

**REVIEW OF WATER COVER
SITES AND RESEARCH
PROJECTS**

MEND Project 2.18.1

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on

**REVIEW OF WATER COVER SITES
AND RESEARCH PROJECTS**

submitted to

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by

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EXECUTIVE SUMMARY

In 1996, the Geotechnical Research Centre at The University of Western Ontario initiated a project to review water cover research and applications. The objectives of the project were to document the design and performance of water covers used to prevent acid generation in sulphide-bearing mine waste and compile results from water cover research. Water cover sites in Canada, Norway and Sweden were reviewed. Research programs reviewed were from Canada, Norway and the United States. They include MEND-sponsored work at the CANMET Elliot Lake Laboratory and Noranda Technology Centre, Pointe-Claire, Québec, and research undertaken by former United States Bureau of Mines, Norwegian Water Research Institute and Norwegian Hydrotechnical Laboratory.

The Province of Ontario, Canada Centre for Mineral and Energy Technology (CANMET) and Brunswick Mining and Smelting Corporation provided funding for the project.

The survey of water cover sites showed that the selection of the minimum depth of water is based on maintaining saturation of tailings in the event of a drought and preventing tailings resuspension. Predictions of resuspension are based on empirical correlations with bed shear stress, bed water velocity, or the wave height to water depth ratio. These correlations were generally obtained from experiments on particles other than tailings. Estimates of bed shear stress, velocity, or wave height are calculated from empirical equations relating these parameters to wind speed and pond fetch length, using linear wave theory. There was no conclusive verification of resuspension prediction at any of the sites we reviewed.

Methods presently used to predict the quality of pond and effluent waters do not consider many biological, geochemical and physical phenomena that may significantly affect water cover performance. In general, not enough long-term field data are available to verify water quality predictions, however, at one site field data have already shown the predictions to be unconservative.

The character of the reviewed sites, in terms of site hydrology, the initial composition of the tailings, the degree of oxidation of the tailings prior to flooding, the physical layout of the site, and the treatment of the tailings before or after flooding, varies considerably. Due to the dissimilarity of the sites examined, no correlation between water depth and design performance could be made. Observations common to many sites include a dramatic increase in pH during the summer, possibly due to microfauna activity, CO₂ degassing and increase in alkalinity from surface runoff, and the formation of a thin coating of orange iron oxyhydroxides and organic matter at the tailings-water interface.

In column and lysimeter experiments for both oxidized and unoxidized tailings, flooding produces a thin surface coating of brownish-orange iron oxyhydroxides which in previously unoxidized tailings marks the extent of vertical oxidation into the tailings. The finite thickness of the oxidation zone apparently occurs because of the attainment of steady-state conditions between pyrite oxidation and sulphate reduction to sulphide. With time, the growth of organic deposits at the surface occurs, which further impedes upward metal flux to the water cover. Column experiments on oxidized tailings show that flooding causes the dissolution of oxidation products and the subsequent release of metals into the water cover, often resulting in metal concentrations higher than permissible levels. The dissolution of some oxidation products, or other minerals, may be slow and contribute to high metal concentrations in the water cover for many years. The use of a sand or a peat protective cover at the tailings-water interface reduces metal fluxes into the water cover to negligible levels. However, metal flux into the water cover may resume when the absorptive capacity of the protective cover is reached. The use of a peat layer creates highly anoxic and reducing conditions in the tailings. Peat also produces acidity that can significantly increase the mobility of some metals in the tailings pore water.

The addition of lime to previously unoxidized tailings prior to flooding inhibits the activity of iron-oxidizing bacteria by keeping pH high. Lime addition to oxidized tailings increases pH and limits the mobility of most metals in the water cover and pore waters. Mixing the top portion of tailings with lime was found to be more effective than simply adding the lime to the surface. Traditional methods used to estimate the required amount of neutralizing material, such as acid-base accounting, were inaccurate when applied to long-term (37-week) experiments on flooded tailings.

Laboratory test results show that coarser tailings under a water cover initially generate higher amounts of acid drainage than finer tailings. This could be due to the initially faster rate of vertical oxygen transport in coarser tailings.

The effectiveness of a Biologically Supported Water Cover (BSWC) has been demonstrated in the laboratory. A BSWC involves colonizing flooded tailings with plants to facilitate organic buildup for limiting resuspension and metal flux through biological adsorption. Preliminary results suggest that its effectiveness in the field is dependent on the depth of water and wind conditions.

It is apparent from laboratory and field studies that flooding tailings is the most successful method presently known for preventing and controlling ARD. However, in many cases the water cover discharge will require treatment to meet regulatory standards, and it is not currently possible to accurately predict the required amount and length of treatment. State-of-the-art predictive contaminant models ignore many important phenomena in water covers, including secondary mineral dissolution and metal release independent of oxidation and resuspension. Some of the work reviewed hypothesized that flooding will eventually establish a diagenetic environment for some tailings minerals with accumulation of organic matter. It is, however, uncertain how long this will take to develop and whether it will occur under all conditions. Furthermore, some metals may continue to be released to the water cover in such an environment. Although water covers are a promising technology, fundamental understanding of many phenomena influencing their performance is unknown, and the minimum depth of water required cannot presently be confidently determined. The present prediction tools do not consider other important tailings properties such as thixotropy and cohesion.

We recommend that additional research and field work be initiated to address the uncertainties and issues raised in this review. The key areas for such work may include verification of resuspension predictions at a few selected sites, fundamental understanding of tailings properties and behaviour, the nature and rate of accumulation of the organic matter-iron hydroxide sediment at the tailings-water interface, and the contribution of resuspension to tailings oxidation and long-term water quality.

SOMMAIRE

En 1996, le Centre de recherches géotechniques de l'université Western Ontario a entrepris un projet de révision des recherches faites sur les couvertures aqueuses et de ses applications. Le projet visait à créer une documentation sur la conception et l'efficacité d'une couverture aqueuse pour empêcher la production d'acide dans les déchets miniers renfermant des sulfures et à compiler les résultats des recherches sur les couvertures aqueuses. Les emplacements où une couverture aqueuse est employée au Canada, en Norvège et en Suède ont fait l'objet d'une revue. Les programmes de recherche revus venaient du Canada, de la Norvège et des États-Unis. Ils incluaient les travaux effectués sous le parrainage du Programme NEDEM au laboratoire de CANMET d'Elliot Lake et au Centre de technologie Noranda à Pointe-Claire au Québec ainsi que la recherche entreprise par l'ancien *United States Bureau of Mines*, le *Norwegian Water Research Institute* et le *Norwegian Hydrotechnical Laboratory*.

L'Ontario, le Centre canadien de la technologie des minéraux et de l'énergie (CANMET) et la *Brunswick Mining and Smelting Corporation* ont assuré le parrainage du projet.

L'étude des parcs à résidus où la technique des couvertures aqueuses est employée a montré que la profondeur minimum d'eau est choisie de façon à maintenir la saturation des résidus en cas de sécheresse et à empêcher leur remise en suspension. Les prévisions relatives à la remise en suspension sont basées sur des corrélations empiriques avec le cisaillement du lit, la vitesse d'écoulement de l'eau dans le lit ou le rapport creux de vague/profondeur d'eau. Ces corrélations sont généralement obtenues à partir d'expériences sur des particules autres que les résidus. Les estimations du cisaillement du lit, de la vitesse d'écoulement ou du creux de vague sont calculées au moyen d'équations empiriques reliant ces paramètres à la vitesse du vent et au fetch de l'étang en faisant appel à la théorie des ondes linéaires. Les prévisions relatives à la remise en suspension n'ont été vérifiées de façon concluante à aucun des sites étudiés.

Les méthodes utilisées actuellement pour prévoir la qualité des effluents et des eaux de parcs à résidus ne tiennent pas compte de plusieurs phénomènes biologiques, géochimiques et physiques pouvant influencer de façon significative sur l'efficacité des couvertures aqueuses. En général, il n'existe pas suffisamment d'études à long terme sur le terrain pour vérifier les prévisions sur la qualité de l'eau. Toutefois, les données recueillies à un des parcs à résidus ont déjà montré que les prévisions ne sont pas précises.

Le caractère des parcs à résidus étudiés varie beaucoup en fonction de leur hydrologie, de la composition initiale des résidus, du degré d'oxydation des résidus avant leur inondation, de la disposition de l'emplacement et du traitement des résidus avant ou après leur inondation. Vu les dissimilitudes des parcs étudiés, il a été impossible de faire quelque corrélation que ce soit entre la profondeur de l'eau et l'efficacité de la couverture aqueuse. Certaines observations sont communes à bien des sites, entre autres : une augmentation dramatique du pH durant l'été, due possiblement à l'activité de la microfaune, un dégagement de CO₂ et une augmentation de l'alcalinité causée par le ruissellement de surface et la formation d'une mince couche orange d'oxyhydroxydes de fer et de matières organiques à l'interface résidus/eau.

Au cours d'expériences sur colonne et en lysimètre, l'inondation des résidus, oxydés ou non, produit une fine couche de surface orange brunâtre d'oxyhydroxydes de fer qui, dans les résidus non oxydés auparavant, marque l'étendue de leur oxydation verticale. L'épaisseur de la zone d'oxydation est apparemment définie par l'atteinte des conditions d'état stationnaire entre l'oxydation de la pyrite et la réduction des sulfates en sulfures. Avec le temps, il se produit un dépôt organique à la surface qui empêche encore le déplacement des métaux vers l'eau de la couverture. Les expériences sur colonne réalisées avec des résidus oxydés montrent que l'inondation provoque la dissolution des produits d'oxydation et, par la suite, la libération de métaux dans la couverture d'eau, entraînant des concentrations de métaux supérieures aux teneurs acceptables. La dissolution de certains produits d'oxydation ou d'autres minéraux peut être lente et contribuer à entraîner pour bien des années des concentrations élevées de métaux dans l'eau de la couverture.

L'utilisation d'une couverture protectrice de sable ou de tourbe à l'interface résidus/eau réduit le flux de métaux dans la couverture à des niveaux négligeables. Toutefois, le flux de métaux dans la couverture aqueuse peut recommencer lorsque le pouvoir absorbant du revêtement protecteur est atteint. L'emploi d'une couche de tourbe crée des conditions réductrices fortement anoxiques dans les résidus. La tourbe produit aussi une acidité qui peut augmenter fortement la mobilité de certains métaux dans l'eau interstitielle des résidus.

