 4.9-3 **Schematic Diagram of the Pit Lake Water Column Structure, Island Copper**



Source: Muggli *et al.* 2000

- Muggli *et al.* (2000) completed numerical modelling of the Island Copper pit lake based on piped flows of runoff from waste rock dumps to an point 200 m below the surface of the pit lake.

Simple underwater disposal is the most commonly applied underwater disposal option. If this option is inadequate, additional engineered controls can be considered (see Options 1b, c and d).

Option 1b – <u>Underwater Disposal with a Surface Barrier</u>

In this option, a barrier is placed over the surface of the submerged wastes to reduce upward contaminant transport into the pit lake. The release of contaminants can be diffusion controlled using a fine-grained barrier (e.g. sand, till, clay). Other potential diffusion barriers that may be considered include a layer of organic material that can act as a diffusion barrier and provide a geochemical environment suitable for biological sulphate reduction and metal sulphide precipitation. Numerical models can be used to determine the required thickness of the barrier.

Option 1c – <u>Underwater Disposal with Groundwater Barriers</u>

Three types of potential groundwater barriers are available for consideration in pits where there is a substantial convective flow of groundwater through the waste material:

Option 1c (i) barriers which block groundwater flow such as liners, clays;

Option 1c (ii) barriers which provide a low resistive flow path to groundwater; and

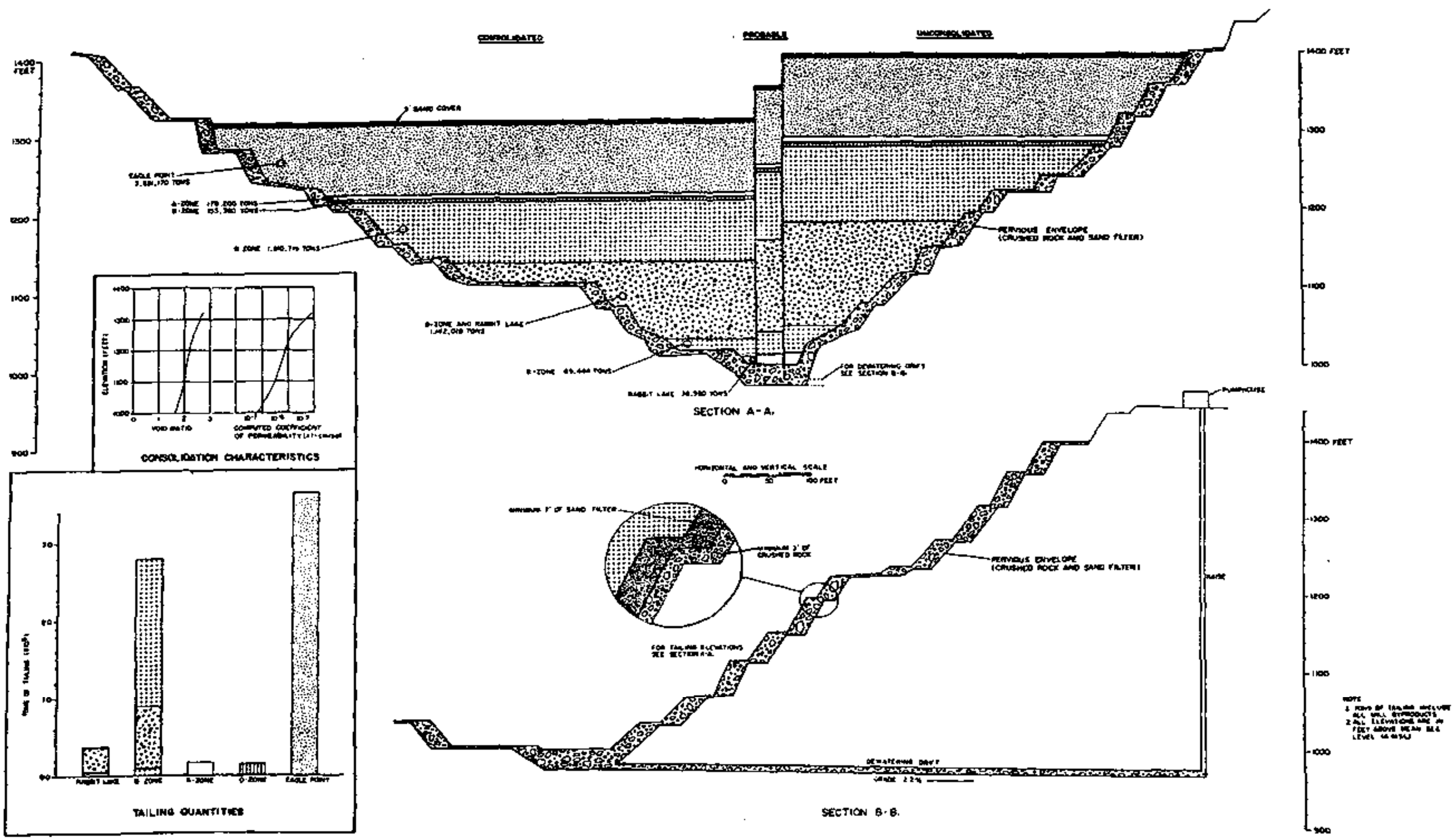
Option 1c (iii) barriers which may remediate contaminated groundwater flow.