L'addition de chaux aux résidus non oxydés avant leur inondation inhibe l'activité des ferrobactéries en gardant le pH élevé. L'addition de chaux aux résidus déjà oxydés augmente le pH et limite la mobilité de la plupart des métaux dans la couverture aqueuse et l'eau interstitielle. Le mélange des résidus de surface avec la chaux s'est avéré plus efficace que la simple addition de chaux à leur surface. Les méthodes traditionnelles employées pour évaluer les quantités requises de neutralisant comme la détermination des acides-bases se sont révélées imprécises lorsque appliquées à des résidus inondés à long terme (37 semaines).

Les résultats d'essais en laboratoire montrent que, par rapport aux résidus plus fins, les résidus grossiers immergés dans l'eau engendrent un drainage acide plus important. Il se peut que cela soit dû à un transport vertical plus rapide de l'oxygène dans les résidus plus grossiers.

L'efficacité d'un système d'immersion à l'eau avec support biologique (BSWC) a été démontrée en laboratoire. Le BSWC met en oeuvre la colonisation des résidus inondés par des plantes de façon à favoriser, par adsorption biologique, l'accumulation de matières organiques pour limiter la remise en suspension et le flux de métaux. Les résultats préliminaires obtenus semblent indiquer que l'efficacité sur le terrain dépend de la profondeur d'eau et des conditions éoliennes.

D'après les études de laboratoire et de terrain, l'inondation des résidus s'avère, parmi les méthodes actuellement connues, la méthode la plus efficace de prévention et de contrôle du drainage rocheux acide. Toutefois, dans bien des cas, les eaux de la surverse du parc à résidus devront être traitées pour respecter les normes réglementaires et il est actuellement impossible de prévoir avec précision l'importance et la durée du traitement nécessaire. Les modélisations de

pointe pour la prévision de la contamination ne tiennent pas compte de bien des phénomènes importants dans les systèmes de couverture aqueuse, entre autres : la dissolution secondaire des minéraux et la libération de métaux sans oxydation ni remise en suspension. Certains des travaux ayant fait l'objet de la revue émettaient l'hypothèse que l'immersion créera, avec l'accumulation de matières organiques, un milieu diagénetique pour certains résidus. On ne sait pas cependant avec certitude le temps qu'il faudra pour qu'il en soit ainsi, ni même si cela sera le cas dans toutes les conditions. En outre, dans un tel environnement, certains métaux peuvent continuer d'être libérés dans l'eau de la couverture. Bien que l'inondation soit une technologie prometteuse, bien des phénomènes qui influent sur son efficacité ne sont pas connus fondamentalement et la profondeur minimum d'eau requise ne peut être déterminée avec certitude actuellement. Les outils prévisionnels actuels ne tiennent pas compte de certaines autres propriétés importantes des résidus comme la thixotropie et la cohésion.

Nous recommandons que d'autres recherches et d'autres travaux sur le terrain soient entrepris pour éliminer les incertitudes et répondre aux questions soulevées dans la présente revue. Les domaines clés de ces travaux peuvent inclure la vérification des prévisions relative à la remise en suspension à quelques sites choisis, la compréhension fondamentale des propriétés et du comportement des résidus, la nature et la vitesse de l'accumulation de sédiments renfermant des matières organiques et de l'hydroxyde de fer à l'interface résidus/eau et la contribution de la remise en suspension à l'oxydation des résidus et à la qualité de l'eau à long terme.

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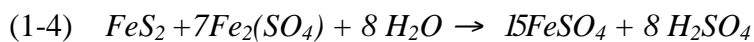
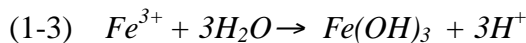
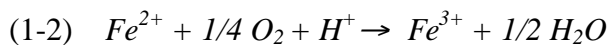
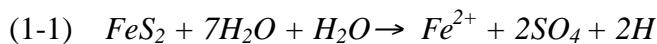
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1. INTRODUCTION AND OBJECTIVES

1.1. ACID ROCK DRAINAGE (ARD) FORMATION

The prevention and control of Acid Rock Drainage (ARD) from mine wastes is a major environmental problem faced by the mining industry in Canada and around the world. The recent failure of a tailings pond in Guyana, and the subsequent environmental impact, underscores the problem and has raised public awareness on the issue. In Canada, at least 15,000 hectares of land are covered with mine wastes, which will take an estimated \$3 billion, with current technologies, to manage (Feasby et al, 1991).

The formation of acidic drainage in tailings and waste rock deposits starts with the oxidation of sulphide minerals, such as pyrite and pyrrhotite, which occurs when they are exposed to oxygen and moisture. The reactions of pyrite oxidation are:

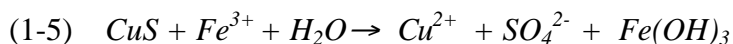


Equation (1-1) describes the direct oxidation of pyrite by oxygen, while Equations (1-2) and (1-3) describe the oxidation of ferrous to ferric iron and the subsequent precipitation of ferric iron as ferric hydroxide. Equation (1-4) describes the oxidation of pyrite by ferric iron. Ferric iron has been shown to oxidize pyrite at a much faster rate than oxygen. Recent research by Moses and Herman (1991) suggests that the role of oxygen is primarily to regenerate ferric iron by oxidizing ferrous iron produced in Equation (1-4) at neutral and acidic low pH values.

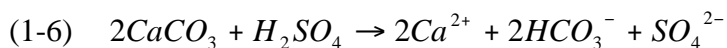
The rate of oxidation is dependent on several factors, including oxygen supply, temperature, pH, bacterial activity, and surface area of both the tailings particles and pyrite grains. While the first three factors influence the rate of reaction, the surface area determines how much reactive mineral is exposed to oxygen and therefore available for oxidation. Initially, at neutral pH, the rate of reaction is limited by the chemical oxidation of Fe (II) to Fe (III) by oxygen, which is slow. However, at lower pH, microbes such as *Thiobacillus ferrooxidans* can catalyze this reaction and increase the overall rate of oxidation by as much as two orders of magnitude.

Other mechanisms that create Fe(III), such as the oxidation of iron by manganese oxide or the dissolution of ferric hydroxides catalyzed by the presence of organic compounds (ligands), may play an important role in pyrite oxidation.

ARD results when the products of sulphide mineral oxidation (sulphate, acidity, iron, and other metals) are entrained in water flowing through a tailings or waste rock dump. The increased acidity of the water will in turn lead to leaching of heavy metals, such as Cu, Zn, Pb, and Hg. Ferric iron may directly leach metals from sulphide ores. An example of this is the dissolution of CuS:



However, the acidification of the water may be impeded by the dissolution of neutralizing minerals such as carbonates:



The released alkalinity (HCO_3^-) can precipitate metals as carbonates and consequently limit their mobility. Some mine wastes with a low concentration of sulphide minerals and a high concentration of neutralizing minerals will not produce ARD.

Sulphide mineral oxidation is often characterized by the precipitation of secondary minerals from the products of oxidation, such as hydroxides (for example, Equation (1-3) above) and sulphates. Although the precipitation of these minerals will initially limit metal mobility, changing conditions in a mine-waste pile, such as changes in pH or soluble metal concentrations, may cause these minerals to dissolve at a later date.

1.2. SUMMARY OF CURRENT MANAGEMENT PRACTICES

Current management practices for preventing and controlling ARD are reviewed below. Preventative strategies aim at stopping the initial formation of ARD by limiting the supply of oxygen to mine wastes, and/or maintaining a neutral pH by the addition of alkaline material.

1.2.1. Pumping and Treatment

The runoff and seepage from the tailings deposit maybe collected and treated by adding lime to increase pH and precipitate metals as sulphides and hydroxides. This generally reduces metal concentrations to environmentally acceptable limits. A disadvantage of this method is that some heavy metals (e.g., manganese) do not easily precipitate. Furthermore, this strategy does not lead to a long-term solution.

1.2.2. Blending and Segregation

One strategy to limit ARD is to blend acid-producing material with neutralizing material. However, the proportion of neutralizing to acid-producing material required cannot always be determined with certainty, as the acid-producing and neutralizing minerals may have different rates of reaction. The use of this method may require a large amount of neutralizing material. Segregation is a similar strategy, whereby layers of acid-producing tailings may be placed between layers of neutralizing minerals. The non-acid producing layers will neutralize water flowing from the acid-producing layers and may precipitate metals in the acid water. This strategy is, however, impractical for tailings and in waste rock, acid water may escape by channelling.

1.2.3. Dry Covers

The principal purposes of dry covers are: 1) to reduce the oxygen supply into mine-wastes by providing a moisture barrier, 2) to limit water percolation into the tailings and thereby reduce the amount of acid water formed, and 3) to reclaim the waste deposit for further use by revegetating the cover. Covers may be constructed of a combination of sand and clay layers, inert or non-reactive tailings, geosynthetics, and organic material.

To effectively stop oxygen transport, an engineered soil cover should include a layer of saturated fine-grained material. This serves as an oxygen and infiltration barrier. Coarse-grained soil layers are generally placed above the fine-grained layer to preserve its moisture content and hydraulic conductivity (Yanful et al. 1993, Yanful 1993). The main disadvantages with this type of cover include its relatively high cost and uncertainties regarding its durability because of possible increases in oxygen ingress and water percolation due to desiccation and freezing and thawing. Organic covers (such as wood waste), though preventing the ingress of oxygen, can produce acids that may initiate sulphide oxidation and metal leaching.

1.2.4. Water Covers

Many development projects and operating mines in Canada with an ARD concern are proposing water covers for controlling acid drainage from exposed surfaces and pits and underground workings, and from the large volumes of tailings and/or waste rock. This is because a water cover is considered the most effective of all the ARD prevention and control measures. The solubility of oxygen in water is quite low (11 mg/L at 20°C) and the diffusion rate of oxygen through water is about 10,000 times slower than through air. Thus, it is considered, that in the absence of convection, the rate of oxygen transport through water is too slow to be of any significance to sulphide mineral oxidation. It is believed that organic matter deposited in the water cover and accumulated at the tailings-water interface, will consume oxygen penetrating to the tailings and also impede the transport of oxidation products from the tailings to the water cover. The resulting anoxic environment enhances the diagenesis of metal sulphides, caused by the reduction of sulphate to sulphide by sulphate-reducing bacteria and subsequent mineralization with

metals. The establishment of thermodynamically stable conditions, under which very little metal release from tailings occurs, have been shown to develop in lakes where tailings have been deposited (Pelletier et al, 1991). Water covers also require reasonably low maintenance and are less costly than soil covers. Water covers do not have the same problems of possible cracking from freeze-thaw and desiccation. A water cover has one major disadvantage. Its implementation involves the construction of dams and dikes that require long-term attention and maintenance.