These barriers tend to be costly, and few applications exist. Options 1c (i) and (ii) require use of hydrogeological models to determine the effectiveness of the barriers. A low permeability barrier, Option 1c (i), would be expected to serve dual functions: to reduce flow through the waste; and to provide a barrier to contaminant diffusion. There is no reported experience with use of low permeability barriers in pits, but the potential benefits have been identified through modelling.

The creation of a preferential flow path for groundwater, Option 1c (ii), was applied at the Rabbit Lake open pit uranium mine situated in northern Saskatchewan. The Rabbit Lake pit was developed in 1975 and operated until 1984. In 1982, approval was received for an engineered pit disposal system, which includes (Figure 4.9-4):

- The placement of a bottom rock drain to collect pit drainage/seepage;
- The application of an engineered porous envelope which served as a drainage conduit during operation and as a pervious envelope at closure;

Figure 4.9-4 **Schematic of Tailings Deposition for the Different Collins Bay Ore Deposits**



Source: MEND 2.36.1

- Placement of tailings in the pit as a filtered cake;
- Placement of a soil/sand surface layer to ensure contaminant release to the pit lake will be controlled by diffusion; and
- Closure with a wet cover (e.g. a pit lake).

Under this concept, a pervious envelope is constructed along the pit perimeter and around the waste to create a preferential path for groundwater flow around tailings. If the permeability contrast between the tailings and the "pervious surround" material is large, groundwater will flow around the tailings mass rather than through it, and metal leaching will be minimal. This type of barrier would not be as effective for coarse rock.

The third potential barrier, Option 1c (iii), would involve a redox barrier or an acid-consuming barrier. Redox (organic) barriers have received considerable attention. They could be constructed within or downstream of impoundments and are being investigated as a method of containing acidity and soluble metals. The success of this technology could reduce the cost of leachate management, but the technology is still in the early stages of its development.

4. Option 1d - Underwater Disposal with Surface and Groundwater Barriers

This option is a combination of Options 1b and 1c. In this application, a complete envelope would be constructed around the waste as is planned at the Rabbit Lake pit. This closure option is also proposed at other sites in northern Saskatchewan.

4.9.2.2 Option 2 – Pit Backfilling – Elevated Water Table

In this option, the pit would be backfilled to near the original ground surface. The primary objective is to control oxidation by raising the water table above the reactive waste and into a layer of clean fill/waste. The major difference between this concept and simple underwater disposal is that the water table is likely to be sloped, therefore, infiltration would likely pass through the waste as would groundwater, unless engineered controls are put in place. The following five options for elevated water tables are shown in Figure 4.9-5.

Option 2a – Saturated Reactive Waste

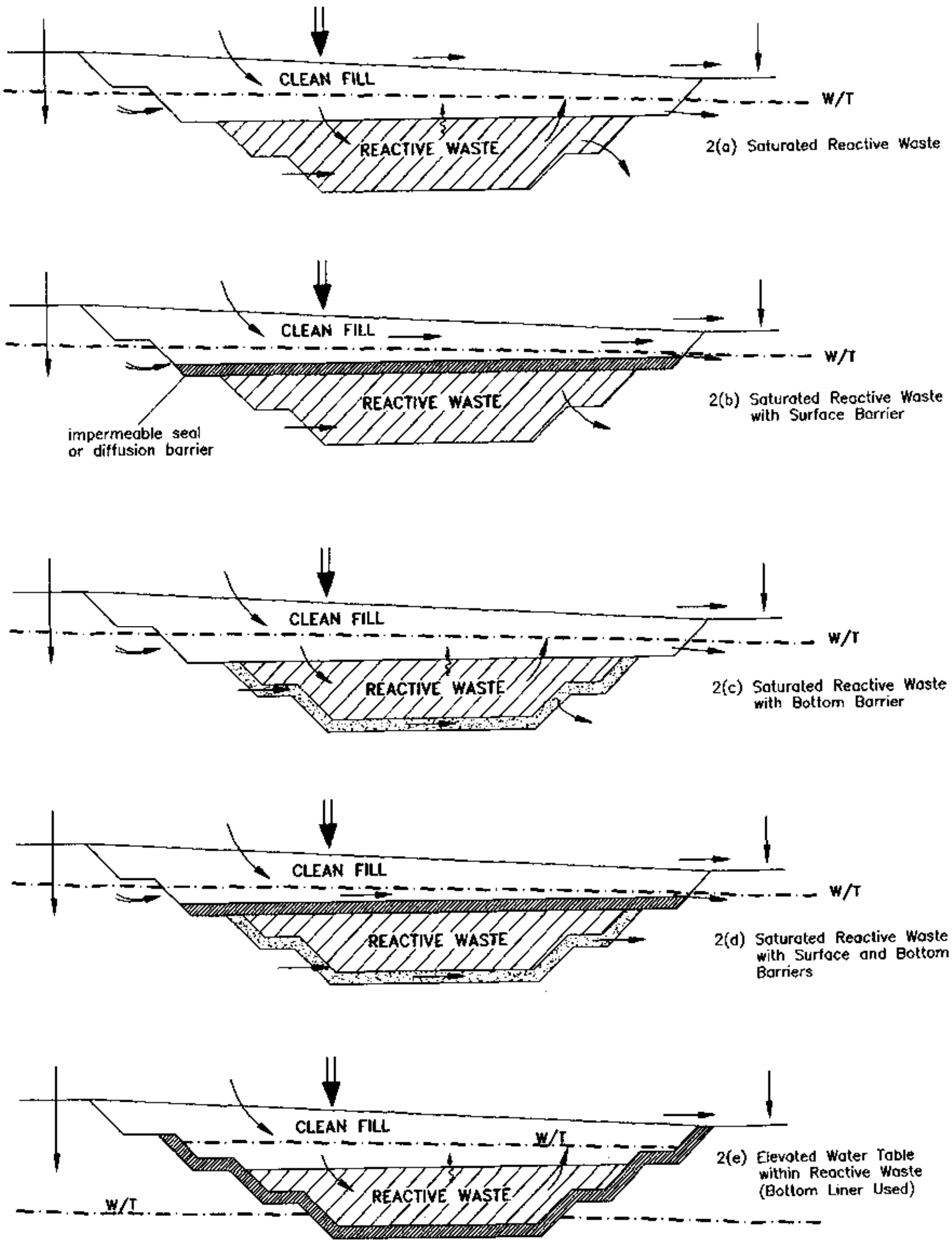
Option 2b –Saturated Reactive Waste with Surface Barrier

Option 2c – Saturated Reactive Waste with Bottom Barrier

Option 2d – Saturated Reactive Waste with Surface and Bottom Barriers

Option 2e – Elevated Water Table within the Wastes using a Bottom Liner

Figure 4.9-5 **Pit Disposal Concepts – Elevated Water Table**



Source: MEND 2.36.1

Options 2a, b, c and d are analogous to the underwater deposition options. In Option 2e, the bottom barrier could be an impermeable liner to trap infiltration water in a region where the natural groundwater table is depressed. The liner would create a basin in which the water level would rise and submerge the reactive waste.

4.9.2.3 Option 3 – Dry Disposal

As shown on Figure 4.9-6, it may not be possible to create a pit lake or an elevated water table condition in some pits (e.g. sidehill). In these situations, the options that can be applied for dry disposal are:

Option 3a – Engineered Cap/Cover

Option 3b – Acid Buffering Barriers

Option 3c – Alkali Blending

Option 3d – Engineered Cover with Water Table Drawdown

Option 3a – Engineered Cap/Covers

As noted in Section 4.4, dry covers were extensively studied under MEND. A dry cover can serve as a barrier to water, a barrier to oxygen, and/or barrier to both water and oxygen.

Option 3b – Acid Buffering Barriers

In this option, the waste would be placed over an alkaline barrier or an organic barrier. The role of these types of barriers were described previously (see Options 1b and 1c).

Option 3c – Alkali Blending

The two basic approaches to alkali addition are:

- The addition of sufficient alkali to consume acid produced by the reactive waste; and
- The blending of alkali and waste materials to produce a net acid-consuming material.

Alkali blending was applied at the Owl Creek open pit in Timmins, Ontario. The Owl Creek mine operated between 1981 to 1989 producing 1,700,000 t of ore and 7,820,000 t of waste. The waste included 3,050,000 t of overburden, 4,235,000 t of volcanic waste rock, and 535,000 t of graphitic argillite waste rock – the latter was later found to be the main acid source. In June 1990, the waste rock pile drainage was found to be acidic. The mine operator responded promptly and constructed a drainage collection and treatment pond. MEND 2.36.1 and Orava *et al.* (1997) report that following intensive field and technical investigations, five options were developed to control acid generation from the waste rock (Table 4.9-1).

Table 4.9-1

Waste Rock Decommissioning Options – Owl Creek Open Pit

Waste Rock Decommissioning Option	**Net Present Value of Decommissioning Cost (in 1992 Dollars)**
Perpetual collection and treatment	$7,400,000
In situ flooding of the waste rock dumps	$10,000,000
Relocate the waste rock to a tailings area	$9,200,000
Engineered cover over the waste rock dumps	$7,600,000
In-pit disposal	$6,200,000

In-pit disposal was selected as the preferred option. Acid generating waste rock was relocated to the mined-out open pit. Limestone was placed with the waste rock at a rate of 9 kg/t to neutralize acidity. The backfilled pit flooded rapidly. The water quality of the pit lake was subsequently monitored and shown to continuously improve. Key factors that led to the success of this program included the neutralization of pore water, and the relatively prompt submergence of the waste rock.