In spite of the enormous popularity and importance of water cover technology, there appears to be very little information in Canada on the parameters required for designing effective water covers. Several efforts have been initiated recently in Canada, the United States, Sweden and Norway on both fundamental and applied research into understanding and applying water covers on potentially acid producing mine tailings. Very little information has been reported from these research endeavours and is not readily available. As a result, it is not known what gaps in water cover technology currently exist. Both industry and regulatory agencies do not have the tools to reliably predict long-term performance. Presently mines proposing to use water covers are required to provide a somewhat arbitrarily selected water cover depth in conjunction with ongoing monitoring. The problem with this approach is the uncertainties involved, the high cost of monitoring and the associated logistical problems should the depth of water cover be found to be too shallow. Such a situation is not acceptable to industry and government.

1.3. OBJECTIVES OF THE PRESENT REVIEW

The present review was initiated in response to the need to identify and assemble existing information on water cover technology in an attempt to provide direction for future work

that will address the issues raised in Section 1.2.4. The review proposed to collect, collate and interpret information on water cover design and implementation and research projects in Canada and elsewhere. Specific objectives were as follows:

- (i) List and summarize Canadian and International mine sites using water covers for managing acid rock drainage.
- (ii) List, review, interpret and report ongoing MEND-sponsored water cover research at the Noranda Technology Centre, Pointe-Claire, Québec, and the CANMET Elliot Lake Laboratory (MEND Project 2.15.1).
- (iii) Review and document water cover design and implementation work carried out by Boliden Minerals and other mining companies in Sweden.
- (iv) Review, document and report status on water cover research conducted by the Norwegian Water Research Institute.
- (iv) List, review and document water cover research and implementation performed in the United States including work conducted by the U.S. Bureau of Mines.
- (vi) Identify any gaps in knowledge in the application of water cover technology in Canada.

The review presented here focuses on shallow water cover studies and applications, and does not deal with underwater tailings disposal (for example, in lakes).

1.4. ORGANIZATION OF THE REPORT

Chapters 2 and 3 describe methods presently used to calculate the necessary cover depth and to predict future water quality in water covers. Chapter 4 summarizes the history, design, and performance of several water cover sites, and also presents the results of field investigations. Chapter 5 describes ongoing laboratory and field research and model

development. Chapter 6 compares the performance of the reviewed water cover sites, and discusses the results and conclusions of the reviewed research efforts. Chapter 7 offers general conclusions from the review and identifies specific uncertainties regarding water cover performance.

Sections 3.3, 4.6, 5.1 (Noranda Technology Centre research), 5.2 (CANMET Elliot Lake Laboratory research), 5.3 and 5.5 of this report specifically address the above-mentioned objectives.

2. METHODS FOR PREDICTING THE REQUIRED DEPTH OF WATER

The selection of the appropriate depth of water cover over reactive sulphide tailings is probably the most important step in implementing an effective water cover technology. The importance lies in the fact that the depth of water has implications in the total volume of water available for tailings storage and the height of the dams and dikes, and hence the cost of decommissioning. Two major criteria are used in the selection of the water cover depth: (i) hydrological forecast or the probability of occurrence of a drought event, and (ii) the minimum depth required to prevent tailings resuspension.

2.1. HYDROLOGICAL FORECASTING

One criterion for the selection of water depth is to maintain moisture saturation of tailings even in the event of a drought. Drought duration is determined by statistically extrapolating from recorded drought events and using a certain probability distribution, such as the Gumbel method. The drought is determined for a certain return period, such as 500 years. The absence of records for realistic periods limits the accuracy of these analyses. Records from neighbouring stations are generally used.

To calculate the necessary depth of water, a hydrological model for the tailings impoundment must be developed. This requires knowledge of the hydraulic conductivity of the tailings and the underlying soil deposits, which may vary significantly. The surface and subsurface hydrology must also be characterized.

2.2. PREDICTING RESUSPENSION

Another criterion for the depth of water is to limit resuspension of the tailings particles. While the net effect of tailings resuspension on ARD generation is unknown, transporting tailings out of the impoundment may eventually lead to the degradation of water quality downstream from a site.

Linear wave theory, which assumes that the height of waves is small compared to the wave length and water depth, is usually adopted in estimating particle resuspension. This assumption linearizes two boundary conditions describing wave motion: the dynamic surface boundary condition which describes the vertical component of water velocity at the surface, and the kinematic boundary condition, which is Bernoulli's equation for unsteady flow at the surface. With these simplifications, water velocities can then be explicitly solved. Of particular importance to resuspension is the horizontal water particle velocity:

$$(2-1) \quad u = \frac{\pi H}{L} \left[\frac{\cosh k(d+z)}{\sinh kd} \right] \cos(kx - \sigma t)$$

where

k is the wave number, defined as $2\pi/L$

σ is the wave angular frequency, defined as $2\pi/T$

z and x are spatial coordinates (z is negative below the water surface)

d is the water depth to the bottom

H is the wave height, which must be obtained from empirical wind-wave relationships

L is the wavelength, and can be solved iteratively using the equation

$$(2-2) \quad L = \frac{gT^2}{2\pi} \tanh \frac{2\pi d}{L}$$

where T is the wave period, which is also obtained from empirical wind-wave relationships

One can further assume that waves are either deep water or shallow water waves. If the depth is greater than $L/2$, the maximum horizontal water velocity is independent of water depth, and may be approximated by:

$$(2-3) \quad u = \frac{\pi H}{T} \exp\left(\frac{2\pi}{L} z\right)$$

If the depth of water is very shallow, as may be the case for engineered water covers over tailings, the velocity equation is

$$(2-4) \quad u = \frac{\pi H}{T} \left(\frac{L}{2\pi d} \right)$$

Note that despite the shallow depth of tailings ponds, the deep water wave condition may still be applicable at some sites if wave heights are small.

2.2.1. Norwegian Hydrotechnical Laboratory (NHL) Approach

Researchers at NHL in Trondheim, Norway use empirical methods to calculate the necessary depth of water to eliminate resuspension, based on two different criteria. For particle sizes greater than 100 μm , a maximum shear force at which particles will resuspend, based on work by Shields (1936), is calculated. For particle sizes less than 100 μm , a maximum shear velocity is determined. Estimates of resuspension consider wind events with a 10-year return period. Wave heights and wave periods are calculated for various fetch lengths. A non-dimensional fetch is calculated using the following relationships:

$$(2-5) \quad \bar{F} = \frac{gF}{u_*^2}$$

where

F is the fetch,

g is the acceleration due to gravity,

and u_* is the air shear velocity, which is calculated as follows

$$(2-6) \quad \begin{aligned} u_*^2 &= C_d U_{10}^2 \\ C_d &= (0.8 + 0.065U_{10})10^{-3} \end{aligned}$$

where

U_{10} is the wind speed at a 10 m height above the water surface, and C_d , the drag coefficient at the air-water interface, is calculated as shown above.

The following non-dimensional parameters are then defined:

$$(2-7) \quad \bar{H}_{mo} = \frac{gH_{mo}}{u_*^2} \quad \text{where } H_{mo} \text{ is the significant wave height}$$

$$(2-8) \quad \bar{T}_u = \frac{gT_u}{u_*} \quad \text{where } T_u \text{ is the upper wave period}$$

$$(2-9) \quad \bar{T}_d = \frac{gT_d}{u_*} \quad \text{where } T_d \text{ is the duration of the wind event}$$

These non-dimensional parameters have been correlated with fetch length using data from the North Sea:

$$(2-10) \quad \bar{H}_{mo} = K_1 \sqrt{F} \quad \text{where } K_1 = 0.0506$$

$$(2-11) \quad \bar{T}_u = K_2 \sqrt[3]{F} \quad \text{where } K_2 = 0.903$$

$$(2-12) \quad \bar{T}_d = K_3 \bar{F}^{2/3} \quad \text{where } K_3 = 23$$

From these parameters the real significant wave heights and periods can be back-calculated. The uncertainty in the above expressions is assumed to be around 10 %.

The bed velocity, U_b , can be calculated from the above wave characteristics using linear wave theory. The shear force at the bottom, τ_b , is then calculated as:

$$(2-13) \quad \tau_b = 0.5 f_w \sigma_w U_b^2$$

where

σ_w is the specific mass of water, kg/m³

U_b is the maximum bottom water velocity calculated from linear wave theory, m/s,

and f_w is a dimensionless shear factor and can be calculated iteratively using the expression:

$$(2-14) \quad \frac{1}{4\sqrt{f_w}} + \log \frac{1}{4\sqrt{f_w}} = -0.08 + \log \frac{a_w}{2.5 d_{50}}$$

where

a_w is the water particle amplitude at the bottom calculated from linear wave theory,

d_{50} is the average tailings particle diameter

As noted earlier, two different suspension criteria are used by NHL:

- 1) Critical shear force
- 2) Critical shear velocity

The shear force criterion is derived from Shield's work, which estimated critical shear force for a given particle experimentally. Plotting shear force versus sediment discharge, and then extrapolating back to zero sediment discharge found the critical shear force.

Based on these experiments, the following empirical equation was derived:

$$(2-15) \quad \tau_{sh} = \tau \sigma_w g d_{50} \left(\frac{\gamma_s}{\gamma_w} - 1 \right)$$

where

τ_{sh} is the critical shear force

τ is a dimensionless shear force obtained from Shield's experiments,

g is the acceleration due to gravity,

and γ_s and γ_w are the specific weights of particles and water.

Then if $\tau_b > \tau_{sh}$ resuspension occurs.

However, for particles sizes smaller than 100 μm , Shield's criterion is unreliable because the background data are sparse, and the cohesive forces between small particles is not taken into account. For particle sizes less than 100 μm a critical shear velocity criterion is used. U_* the shear velocity, is calculated as:

$$(2-16) \quad U_* = \sqrt{\frac{\tau_b}{\rho_s}}$$

where ρ_s is the tailings particle density

Then an empirical criterion is used to assess if suspension occurs:

$$(2-17) \quad U_* > 1.7w$$

where w is the settling velocity.

The settling velocity of a single, spherical, and cohesionless particle is assumed to represent the average settling velocity of suspended tailings of a certain grain size. Settling velocity is obtained from the results of Rouse (1937b) who performed experiments with spherical quartz particles of density of 2.65 g/cm^3 (Fig. 2-1). The settling velocity is also used to calculate the time particles of a certain size remain in suspension after wave activity. For a particle to reach the water surface, U_* must be 20 times the settling velocity.

At this shear velocity, particle concentration at the middle water depth will be about 60 % of that at the bottom, and particle concentration at the surface will be about 40% of that at the bottom. To get a homogeneous suspension over the whole water depth, the shear velocity must be 200 times the settling velocity.

2.2.2. UBC Approach (Lawrence et al., 1991)

Lawrence et al. (1991) used linear wave theory and wave hindcasting procedures to determine the depth of water cover needed to prevent the resuspension of Syncrude oil sands tailings in Northeastern Alberta. Ward et al. (1994) developed a simple model to predict the amount of wind-entrained tailings and found good agreement between the model results and measured data. As with the NHL method, the design wind speed has a 10-year wind speed of 1-hour duration. Wave characteristics were calculated from wind speed and fetch using empirical relationships derived by the U.S. Coastal Engineering Research Center similar to Equations (9), (10) and (11) above, except that the coefficients K_1 , K_2 , and K_3 were 0.0016, 0.2714, and 68.8 respectively.

Assuming that the bottom wave velocities can be calculated using linear wave theory, and that the waves causing suspension are deep water waves then

$$(2-18) \quad U_b = \frac{2\pi H}{T \exp(2\pi d / L)}$$

where

H and T are the wave height and period,

d is the pond depth,

and $L = gT^2/2\pi$, the deep water wavelength.

Note that $d > L/2$ is assumed in the derivation of Equation (2-18).

Resuspension will occur when U_b exceeds some threshold velocity U_t . Lawrence et al

(1991) noted that because of the thixotropic nature of tailings there will be a significant difference between the behaviour of tailings and non-cohesive particles. Therefore, rather than using empirical expressions derived from experiments on non-cohesive particles, they performed experiments on Syncrude oil sands to obtain U_t . The experiments were conducted in a wave tank, where the generated wave height was varied while the wave period was kept constant until resuspension occurred. U_t was then back-calculated from the wave speed and wave period.

2.2.3. Sverdup-Munk-Bretschneider Method

As with the above methods, this approach also relies on empirically derived expressions relating maximum wind speed, fetch length, and wind duration to wave characteristics (see Fig. 2-2), but assumes waves are shallow. The method estimates the maximum wave height. Some investigations of resuspension have used a wave height to water depth ratio as a criterion for resuspension (for an example, see Section 5.4) The maximum wave height is calculated as follows:

$$(2-19) \quad H_{\max} = (0.5 \ln NIW)^{0.5} H_s$$

where H_s is the significant wave height obtained from Fig. 2-2, and NIW is the number of incident waves:

$$(2-20) \quad NIW = \frac{T_d}{T_p}$$

where T_d is the duration of the wind event, and T_p is the peak energy period, both obtained empirically using Fig. 2-2. To verify whether shallow wave conditions exist, the wavelength, $L = gT^2/2\pi$, must be less than twice the depth of water.

2.2.4. Rodney and Stefan (1987) Method

The method of Rodney and Stefan (1987) estimates the shear force at the bed in shallow ponds, which is employed in two empirical equations to estimate the rate of resuspension. The rate of resuspension and the settling velocity of the particles considered are used to

calculate an equilibrium concentration of suspended solids in the pond. The bed shear stress is calculated considering three different wind-induced phenomena: return currents, progressive waves, and standing waves. The shear stress on the bed from a return current for non-stratified turbulent flow is approximately 10% of the shear stress generated on the surface (Baines and Knapp, 1965), and is given by:

$$(2-21) \quad \tau_{w1} = 0.1C_d \rho_a U_{10}^2$$

where

U_{10} is the wind speed,

C_d is the drag coefficient and $C_d = 0.0005 U_{10}^{1/2}$ for $U_{10} < 15$ m/sec

and $C_d = 0.0026$ for $U_{10} > 15$ m/sec

ρ_a is the density of air (kg/m³)

The shear force from progressive waves can be calculated from the bed velocity, U_b :

$$(2-22) \quad \tau_{w2} = \rho f U_b^2$$

where ρ is the density of water and f is the wave friction factor, for which a value of 0.004 for flow in lakes was used. The bed velocity is calculated from linear wave theory.

Wave height (H) and period (T) are calculated using the following empirical equations:

$$(2-23) \quad H = \left(\frac{U_{10}^2}{g} \right) 0.283 \left[\tanh 0.53 \left(\frac{gd}{U_{10}^2} \right) \right]^{0.75} \tanh \left\{ \frac{0.0125 \left(\frac{gF}{U_{10}^2} \right)^{0.42}}{\tanh \left[0.833 \left(\frac{gd}{U_{10}^2} \right) \right]^{0.75}} \right\}$$

$$(2-24) \quad T = \left(\frac{2\pi U_{10}}{g} \right) 1.2 \tanh \left[0.833 \frac{gF}{U_{10}} \right]^{.375} \tanh \left\{ \frac{0.077 \left(\frac{gF}{U_{10}^2} \right)^{0.25}}{\tanh \left[0.833 \left(\frac{gd}{U_{10}^2} \right) \right]^{0.375}} \right\}$$

where F, g, and d are fetch length, acceleration due to gravity, and pond depth respectively.

The stress caused by standing waves is periodic over a large time scale. The wind setup, or the difference in water depth from the downwind to the upwind end of a body of water caused by wind action, is calculated using the following empirical expression:

$$(2-25) \quad S = \frac{3.37 \times 10^{-7} U_{10}^2 F}{d_m}$$

where d_m is the average lake depth (m)

The maximum rise above the mean water depth, d_r , is given by

$$(2-26) \quad d_r = 0.57S$$

and the maximum flow velocity U_m by

$$(2-27) \quad U_m = d_r \sqrt{g / d_m}$$

The flow velocity at any time t and at any distance x from the centre of the pond for a given fetch F is given by

$$(2-28) \quad U = U_m \cos \frac{\pi x}{F} \sin \frac{2\pi t}{T}$$

Then the shear due to standing waves is obtained by substituting U into Equation (2-21) above.

The method predicts an equilibrium concentration of suspended solids by finding the concentration when the settling rate of the particles is equal to the rate of resuspension. The settling velocity of the particles is based on the settling rate of a single particle. The rate of resuspension is evaluated using two empirical equations, one linear (Kendrick, 1972) and the other non-linear (Akiyama and Fukushima, 1985). These equations were formulated using data from experiments on river sediments, not tailings. The two equations predict widely different rates of resuspension, resulting in differences of up to two orders of magnitude between estimates of the equilibrium concentration of suspended solids. However, the predictions of threshold wind speed for resuspension do not vary significantly. Figure 2-3 shows the threshold wind speed as a function of grain size for two different water depths, 1 m and 20 m. The prediction indicates little advantage is gained in avoiding resuspension by increasing the depth of the pond beyond 1 m.

2.2.5. Summary

In the methods for predicting resuspension reviewed above, four criteria for tailings resuspension were found:

- (i) Critical bed shear stress
- (ii) Critical shear velocity, which is compared to the settling velocity of a single spherical particle
- (iii) Critical bed velocity, and
- (iv) Maximum wave height to water depth ratio

All these criteria rely on equations derived from experiments on non-cohesive particles or sediments from a particular water body. These equations may be inadequate for predicting tailings resuspension, since tailings are cohesive and

thixotropic. Raudkivi (1990) found that if only 10% of sediment was of clay size, it would behave as clay. Pending verification of the above models, it may be necessary to develop a more fundamental and physically based model for predicting tailings resuspension.

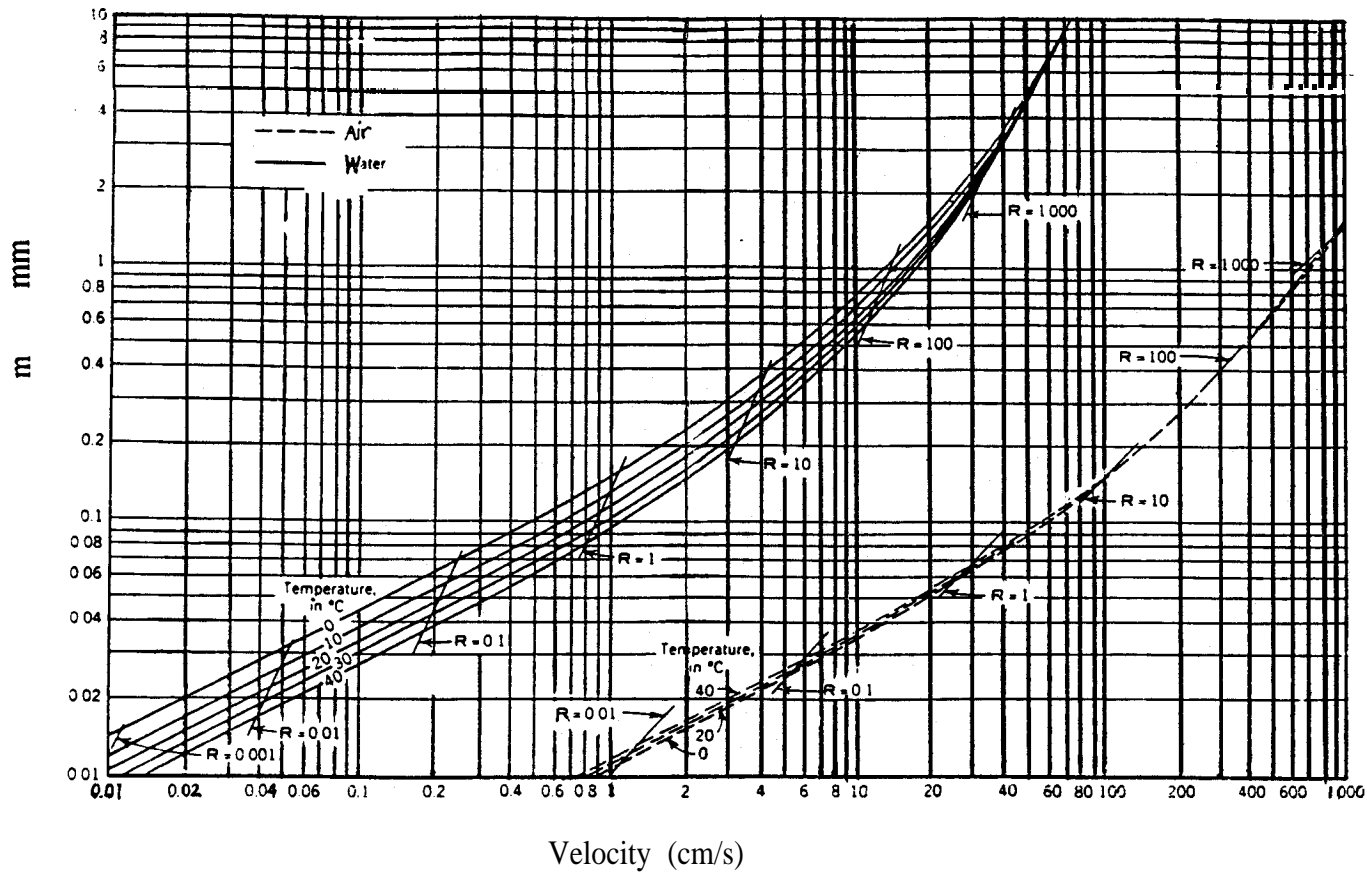


Figure 2-1

Fall velocity of quartz spheres in air and water (after Rouse, 1937b)