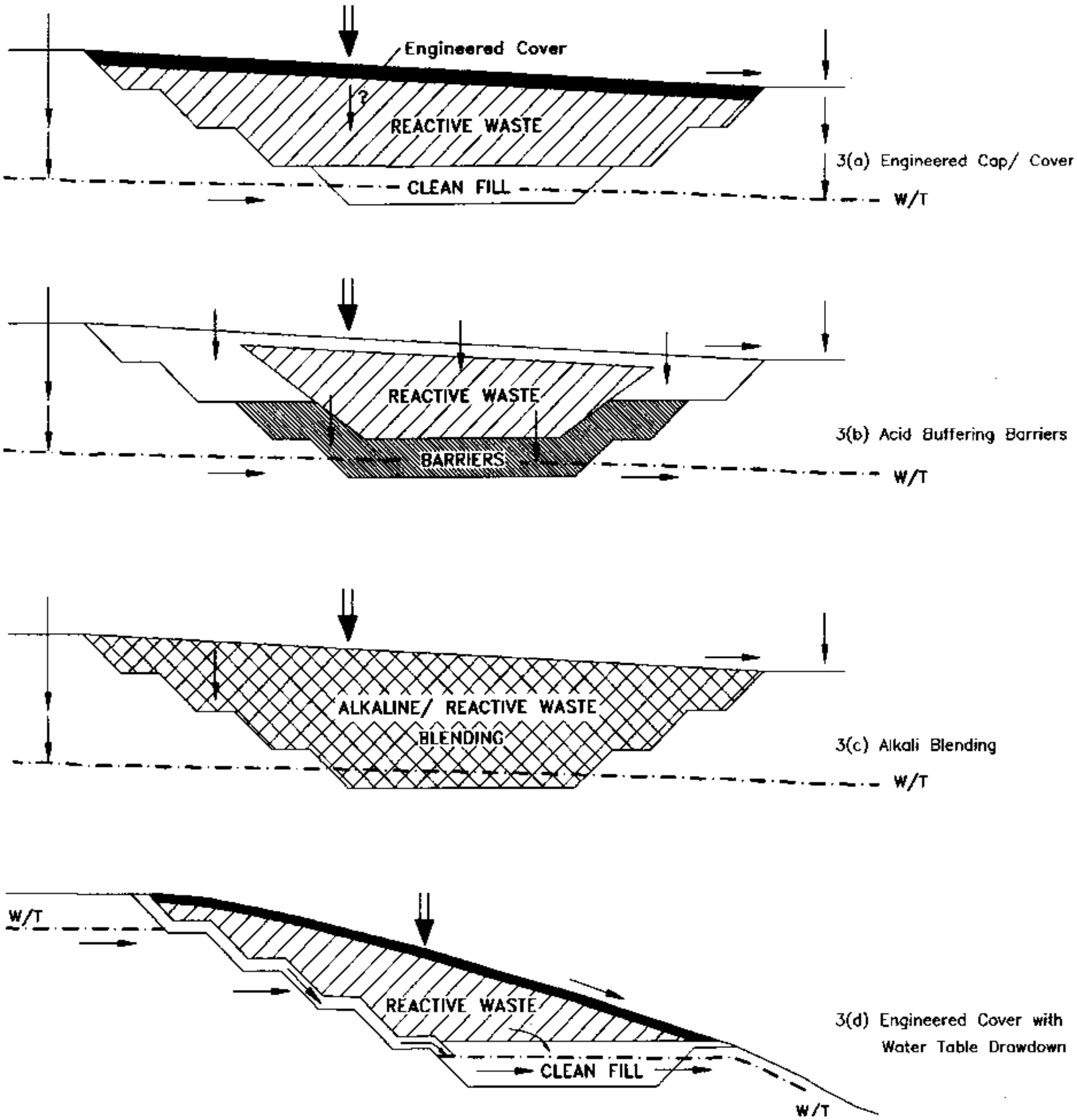
Option 3d - <u>Engineered Cover with Water Table Drawdown</u>

Open pits located on hillsides can be used for reactive waste disposal with engineered controls. The example shown in Figure 4.9-6 (Option 3d) is a sidehill pit where the water table emerges in the head wall. This general configuration for waste disposal was adapted for use at the Richmond Hill mine in the Dakotas. The engineered cap minimizes infiltration while the bottom barrier drain prevents groundwater from entering the waste. In some cases, interim treatment of the drainage could be required.

4.9.2.4 Option 4 – Pit Disposal – Perched Saturated Layer in Cover

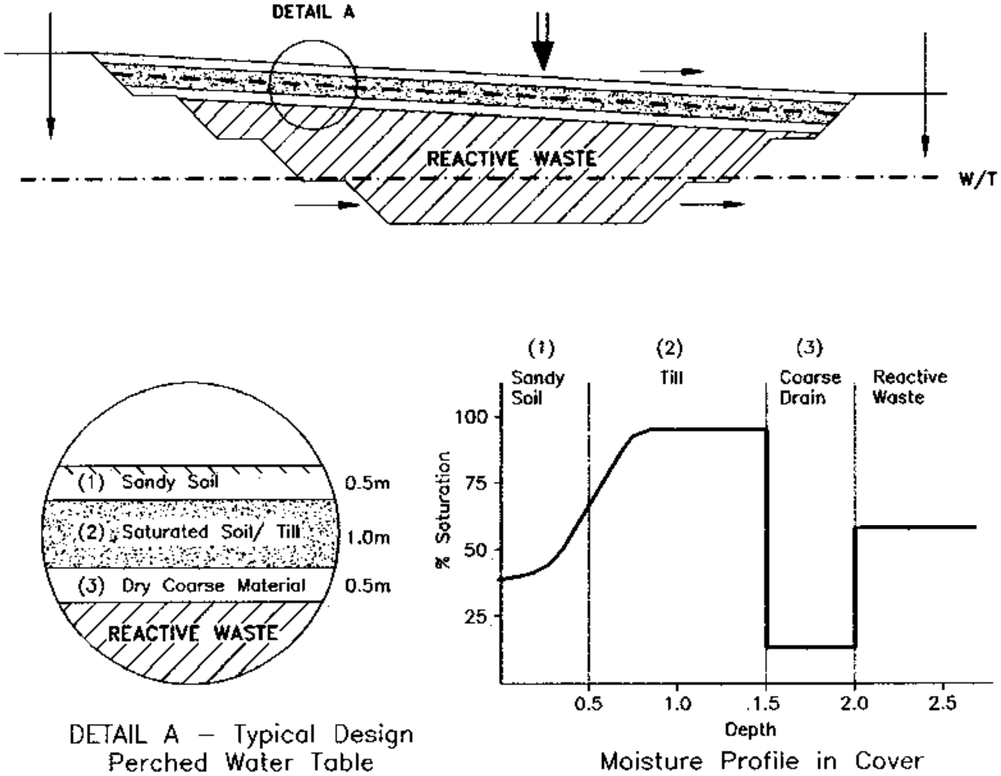
Option 4 is a dry disposal technique that includes a saturated oxygen barrier in the cover over the dry wastes. An example is shown in Figure 4.9-7. Substantial research is being conducted on this concept. The challenge in the design is the development of the saturated layer. This layer could be thickened tailings, tills, or any well-graded material that has a naturally high degree of saturation. Most areas of Canada have local tills that could perform well as the saturated layer. An example of the application of this concept is the Equity Silver Southern Tail pit near Houston, in the central interior of British Columbia. Approximately 5.0 million tonnes of potentially acid generating waste rock from the Main Zone pit was backfilled into the mined-out Southern Tail pit. Aziz and Ferguson (1997) report that compacted clay covers were constructed over the Southern Tail waste pile in 1990, and progressively constructed over the balance of the resloped Main and Bessemer waste rock piles between 1990 and 1994. The covers were

Figure 4.9-6 **Pit Disposal Concepts – Dry Disposal**



Source: MEND 2.36.1

Figure 4.9-7 **Pit Disposal Concepts – Perched Saturated Layer in Cover**



Notes: Zone 1 Permeable zone of sandy soil to allow infiltration to maintain till near saturation and to protect till from evapotranspiration and dessication

Zone 2 Well graded glacial till with naturally high level of saturation

Zone 3 Coarse drain to act as a capilliary break and possibly low hydraulic conductivity barrier to infiltration

Source: MEND 2.36.1

designed to reduce the infiltration of water and air to the waste rock and hence decrease acid generation.

4.9.3 APPLICATIONS AND LIMITATIONS

4.9.3.1 Site-Specific Application

The backfilling of pits is highly dependent on site-specific conditions, and as such, the approach for individual sites may vary. Some planning and design aspects will be common to all approaches and are likely to include (MEND 2.36.1):

- The definition of the quantity, mineralogy, acid generating characteristics and oxidation state of the proposed backfill;
- An evaluation of the future mining potential of the pit or adjacent underground workings and mineralized zones. This can, as an example, include mineralization remaining below the pit;
- Hydrogeological studies to determine the final flood level of the pit (or position of the water table), flooding rate and flooding acceleration options, potential for contaminated plume and impact on water resources, and effect of connections to other mine workings;
- Geotechnical studies to determine the characteristics of the backfill and pit walls;
- Modelling studies to predict pit lake water quality, and physical behaviour (e.g. stratification);
- Monitoring requirements; and
- Passive and chemical effluent treatment requirements.

At the Rabbit Lake in-pit tailings management facility, located in northern Saskatchewan, deposited tailings during winter become frozen in layers of 1 to 5 metres in thickness. During the summer period, previously frozen tailings become buried and remained frozen. The presence of frozen layers could jeopardize the objectives of the pervious surround method of deposition by preventing full consolidation of the tailings. A full-scale trial program of warm tailings injection was begun in 1999. Preliminary findings have shown that deep tailings injection under gravity head is possible and that positive thawing benefits can be realized (Landine *et al.* 2001).

4.9.3.2 Open Pit Planning

The physical layout of an open pit mine, and the locations of open pit mine waste disposal areas, are planned in advance of ore mining and waste stripping operations. The pit planning process focuses on optimizing the profitable extraction of ore in a safe and environmentally responsible manner. In concept, the economic value of a tonne of ore must be greater than the cumulative costs associated with the mining and processing of that tonne of ore, plus any other costs associated with the stripping and disposal of perhaps several tonnes of mine waste per tonne of ore.

Economic evaluations of open pits at the planning stage are typically used to develop a cut-off grade which is the minimum grade of ore that can be mined profitably or, in some cases, to breakeven. The cut-off grade is different for each open pit mine and is a function of the anticipated revenues and costs. The cut-off grade and the physical limits of an open pit mine are, therefore, sensitive to changes in revenue (e.g. metal price fluctuations) and costs (i.e. mining, processing, taxes, mine decommissioning, etc.).

At most new open pit mines, the potential for acidic drainage is determined early in the planning process. A decision can be made at the planning stage to segregate reactive wastes and relocate the wastes to the open pit once it is mined out. In such a case, the economic evaluation of the open pit and the pit design would take into consideration all anticipated costs to implement an in-pit disposal program. In some cases, it may be possible to delay the onset of acid generation. In this vein, Sheremata *et al*. (1991) investigated the addition of limestone to waste rock in laboratory experiments and showed that acid generation could be delayed.

The separate stockpiling of mine waste rock may be difficult to carry out in practice due to the mixing of wastes during the blasting and removal cycle, and the availability of suitable storage areas within an economic truck haul distance of the open pit. At some operation, with more than one pit, it may be possible to dispose the waste rock in mined-out pits (e.g. Rabbit Lake, Equity Silver).

The economic evaluation and design for an operating open pit mine, where acidic drainage from mine waste was not anticipated, would likely have been based on the permanent disposal of mine wastes in engineered waste rock stockpiles and tailings disposal areas. At some sites mine wastes have later been found to have the potential to produce acid; and as a result, alternate mine waste management strategies needed to be developed and implemented.

4.9.3.3 Mining Related Constraints

The residual (uneconomic) mineralization may be economically recoverable in the future. In addition, pit backfilling may not be practical if it impacts existing or potential future mine workings. There may also be concern over the safety aspects of working in an old pit where the pit walls may pose a risk.