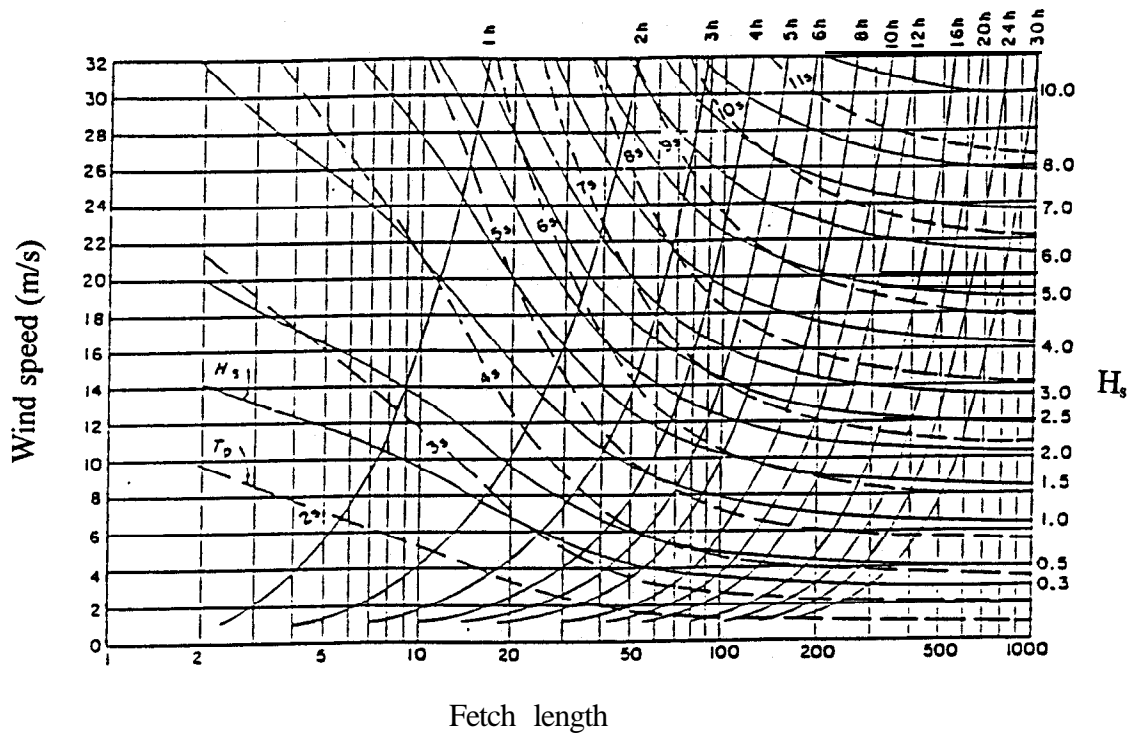


Figure 2-2

Chart of empirical correlations between wind and wave characteristics for shallow water waves (after U.S. Army Corps of Engineers Shore Protection Manual)

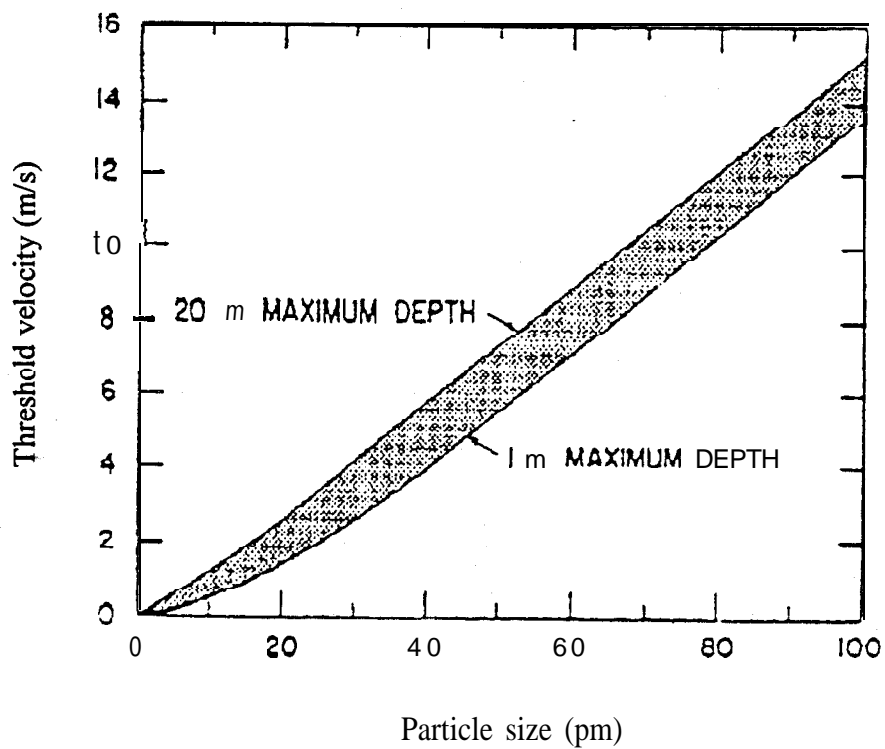


Figure 2-3

Approximate threshold wind speed for resuspension versus particle size for a particle density of 2600 kg/m^3 (after Rodney and Stefan, 1987)

3. METHODS FOR PREDICTING OXYGEN FLUX AND METAL RELEASE RATES

In the design of tailings ponds to date, the prediction of water quality has generally been limited to modelling metal release as a function of pyrite oxidation. These models all consider diffusion as the mechanism whereby oxygen is supplied to the tailings and the water cover is generally assumed to be saturated with oxygen, regardless of depth. Other geochemical considerations such as the kinetics of dissolution of various neutralizing or buffering minerals and secondary mineral formation and dissolution are typically ignored. The following models are employed in water quality predictions in the water cover cases reviewed in Chapter 4.

3.1. THE SIMPLE MOLECULAR DIFFUSION APPROACH

In the approach proposed by Morin (1993), the rate constant for the oxidation of pyrite was estimated by fitting experimental data, assuming that tailings particles are spherical, and that the rate is first order with respect to oxygen and particle surface area:

$$(3-1) \quad k = \frac{(3.4)(10^{-10})(1-n)(\%PYR)}{d_p}$$

where

d_p is the particle diameter,

n is the porosity of the tailings deposit, and

%PYR is the percentage of pyrite in the tailings.

Assuming that the concentration of oxygen in the water cover is homogeneous, and the transport of oxygen through the tailings is governed by diffusion, then the concentration of dissolved oxygen in mg/L at a depth z below the tailings surface is:

$$(3-2) \quad DO(z) = DO_{(pond)} \exp\left[-\sqrt{kD_e} z\right]$$

In equation (3-2) the effective diffusion coefficient of oxygen, D_e , is given by

$$(3-3) \quad D_e = Dn/T_r$$

where

D is the diffusion coefficient (m^2/s) of oxygen in water,

n is the porosity,

and T_r is the tortuosity.

The tortuosity describes the deviation of the actual diffusion path length (P_a) due to the presence of the particles, from the straight line path (P_0), and is defined as

$$(3-4) \quad T_r = \frac{P_a}{P_0}$$

The oxygen flux ($mg/m^2/s$) into the tailings is then:

$$(3-5) \quad J = DO_{(pond)} \sqrt{kD_e}$$

The rate of sulphate and metal release can then be predicted assuming they are released in stoichiometric proportion to the flux of oxygen into the tailings.

3.2. SHARER ET AL (1991)

The derivation of the acid flux in the method used by Scharer et al (1991) assumes that the oxidizing zone in the tailings will stop at a certain thickness when a steady-state between oxygen availability and pyrite consumption is reached. As in the first method, this assumption ignores the depletion of pyrite and the subsequent advancement of the

oxidation zone as the oxygen concentration increases in the pyrite-depleted tailings. Unlike the method described in Section 3.1, the consumption of pyrite does not vary with depth inside the oxidizing zone. The thickness of the oxidizing zone is defined by

$$(3-6) \quad \lambda = \sqrt{\frac{8}{7.5} C_{(o,t)} \frac{D_e}{\rho R_s}}$$

where

D_e is the effective diffusion coefficient of oxygen in the tailings, m^2/month ,

R_s is an assumed sulphate generation rate, moles $\text{SO}_4/\text{kg tailings}/\text{month}$,

ρ is the density of the tailings, kg/m^3 .

and $C_{(0,t)}$ is the concentration of oxygen at the tailings-water interface.

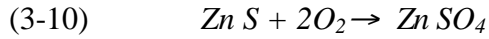
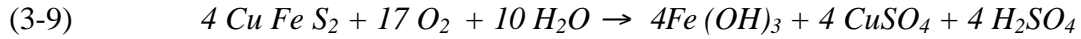
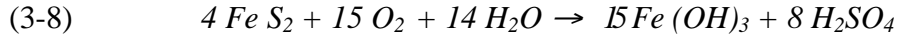
Then the flux is

$$(3-7) \quad J_s = \lambda R_s \rho$$

3.3. NIVA (NORWEGIAN INSTITUTE FOR WATER RESEARCH)

The Norwegian Institute for Water Research (NIVA in Norwegian) has developed a model for predicting metal fluxes in flooded tailings (NIVA, 1993). A key premise of the model is that transport of oxygen in the tailings is slow and therefore controls the release of metals from the tailings. Transport of oxygen and metals within the tailings is assumed to take place by diffusion. The model calculates the increase over time of sulphate, zinc, and copper fluxes from the tailings surface. The fluxes are then used to calculate the concentration of these species in the tailings pond effluent.

The oxidation of sulphide minerals is assumed to occur according to the following reactions:



Transport of oxygen from the air to the water cover may be expressed as:

$$(3-11) \quad q_l = (AK_l + Q_v) (C_m - C_o)$$

where

q_l is the mass of oxygen supplied to the water cover from the air and from surface flow,

K_l is the transfer coefficient of oxygen between water and air,

Q_v is surface water inflow,

A is the surface area of the pond,

C_m is the saturated concentration of oxygen in water, and

C_o is the actual concentration of oxygen at the surface of the pond

If no significant temperature difference between the top and bottom of the water cover exists, turbulent mixing will transport oxygen to the top of the tailings, and the flux of oxygen from the water cover to the surface of the tailings can be written as:

$$(3-12) \quad q_2 = K \frac{A}{d} (C_o - C_h)$$

where

d is the depth of the pond,

C_h is the oxygen concentration in the water just above the tailings,

and K represents the action of turbulent mixing and circulation in the tailings pond.

The transfer coefficient at the air-water interface, K_l , is expressed by Banks (1975) as:

$$(3-13) \quad K_l = 4.2 \times 10^6 (U)^{0.5} ; U < 5.5 \text{ m/s}$$
$$K_l = 0.32 \times 10^6 U^2 ; U > 5.5 \text{ m/s}$$

The model assumes that oxygen will not penetrate ice. During winter, water inflow is therefore assumed to be the only source of oxygen. Under this condition, K_l is set to zero.

The vertical mixing coefficient K for the water phase over the sediment varies with wind conditions. The model assumes that vertical diffusion is so large that oxygen transport in the water cover will not limit the release rate of metals, so $C_0=C_h$. This is a conservative assumption as it provides an upper limit for available oxygen. D_s is assumed to be the same as the diffusion coefficient of oxygen in water.

Assuming that oxygen consumption within the tailings occurs in a very thin layer, then the oxygen transport into the sediment can be expressed as:

$$(3-14) \quad q_3 = \frac{AD_s}{\delta}(C_h - C_s)$$

where

C_s is the oxygen concentration in the layer where reactions are consuming oxygen, δ is the depth to the layer of sediment where the oxygen consumption is occurring, and D_s is the diffusion coefficient of oxygen in the tailings.

If δ and C_s are allowed to vary across the surface area the flux becomes

$$(3-15) \quad q_3 = \int \frac{D_s}{\delta} (C_h - C_s) dA$$

Since δ may be very difficult to define as it changes with time and degree of oxidation, the evaluation of the integral could be difficult.

The transport model is coupled to expressions for oxygen consumption by sulphide oxidation as follows:

$$(3-16) \quad q_{SO_4} = \frac{AD_s}{\delta_{SO_4}} [O_2] f_{SO_4}$$

$$(3-17) \quad q_{Cu} = \frac{AD_s}{\delta_{Cu}} [O_2] f_{Cu}$$

$$(3-18) \quad q_{Zn} = \frac{AD_s}{\delta_{Zn}} [O_2] f_{Zn}$$

$$(3-19) \quad q_{Total} = q_{SO_4} + q_{Cu} + q_{Zn}$$

where f is the fraction of oxygen consumed in each of the three above reactions.

f can be determined experimentally using flask experiments.

Note that if the value of δ is small, the calculated oxygen consumption will be unrealistically large. To address this problem, a maximum rate of oxygen consumption for each sulphide mineral, q_{max} is determined from column experiments, and the calculated consumption of oxygen, q_{calc} , is modified as follows:

$$(3-20) \quad q = \frac{q_{calc} q_{max}}{\sqrt{q_{calc}^2 + q_{max}^2}}$$

The following differential equations for oxygen transport and metal release may be written:

$$(3-21) \quad V \frac{\partial [SO_4]}{\partial t} = r_{SO_4} q_{SO_4} + s_{Cu} q_{Cu} + s_{Zn} q_{Zn} - Q_v [SO_4]$$

$$(3-22) \quad V \frac{\partial [Cu]}{\partial t} = r_{Cu} q_{Cu} - Q_v [Cu]$$

$$(3-23) \quad V \frac{\partial [Zn]}{\partial t} = r_{Zn} q_{Zn} - Q_v [Zn]$$

where

V is the volume of water in the water cover,

Q_v is the effluent flow rate,

r states how many grams of each component are produced per gram of oxygen consumed, and

s_{Zn} and s_{Cu} give how much H_2SO_4 is produced in the copper and zinc reactions.

The increase in depth to the oxidation front, δ , for each sulphide mineral, can be described as follows:

$$(3-24) \quad A \frac{\partial \delta_{SO_4}}{\partial t} = \frac{r_{SO_4}}{w_{SO_4}} q_{SO_4}$$

$$(3-25) \quad A \frac{\partial \delta_{Cu}}{\partial t} = \frac{r_{Cu}}{w_{Cu}} q_{SO_4}$$

$$(3-26) \quad A \frac{\partial \delta_{Zn}}{\partial t} = \frac{r_{Zn}}{w_{Zn}} q_{SO_4}$$

where w is the composition of the respective component in the tailings as a weight per unit volume.

If one assumes that $[O_2]$ is constant with time, then the depth of the oxidation front can be expressed as a function of time, for example:

$$(3-27) \quad \delta_{SO_4}(t) = \sqrt{2 \frac{r}{w} D_s [O_2] f_{SO_4} t}$$

3.4. COLUMN EXPERIMENTS

These experiments are used to simulate metal release from flooded tailings. The experiment involves semi-continuous replacement of the cover water above the tailings with new water of known chemical composition. The columns are covered with a lid during the test, but the lid has holes for the inflow of air. The columns are placed in a water bath with a constant temperature of 6°C, and a capillary tube supplies airflow. The air current maintains oxygen concentration in the water phase.

The effluent water samples are analyzed for metal concentrations. The concentration at the start of each time interval, C_i , can be calculated using the following expression:

$$(3-28) \quad C_i = ((V - v) C_s + v C_o) / V$$

where

V = volume of water over sediment.

v = removed volume of water analyzed

C_s = end concentration in preceding time interval

C_o = concentration in water as added

The flux, in mass per area per day, may be written

$$(3-29) \quad Flux = (C_s - C_i) V / (A D)$$

where

A is the area of the sediment in contact with water, and

D is the length of the experiment in days.

These calculated flux values are used to estimate the maximum release rate of metals to be used at the start of modelling.

Flask Experiment

This experiment mixes water and tailings in a flask that is shaken continuously to find a maximum release value of metals and sulphate from tailings. This ensures maximum contact between the water and solid phases. NIVA assumes the best conditions (worst case scenario) for release of metals and sulphate. The results from these experiments are used to calculate the distribution of oxygen consumed in the chemical reactions taking place.

4. REVIEW OF WATER COVER SITES

The following chapter summarizes the history, design, and performance of several water cover sites in Canada, Sweden and Norway. The results of field investigations conducted at each site are also presented.

4.1. EQUITY SILVER, BRITISH COLUMBIA, CANADA

4.1.1. Site Description and History

The Equity Silver Mine is located near Houston, B.C., and operated from 1980 to January 1994. From the start of operations, tailings containing arsenopyrite, pyrite, and pyrrhotite were deposited in a man-made pond. The tailings were treated in a sulphide-recovery plant. A cyanide-destruction plant has operated since 1985. The area of the pond is 120 ha, and the median water depth of the pond is 4.3 m. Only 8% of the pond has depths shallower than 1.5 m, the chosen closure depth. There are no surface inflows and outflows from the pond, although there is an overflow constructed to prevent dike overtopping. The hydrological inputs and outputs are precipitation, evapotranspiration, and a small seepage (HAY Consultants, 1996). Presently, measured sulphide content in the upper tailings ranges from 2 to 10%, iron from 4.8 to 11%, copper 0.11%, zinc 0.06%, and arsenic 0.5 to 5% (Rescan, 1996).

4.1.2. Water Cover Design

HAY Consultants (1996) predicted the water depth necessary to prevent resuspension of the Equity Silver tailings by relating depth to wave height and sediment characteristics. The highest wind speed observed at the station over a three-year study period when the pond was ice-free was 18.6 m/s. It was calculated that this wind speed has a return period of 8 years. This wind event was calculated to cause resuspension at depths shallower than 1.4 m. The determination of the necessary water depth was based on the work of Lawrence et al (1991) discussed in Section 2.2.2. The terminal velocity at the bottom of the water cover, required to resuspend tailings particles, was determined using the following empirical equation:

$$(4-1) \quad U_t = \left[0.21 \left(\frac{\rho_s - \rho}{\rho} \right) g \left(\frac{D_{50}}{10} \right)^2 T_{sig} \right]^{2/3}$$

where

ρ_s is the density of the tailings solids

ρ is the density of water,

D_{50} is the median grain size, and

T_{sig} is the significant wave period

The 1.5 m water cover was originally chosen because it was assumed that this would provide an adequate oxygen barrier. No oxygen transport modelling was performed. It was assumed that the water cover would be saturated with oxygen at all depths.

4.1.3. Site Investigations

Summaries of two different field studies conducted at the site are presented. One field study consisted of a bathymetric survey, collection of surface sediments and tailings cores, and observations of bedforms at different depths and locations (HAY Consultants, 1996). The surface sediments and cores were analyzed for moisture content, submerged weight density, and grain size distribution.

A bathymetric survey was conducted using a chart-recording depth sounder for depth and a Global Positioning System (GPS) for exact location. Certain water depth measurements were duplicated and no noticeable change in values occurred, assuming the depth measurements were accurate to ± 0.1 m. The accuracy of the GPS was assessed at ± 3 m. Thirteen surficial sediment grab samples and two sediment core samples were obtained from each of three locations, at depths of 1.3 m, 2.3 m, and 3.4 m respectively. The depths were chosen to evaluate the assumed water cover depth of 1.5 m, and to verify predictions of the water depth necessary to prevent resuspension. The locations were chosen along the longest fetch of the pond. Sediment was sampled by inserting a PVC pipe into the bed to depths between 0.3 m and 0.6 m, sealing the top to provide suction,

and then lifting. Once in the boat, the samples could be extruded. Nine of the surficial samples were collected within a 1 m radius of each location, while the remaining four were collected within 5 m and 10 m radii. Sediment cores were extracted using a 0.08 m diameter barrel. The cores were 1 m in length. The cores were extracted within 2 m of each designated location, but away from the sample grab areas. The top 0.30 m of each core was used for grain size and submerged weight analyses. Bedforms were observed during reconnaissance dives at the deepest and shallowest water depth locations. Bedforms were also observed around the pond perimeter from the surface. Meteorological data was obtained from a station located 1.5 km south of the tailings pond. The climate station collects average hourly wind speeds and daily maximums.

4.1.4. Results

Bathymetric Survey

The median water depth of the pond was 4.3 m. Only 8 % of the pond has depths shallower than 1.5 m, the required closure depth.