4.9.3.4 Pit Wall Mineralization

Seepage from mineralization along joints, fractures, faults and exposed pit walls may be acidic and may impact on the utility of using a pit for waste disposal.

4.9.3.5 Hydrogeology/Hydrology

The hydrogeology of a pit is often the critical factor in assessing the applicability of a pit for waste disposal, and often determines what engineered controls are necessary. Key factors to consider are: the presence of faults and major flow pathways; the bulk permeability of the rock around the pit; hydraulic connections to other mine openings; groundwater flow paths and potential downstream receptors; the position and gradient of the groundwater table; and the stratigraphy/permeability of the overburden and bedrock.

Ideally, a pit used to dispose of wastes that produce a contaminated leachate would have minimal to no groundwater gradient across the pit, and low permeability bedrock with few faults and fracture zones. However, few pits have these ideal characteristics, but may have other hydrogeologic features that will make them suitable for waste disposal. These include:

- Minor groundwater inflows which result in negligible downstream environmental impact; and
- Naturally beneficial geochemical characteristics in surrounding rock that buffer acidity and mitigate contaminant migration.

The hydrology of the region surrounding a pit is of interest as it may affect; the flooding of the pit, the final water table elevation, the concentrations of contaminants in leachate, dilution, and flowpaths between a backfilled pit and other surface water bodies.

4.9.3.6 Environmental Aspects

Consideration should be given to both the short-term and long-term environmental implications of an in-pit disposal concept. Every site is unique, and as such a pit can have site-specific constraints (i.e. related to groundwater and surface water quality and use, and sensitive ecological communities) to the implementation of an in-pit disposal program.

4.9.3.7 Regulatory Constraints and Considerations

The in-pit disposal of mine wastes is generally well received by regulators. As an example, the Province of British Columbia (BC RAC 1993) Interim Policy for Acid Rock Drainage at Mine Sites stated:

> *"A water cover is currently an acceptable form of acid rock drainage prevention for underground workings or open pits.*
>
> *The timing and inflow rate requirements in flooding open pits and underground workings will be based on the hydrologic conditions, the relative reaction rates of acid generation versus neutralization and the potential release of acid products. Proponents must demonstrate that any water released to the environment will be of acceptable quality."*

4.9.4 BASE METHODS

MEND 2.36.1 identified over sixty in-pit disposal sites. A partial list of in-pit disposal sites located in Canada and the United States are listed in Tables 4.9-2 and 4.9-3 respectively. The data show that in-pit disposal has been widely practiced, with underwater in-pit disposal being commonly used. Twelve in-pit disposal case studies (Table 4.9-4) are presented in the report. These case studies serve as useful references for persons involved in the planning or implementation of an in-pit disposal program. Three of the case studies are briefly reviewed.

1. The Berkeley Pit. The Berkeley Pit is a well-known example of a serious acidic drainage problem in a flooding pit. The pit, located in Butte, Montana, is a Superfund site, and has been the subject of numerous technical investigations. The pit is 1,780 feet deep and encompasses an area of 675 acres, and a volume of 26 billion gallons of contaminated water (MEND 2.36.1). MEND 2.36.1 reviews the history of the pit and regulatory status, as well as pit lake water quality investigations and results. Jonas (2000) presents the results of more recent Berkeley pit lake sampling (e.g. fall 1997 and spring 1998) undertaken to assess seasonal effects on the pit lake water column.

2. The Solbec Pit. The Solbec base metal mine, located in the Eastern Townships of Québec, operated from 1962 to 1970 to produce approximately 1.75 million tonnes of ore, including 1.4 million tonnes from underground cut and fill stopes, and 0.35 million tonnes from an open pit (crown pillar). Most of the waste rock was used as surface fill or stock piled at the site. In 1987 a feasibility study was carried out to investigate the relocation of the waste rock to the pit where it would remain submerged. A bathymetric survey was completed and the relocation proposal appeared feasible.

The pit was the subject of intensive field investigations including a hydrogeological study undertaken under the auspices of MEND (MEND 4.8.1). The pit was dewatered prior to the disposal of oxidized waste rock that was relocated from the mine/mill site. An initial estimate of the volume of waste rock to be relocated to the pit was 115,400 m^3 over an area of 7.4 ha. A total of 276,000 m^3 of wastes including contaminated soils were ultimately removed from an area of 12.1 ha.