Surficial Grab Samples

The grab samples were analyzed for moisture content, grain size, and submerged weight density. The grain size distributions and sediment density were significantly different at each site. These differences were caused by the settling pattern of particles during discharge. During discharge the coarser particles remain near the discharge point while the fine particles travel further before settling. The shallow site contained the highest proportion of fine sands and silts, with a D_{50} (median grain size) of 0.0512 mm, while the next deepest site had a D_{50} of 0.0075 mm, and the deepest site had a D_{50} of 0.0040 mm. The variability of grain size was smallest at the deepest site.

The moisture content distribution at the shallow site was significantly different from distributions at the two deeper sites, but there was no significant difference between the moisture content distributions at the two deeper sites.

The average submerged weight density for all the sediment samples was $1.318 \pm 0.328 \text{ kg/m}^3$. Density decreased with water cover depth with the formation of a lutocline (a transitional layer between sediment and water where no clear interface exists).

Core Logs

The cores near the shallowest depth are characteristic of episodic discharge, with layers of unconsolidated tailings sitting on top of consolidated tailings, with small (0.01 m to 0.02 m) layers of orange flocculates at the surface and at historical surfaces. At the deeper locations, this oxidized layer is either thinner or does not exist. Cores from the deeper locations show no evidence of wave agitation.

Bedform Observation

At both 3.4 m and 1.3 m, disturbance by the diver's motion reduced visibility to near zero within 30 seconds. While bedforms were observed at 1.3 m, none were noticeable at 3.4 m. However, lutocline layers at deeper depths would not necessarily form bedforms even though wave action is present; particles may resettle uniformly and the shear stress required to form bedforms may be significantly greater than the stress required to suspend clay size particles (Raudkivi, 1990). Sediment resuspension in the tailings pond may be hindered by the surface orange flocculated layer, which provides some resistance to particle entrainment. The resistance to resuspension may be due to particle binding by microfauna secretions (Raudkivi, 1990, p339). When disturbed, the layer peeled back, exposing underlying tailings that readily resuspended. It was noted from observations around the pond perimeter during an earlier field study that ripples were absent from beds deeper than 1.4 m.

Wind Data and Resuspension Calculations

HAY Consultants (1996) calculated the depth of water cover necessary to prevent resuspension based on the work of Lawrence (1991) discussed in Section 2.2.2, which relate depth to wave height and sediment characteristics. The calculations concluded that

the design wind speed of 18.6 m/s would resuspend tailings that are shallower than 1.4 m, which is approximately the depth where bedforms were noted to have disappeared during diving.

The second field study was a geochemical assessment of the site (Rescan, 1996). The objectives of the study were to assess the water column chemistry, to compare dissolved ion concentrations in interstitial waters with the composition of the solid phase, and to determine metal fluxes from the sediment. The study involved examining tailings cores to determine solid composition, sampling interstitial waters using dialysis-membrane array-samplers (peepers), and sampling the water cover. Sampling was conducted at a shallow site (1.8 m) and a deep site (5.0 m). Chemical species analyzed included N, NO₃, PO₄, Cl, CN, Sb, As, Cd, Fe, Pb, Mn, Zn, organic carbon and dissolved oxygen.

The depositional history at the shallow sites is reflected in changes in the pore water chemistry. At the deep site, tailings accumulated in an uninterrupted fashion. Organic carbon at both sites is near or below the detection limit of 0.05 % wt.

4.1.5. Solids Composition

The percentage of iron and sulphide in the solid phase correlates well with each other, except below 34 cm of sediment at the shallow site. There the percentage of sulphide increases dramatically, while the amount of iron remains the same. At the shallow site, the concentrations of iron and sulphide increase with depth for the first 10 cm (6-10% wt for sulphide, 80000-110000 µg/g for Fe) and then gradually decrease with depth. At the deep site, the concentrations of sulphide and iron decrease with depth in the first 10 cm (4 -12% wt sulphide and 66000-48000 µg/g iron), then increase with depth from 10-20 cm (3 % wt. sulphide and 58000 µg/g iron), and then remains fairly constant. The differences in the distribution of iron and sulphide between the shallow and deep sites may be explained by grain size segregation (Davé, 1997). The overall greater percentage of sulphide and iron, observed in the shallow site, indicates that the coarser fraction of tailings contains a greater proportion of sulphide and iron. The coarser fraction has been known to settle in the shallower waters.

Zinc and copper solid phases at the shallow site are distributed uniformly at concentrations of 600 $\mu\text{g/g}$ and 1100 $\mu\text{g/g}$ respectively to a depth of 34 cm where the concentrations increase to 6200 $\mu\text{g/g}$ and 7300 $\mu\text{g/g}$, which is a reflection of tailings deposition history. At the deep site, concentrations of copper and zinc are uniform with depth at 600 $\mu\text{g/g}$.

The solid composition of arsenic is relatively high in the upper 34 cm of the tailings at the shallow site (5-15 mg/g at the shallow site, but less than 5 mg/g at the deep site). Both sites, however, show similar arsenic concentrations of approximately 2 mg/g below 34 cm. The concentration of antimony in the solid phase is similar (100-300 $\mu\text{g/g}$) at both sites.

4.1.6. Water Cover and Interstitial Water

Dissolved iron concentration peaks sharply at the water-tailings interface, which indicates iron reduction within the upper 2 cm of the tailings. Iron concentrations both above and below the interface is low probably because of rapid precipitation in the lower layers and iron oxidation and precipitation as iron hydroxide in the water column. Dissolved manganese decreases across the interface from 1.3 mg/L in the water column to less than 0.5 mg/L at both locations, but most rapidly at the shallow site.

The dissolved copper concentration in the water column and in the upper 34 cm of the tailings is very low, except in one of the peepers at the shallow site, where concentrations peak at 0.19 mg/L at the interface. The concentration of copper in the three peepers at the shallow site rises to 0.63 mg/L below 34 cm, which corresponds to the increase in the copper concentration in the solid tailings below that depth. Copper concentrations are negligible except in one peeper at the deep site where copper concentrations are elevated from 0.02 to 0.12 mg/L between 0.1 m and 0.3 m depths. Zinc concentrations are relatively uniform with depth (0.1 to 0.5 mg/L) at both sites.

Arsenic concentrations decrease rapidly across the interface into the water column. Antimony concentrations are higher in the water cover than in the pore water at both sites. Antimony is consumed within the tailings, resulting in a diffusive influx across the tailings-water interface. The source of antimony in the pond is unknown; concentrations are two orders of magnitude higher than typical values in lakes. The pH at both locations is alkaline.

The dissolved oxygen (DO) concentration at the bottom of the water cover at both sites is 5 mg/L. Oxygen concentration below the tailings-water interface follows the typical concave profile characteristic of combined diffusion and consumption. Oxygen fluxes were estimated from changes in concentration across the interface to be 30.7 mg O₂/m²/day and 14.0 mg O₂/m²/day at the shallow and deep sites, respectively. A diffusion coefficient of 15.7 x 10⁻⁵ cm²/s was used in flux calculations (Broeker and Peng, 1974).

The oxygen consumption rate of the tailings was an order of magnitude less than oxygen uptake in meso-eutrophic lakes. Oxidation rates in tailings previously measured by David and Nicholson (1995) ranged from 2.6 to 8.7 mg O₂/m²/day . These rates were measured in pure sulphide tailings without organic carbon. The amount of O₂ consumed by organic carbon was estimated by assuming it was restricted to the tailings-water interface: i.e., the oxygen concentration gradient across the interface was used to calculate oxygen consumption by organic matter. The oxygen flux from pyrite oxidation was obtained by subtracting the flux due to organic consumption from the flux estimated by fitting the O₂ profile within the tailings. The rate of pyrite oxidation was calculated to be 5 mg/m²/day. If all the sulphate generated through sulphide oxidation were to migrate to the water cover, the equivalent oxygen flux would be 6.7 mg/m²/day. The difference may be attributed to oxygen consumption by cyanide oxidation (Davé, 1997).

Sampling of the tailings supernatant shows a declining trend in cyanide, copper, zinc and antimony concentrations. The pH has remained steady at about 7.0.

In conclusion, concentrations of those species diffusing from the tailings into the water cover are low. The dissolved concentration of copper varies from peeper to peeper, perhaps reflecting small-scale heterogeneity of a certain copper-bearing solid in the tailings. The method used to determine the necessary depth of water to avoid resuspension is supported by observations of the bedform, but is not conclusively verified. The formation of an orange oxidized layer at the surface of the shallower tailings seems to serve as a barrier to oxygen penetration into the tailings. This layer has not formed at the deepest observed site (2.3 m).

4.1.7. Performance to Date

The present concentrations of iron, manganese, copper, zinc, antimony, arsenic, and cyanide in the water cover are all low (below detection, < 0.5, <0.02, <0.5, <0.04, negligible, and < 0.06 mg/L respectively). Copper concentrations have decreased from 25 mg/L when tailings deposition was still active.

The pH in the water cover ranges between 7 and 8. Bedform observation indicates tailings resuspension at depths less than 1.4 m.

4.2. CELL 14, QUIRKE LAKE TAILINGS TEST SITE, ELLIOT LAKE, ONTARIO

4.2.1. Site Description and History

The Quirke Lake tailings were produced from uranium mining and have a pyrite content of 5.3 %. Tailings deposited in the tailings pond are predominantly sand and silt sizes (see Fig. 4-1). Acid leaching in the milling process removed any natural alkalinity, such as calcite, present in the tailings.

The selection of a tailings pond design for the reclamation of these tailings is outlined in the report entitled “Environmental Impact Statement for the Decommissioning of the Quirke and Panel Waste Management Area”, submitted in August 1993. The pond was designed as a series of five terraced cells (Fig. 4-2), to accommodate the stratified

deposition of the tailings and therefore minimize tailings relocation. An adjacent lake (Gravel Pit Lake) supplies water to the site.

Final flooding of all cells was completed by the fall of 1995. All the cells, except Cell 14 flooded in 1992, are currently being stabilized with lime against acidity generated from previously oxidized tailings. Cell 14 required only a month of stabilization. In the summer of 1995, it was inhabited by a family of ducks. The minimum water cover depth is 0.6 m.

4.2.2. Water Cover Design

The depth of the water cover was selected to:

- 1) ensure saturation of the tailings in the event of a drought; and
- 2) facilitate the growth of vegetation, such as cattails

Organic material was placed at the cell edges to encourage the growth of vegetation and to inhibit erosion.

Nowhere in the Environmental Impact Statement, or in supporting documents, are the effects of wave action on the resuspension of tailings discussed. Resuspension was not considered a potential problem because of the coarse nature of the tailings and the small wave fetch at the site (personal communication, Al Vivyurka, 1996) .