Table 4.9-2
List of In-Pit Disposal Sites In Canada

Name of Site	Location	Status	Types of Wastes	Cover	Monitoring Data
Cluff “D”	Cluff Lake, Saskatchewan	Active	Reactive pit walls	Wet	SW & GW
Collins “B”	Rabbit Lake, Harrison Peninsula, Saskatchewan	Active	Reactive waste rock	Wet	SW & GW
Owl Creek	Timmins, Ontario	Historic	Reactive waste rock	Wet	SW
Solbec	Aylmer Lake, Eastern Townships, Stratford, Québec	Historic	Reactive waste rock, tailings and contaminated soil	Wet	SW & GW
Gunnar	Lake Athabasca, Crackingstone Peninsula, Saskatchewan	Historic	Reactive waste rock and uranium tailings	Wet	SW & GW
Pits #1 and #2 (Gloryholes)	Buchans, Newfoundland	Historic	Reactive mine tailings	Wet	SW
East Kemptville	Nova Scotia	Historic	Sludge	Wet	-
Rabbit Lake	Rabbit Lake, Saskatchewan	Active	Reactive tailings	Wet	SW & GW
Island Copper	Vancouver Island, British Columbia	Proposed	Reactive waste rock	Wet	-
Brenda Mine	Peachland, British Columbia	Historic	None	Wet	SW & GW
Brunswick No. 6	Bathurst Area, New Brunswick	Historic	Reactive waste rock and tailings	Wet	SW & GW
Heath Steele “A”	Bathurst Area, New Brunswick	Proposed	Waste rock	Wet	SW & GW
Mattabi and “F” Group	Ignace, Ontario	Historic	Waste rock and sludges	Wet	SW & GW
War Eagle	Whitehorse, Yukon	Historic	Municipal waste	Wet	SW
Gibraltar	McLease Lake, British Columbia	Inactive	No waste disposed in flooded pit	Wet	-
Highland Valley Copper	Highland Valley Area, British Columbia	Historic	No waste disposed in flooded pit	Wet	-
Similco Ingerbelle	Similkameen Area, British Columbia	Historic	No waste disposed in flooded pit	Wet	-
East Sullivan Mines	Val d'Or, Québec	Historic	Flooded crown pillar	Wet	SW & GW
Crown Pillar	Ignace, Ontario	Historic	Reactive mine tailings (AMD)	Combined	GW
Deilmann	Key Lake, Saskatchewan	Active	Reactive waste rock and tailings	Combined	GW
Bell Mine	Babine Lake, Newman Peninsula, Saskatchewan	Historic	Reactive waste rock and tailings	Combined	SW
Stratabound	Bathurst Area, New Brunswick	Historic	Reactive waste rock	Combined	SW & GW
Equity Silver	Houston, British Columbia	Historic	Waste rock	Combined	SW & GW
Nickel Plate South	Penticton Area, British Columbia	Historic	Waste rock	Combined	SW
Mount Washington	Vancouver Island, British Columbia	Historic	Waste rock	Dry	SW & GW
Faro Mine	Vangorda Creek-Anvil Area, Yukon	Historic	Planned-tailings	Wet	SW & GW
Sturgeon Lake	Ignace, Ontario	Historic	Waste rock	Wet	SW

Table 4.9-3

List of In-Pit Disposal Sites in the United States

Name of Site	Location	Status	Types of Waste	Cover	Monitoring Data
Midnite Mine (Pits 3 and 4)	Wellpinit, Washington	Historic	Reactive waste rock and tailings	Wet	SW & GW
Nevada Precious Metal Mines	Nevada	Historic	Reactive waste rock and tailings	Wet	GW
Robinson Mining (Veteran, Tripp, Liberty, Ruth and Kimbley pits)	White Pine, Nevada	Historic & proposed to reinitiate	Reactive waste rock and tailings	Wet	SW & GW
Berkeley	Butte, Montana	Historic	Acid mine water from reactive waste rock, tailings and underground area	Combined	SW & GW
Iron Mountain Mine	Shasta, California	Historic	Acid mine water from reactive waste rock and tailings	Combined	SW & GW
Jackpile-Paguate	Albuquerque Area, New Mexico	Historic	Waste rock, overburden	Combined	SW
Placer Mining (Gold Mine)	Montana	Active	Non-reactive waste rock and overburden	Dry	SW
Calaveras Asbestos Mine	Calaveras, California	Historic	Asbestos fiber and tailings	Dry	SW & GW
Hecla Mining - Yellow Pine Gold Mine	McCall Area, Idaho	Historic	Non-reactive waste rock	Dry	None
Summitville Mine	Del Norte Area, Colorado	Historic/Active	Reactive waste rock and tailings	Dry	SW & GW
Coaltrain Corp.	Bakerstown and Pittsburgh, West Virginia	Active	Coal refuse	Dry	-
Bakerstown and Freeport	West Virginia	Active	Waste rock, overburden	Dry	SW and GW
Richmond Hill Mine	South Dakota	Active	Waste rock	Dry	GW

Table 4.9-4
Case Study Sites

Case Study No.	Open Pit	Location	In-Pit Disposal Concept
1	Owl Creek	Ontario	Underwater Disposal with Alkaline Blending
2	Rabbit Lake	Saskatchewan	Underwater Disposal with Groundwater Barriers
3	Collins “B”	Saskatchewan	Underwater Disposal
4	Island Copper	British Columbia	Underwater Disposal
5	Solbec	Quebec	Underwater Disposal with a Top Barrier
6	Udden	Sweden	Underwater Disposal
7	Stratabound CNE	New Brunswick	Underwater Disposal with a Top Barrier
8	Robinson	Nevada, U.S.A.	No Waste Disposed - Flooded Pit Study
9	Gunnar	Saskatchewan	No Waste Disposed - Flooded Pit Study
10	Cluff “D”	Saskatchewan	No Waste Disposed - Flooded Pit Study
11	Deilmann	Saskatchewan	Underwater Disposal with a Top Barrier
12	Berkeley	Montana, U.S.A	No Waste Disposed - Flooded Pit Study

Note: Detailed case studies of these sites are included in Volume 2 of MEND 2.36.1

The case study reviews the technical studies that were completed in ascertaining the viability of the in-pit disposal program, the costs of the program, which were equivalent to $3.87/m^3 (in 1993 dollars) for the relocated waste. The underestimation of the waste volume, and the limited pit volume available to flood the reactive waste resulted in the construction of a water retaining containment structure across the south end of the pit to allow for the waste to be flooded. Preliminary results are positive. A multi-media passive treatment system was constructed to treat the effluent to reduce the zinc concentration. The levels of zinc are expected to reach discharge levels in the near future. The water quality-monitoring program is also reviewed in the case study.

3. The CNE Pit. The CNE lead-zinc open pit operated between 1990 and 1992. At the mine feasibility stage, the operators of this small open pit recognized that the waste rock contained in the order of 10% pyrite, and undertook to stockpile the waste rock to facilitate rock dump seepage and runoff collection and chemical treatment. The waste rock was later relocated to the open pit and clay capped. The case study reviews the completed in-pit disposal program and water quality monitoring data for the clay cap seepage, and groundwater collected from an inclined diamond drill hole that extends beneath the filled pit.

Two in-pit disposal programs that are not described in MEND 2.36.1 are outlined below.

Lanteigne (2000) reports that INCO Limited operated the Whistle open pit mine, located approximately 30 kilometres north of Sudbury, from 1988 to 1991 and again from 1994 to 1998 to produce nearly 5 million tons of ore and over 7 million tons or 4 million cubic yards of waste rock. The waste rock is stored in two waste dumps immediately north of the open pit, covering an area of approximately 35 acres. The open pit itself has an estimated volume of 4,150,000 cubic yards, and covers an area of 1,045,900 square feet. The waste rock was characterized under MEND 1.41.4.

Closure plans for this site include the relocation of the waste rock into the pit and the placement of an engineered cover over the filled pit to reduce both oxygen and water infiltration. The waste rock is currently being placed back into the pit in 2.5 metre lifts; material is dumped, spread, and compacted in shallow lifts in order to avoid excessive settlement in the fill material once the cover has been placed. The final lift of waste rock will be reduced to four feet and will comprise smaller sized particles to facilitate better compaction and a smoother surface for the cover. The cover will consist of a barrier material with a low hydraulic conductivity to minimize the infiltration of water into the pit, as well as a coarse-grained layer that will act as an evaporative break and ensure the saturation of the barrier material. The cover design also includes a geotextile that will separate the waste rock from the barrier material and add an element of tensile strength to the cover in the event of greater-than-expected settlement of the waste rock.

Test plots have been constructed on site in September 2000 to test a variety of potential barrier materials: a sand-bentonite mixture, a silt/trace clay, and a geosynthetic clay liner (GCL). The plots have been extensively instrumented with lysimeters, an O_2/CO_2 gas measurement system, soil suction and temperature sensors, *in situ* volumetric water content sensors, a surface runoff collection and monitoring system as well as a meteorological station. The performance of the various covers will be monitored over the next few years to determine the most suitable material for full-scale use. Similar performance monitoring instrumentation will be installed in the final pit cover. Geochemical modelling was conducted by SENES (MEND 1.41.4) to simulate the pit filling process and determine the predicted pit water quality after closure. The results showed that the addition of one kilogram of lime per tonne of waste rock will effectively neutralize the acidity as well as lower the concentration of metals in the pit water overflow. Lime is now being added at the prescribed rate to each load of waste rock prior to dumping. Acidity testing of the waste rock is being conducted on a regular basis to ensure that the lime addition rates are appropriate.

In-Pit Disposal at the Fosterville Sulphide Project: Trevenen *et al*. (2000) report that the in-pit disposal of tailings was identified to be the most suitable method in relation to the proposed expansion of the Fosterville Sulphide Project located in Central Victorian Goldfields, Australia.

The proposed approach would use a conventional wet deposition method with a slimes layer. Water management would encompass tailings dewatering and a system of dewatering wells around the pit perimeter.

4.9.5 COSTS

Costs are site specific and strongly dependent on the in-pit disposal option selected. However, as a general comment, unit costs for disposal of waste may fall in the range of $1 to $2 per tonne of waste rock, and $1 to $3 per tonne of tailings before site-specific factors. In comparison, an IGWG-Industry Task Force on Mine Reclamation (MEND 5.8 and Feasby and Jones 1994) estimated the financial liability associated with acidic drainage from some 738.9 million tonnes of acid waste rock and 1,877.7 million tonnes of tailings. The equivalent unit liabilities (converted to year 2000 dollars) are $1.95/t acid tailings, and $3.20/t acid waste rock.

MEND 2.36.1 demonstrates that in-pit disposal programs provide significant cost saving in comparison to other closure options for tailings and waste rock. In particular, in-pit disposal can be cost advantageous for the disposal of historic wastes, provided conditions are suitable. The application of in-pit disposal techniques is, however, subject to site specific conditions and regulatory review, and as such these programs need to have well founded technical and environmental bases.

4.9.6 MEND AND RELEVANT PUBLICATIONS

MEND 1.41.4 1997. Whistle Mine Waste Rock Study. December.

MEND 2.17.1 1996. Review of Use of an Elevated Water Table as a Method to Control and Reduce Acidic Drainage from Tailings. March.

MEND 2.36.1 1995. Review of In-Pit Disposal Practices for the Prevention of Acid Drainage – Case Studies. September.

MEND 2.36.3 1999. Assessing the Subaqueous Stability of Oxidized Waste Rock. April.

MEND 4.5.1 1994. Review of Waste Rock Sampling Techniques. June.

MEND 4.8.1 1993. Auscultation de la fosse Solbec – Rapport final, première année d’observation. Mars.
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