4.2.3. Predictive Modelling of Sulphide Oxidation

Predictive geochemical modelling for environmental impact assessment was performed using UTAP3, a computer program developed through NUTP (National Uranium Tailings Program) by SENES Consultants. UTAP is a sophisticated probabilistic fate model for predicting contaminant concentrations.

The modelling of pyrite oxidation by the program is based on the work of Scharer et al. (1991) and Jaynes et al. (1984). The UTAP model estimates a first order reaction rate k , based on expressions similar to Equations 3.6 and 3.7 in Section 3.2:

$$(4-2) \quad k = J_s / (2 L_w [Fe S_2]^0)$$

where L_w is the depth of the oxidizing zone and $[Fe S_2]^0$ is the initial pyrite concentration.

The model assumes that under a water cover, the oxidizing zone will have a maximum thickness of 10 cm, as it is believed that oxidation would initially proceed very slowly due to the low rate of oxygen diffusion in water, and would eventually cease when a sustainable organic cover formed.

This is not substantiated in any document pertaining to the environmental review; although investigations of other sites, including the Panel wetland (see below) give credence to this statement. No methodology is, however, given to estimate the maximum depth of the oxidation zone or to determine the conditions under which oxidation will stop.

From the above, the sulphate concentration at the bottom of the oxidation front is calculated:

$$(4-3) \quad [SO_4] = 2 L_w [Fe S_2] [1 - e^{-k t}] / (v_{infil} \cdot t)$$

where $[Fe S_2]$ is the current concentration of pyrite in the oxidizing zone and v_{infil} is the velocity of water infiltrating from the water cover into the tailings. The equation implicitly assumes that pyrite is consumed at the same rate everywhere in the oxidizing zone.

Further details of this model are discussed in Appendix B of " Supporting Document II to the EIS for the Quirke and Panel waste management areas " and the UTAP manual. The model considers precipitation and dissolution of sulphate minerals and iron, and includes a method for estimating the neutralizing effects of calcium carbonate. The depletion of

each sulphide mineral is modelled individually by assigning each sulphide mineral a depletion front. The positions of these fronts are used to estimate the flux of sulphate from the oxidation zone to the above pond water.

4.2.4. Performance to Date

1991-93 Pore Water Quality

The pore water quality in Cell 14 at Borehole 4,6 (see Fig. 4-3) improved significantly in 1991 and 1992, but seems to have stabilized in 1993 around a pH of 4, while maintaining a pH of 8 at lower depths. The pore water in the adjacent cell has a pH of 6.5 in the upper tailings, and 9.0 in the lower tailings. The decline in pore water quality in Cell 15 may be due to the dissolution of past oxidation products.

1993 -95 Pond Water Quality

Pond water quality was reported from January 1993 to December 1995. The pH and sulphate concentrations are plotted in Fig. 4-4. With the exception of two points, the pH varies between 6.0 and 7.5, with low values measured in the spring and high values measured in the summer. The sulphate concentrations show a slightly decreasing trend (Fig. 4-5) and are all well below the predicted maximum of 350 mg/L. The concentrations of iron and zinc are shown in Fig. 4-6. In the third year, iron and zinc concentrations seemed to have stabilized below 0.2 mg/L and 0.01 mg/L, respectively.

Assuming an average flow through the cell of 40 L/s (50 in summer and 30 reported in winter), and with an area of 66 ha, the average sulphate flux from the tailings in 1995 was:

$$(4-4) \quad J_s = CQ/A = (31 \text{ mg/L})(40 \text{ L/s}) / (66 \text{ ha}) = 19 \text{ mg } SO_4^{2-} / \text{ha} / \text{s} \text{ or } 0.0019 \text{ mg/m}^2 / \text{s}$$

4.3. PANEL WETLAND, ELLIOT LAKE, ONTARIO

4.3.1. Site History and Description

The Panel wetland basin is 14.5 ha in size, with tailings covering 12.9 ha, about 1.9 ha of which is above water. The average depth of tailings is 0.92 m. The eastern part of the basin is covered with 0.4 m to 1.4 m of water, while in the west the land slopes up and the tailings are either exposed or covered with a very shallow (< 0.5 m) water cover. The tailings were deposited in the late 1950's. The drainage area surrounding the wetland provides enough water for a six-fold dilution of water quality. Surface water flow is very variable, and was measured at 16.5 L/s in the fall of 1991. The Panel tailings are similar in composition and grain size to the Quirke tailings (MEND Report 3.12.2, 1993).

4.3.2. Site Investigations

Site investigation by CANMET revealed an oxidizing zone under shallow water cover at the tailings-water interface. This results from the transportation of oxygen by plants to the root zone. The oxidation processes seemed to be in equilibrium with reducing processes, i.e., the oxidation of sulphide minerals and metallic ions such as ferrous ion, and nitrification are in equilibrium with denitrification, the reduction of metals, and the reduction of sulphate to sulphide by sulphate-reducing bacteria. These sulphides then recombine with metals to reprecipitate minerals, such as pyrite.

Oxidized tailings are visible in areas exposed or covered by very shallow water (< 0.3 m) which may be explained by the natural fluctuation of the water level. The zone of oxidation is approximately 5 to 10 cm thick, and is yellowish green in colour above the water surface, and brownish orange below. The yellowish green colour is characteristic of jarosite precipitates, while the brownish orange colour is characteristic of ferric hydroxides. Cattails are the dominant species in the zones of shallow water. In the exposed tailings, vegetation appears to be promoting the oxidation of tailings in the root zone, instead of producing a reducing environment.

In areas where the water cover exceeds 0.3 m, the pH ranges from 6.5 to 9.8 during the summer. This rise in pH is possibly caused by the use of calcium as a substrate by algae. T.P. Lim of CANMET Elliot Lake Laboratory suggests the rise in pH is localized. Lim measured a pH of 13 on algae, while in an adjacent area inhabited by fish the surrounding water had a pH of 7. The Panel report (MEND Report 3.12.2) recommends further investigation of this phenomenon. The sulphate and iron concentrations in the deep water ranged from 50 to 100 mg/L and 0.002 to 0.4 mg/L, respectively.

Bacteria counts were taken across the basin. While the concentrations of iron-oxidizing bacteria decreased to insignificant amounts under deep water, no trend was observed in the concentrations of sulphate-reducing bacteria.

4.3.3. Performance to Date

The area has generated stable levels of low acidity, sulphate, and iron for over 10 years. However, the oxidation-reduction zone equilibrium, the development of the oxidizing zone, and the high pH values observed in summer all need further investigation.

4.4. HJERKINN TAILINGS POND, NORWAY

4.4.1. Site History and Description

Three hundred and twenty-five thousand tonnes of sulphur-containing tailings from a copper-zinc mine were deposited subaqueously in a tailings pond from 1968 to 1993. The sulphur content of the tailings varied from 18 % measured in 1976 to less than 5% measured in 1989. Copper, zinc, and iron composition in a single sample taken from the sediment in 1989 was 0.26 %, 0.49 %, and 17.5 % respectively. The area of the tailings pond is 1 km² (100 ha), and the drainage area is 13 km². The outflow from the pond varies significantly throughout the year: an average low of 70 L/s in January and an average high of 452.5 L/s in May. Minimum water depth is estimated to be 1 m, which is not sufficient to prevent resuspension (NIVA, 1993).

4.4.2. Water Cover Design

Tailings resuspension was estimated using the NHL method outlined in Section 2.2.1 above, and using Shield's criterion for bed stability. Grain size analysis showed that 80% of tailings particles were finer than 75 µm. The design wind speed with a 10-year return period was used. It was concluded that a minimum depth of 1.5 m is required to stop resuspension (NHL, 1986).

4.4.3. Performance to Date

The pH and sulfate concentrations, averaged over each year, remained steady during the deposition stage (measured from 1975 to 1991, see Fig. 4-7): pH was approximately 7.2 and SO_4^{2-} was 600 mg/L. Zinc and iron concentrations followed a net increase (from 50 to 200 µg/L and 200 to 600 µg/L respectively, Fig. 4-8) as do suspended solids and turbidity (from 1.5 to 2.5 mg/L, 1.8 to 5.1 BTU). After deposition ceased, the concentrations of zinc and sulphate decreased to 147 µg/L and 70 mg/L respectively by 1995. The sulphate fluxes in 1993 and 1995 were 0.077 mg/m²/s and 0.0035 mg/m²/s respectively (NIVA,1996). Modelling done by NIVA predicts an eventual reduction in SO_4^{2-} concentrations to 6-14 mg/L and zinc concentrations to 10-20 µg/L (NIVA,1993). The high concentrations are predicted to occur in winter when the discharge volume and therefore the flushing of the water cover is low. It is shown in Fig. 4-9 that the molar ratio of calcium and sulphate remains approximately one throughout the measurement period, indicating that the dissolution or precipitation of gypsum controls the amount of sulphate in the water cover.

4.5. LØKKEN, NORWAY

4.5.1. Site Description and History

The pond was constructed in 1974 and subaqueous disposal of tailings continued until 1987. A total of 3.2 million tonnes of tailings were deposited in the pond. The tailings dam had no core, as it was built from waste rock. Since the original construction, a geomembrane liner has been placed on the face and held down with poured concrete.

Riprap was used to strengthen the upstream face of the dam against wave action and till was placed on the downstream face and then vegetated. The tailings dam is 25 m high and 75 m wide at the bottom.

The pond has an area of 200,000 m². For most parts of the nonfreezing year, the pond water level is typically at the height of the dam and the minimum water cover depth is 1 m. Some areas are as deep as 4 m (based on observations made by NIVA). The tailings are composed of 40 % sulphur, 0.32 % zinc and 0.32 % copper (NIVA 1993).

4.5.2. Water Cover Design

Tailings resuspension was not considered in the design and construction of the dam. NIVA conducted chemical modelling in an attempt to predict post-deposition effluent concentrations.

4.5.3. Site Investigations

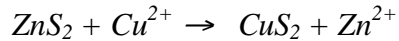
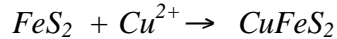
The following information is largely based on the first author's (Dr. E. K. Yanful) personal observations and discussion with site personnel during a visit in July 1996.

Pollution for the whole Løkken area has been controlled by collecting all seepage from mine waste between the Raubekken and Bjornlivation rivers (see Fig. 4-10). The seepage is pumped into an underground mine where heavy metals and suspended solids settle to the bottom. Clean water is pumped out of the mine. Approximately 500,000 m³ of contaminated water with a copper concentration of 300 mg/L and pH 2.4 are pumped into the mine. The water pumped out of the mine has a copper concentration of 1-2 mg/L (a decrease of 99%) and a pH of 6.4.

The decrease in Cu concentration has been explained as follows:

- 1) Oxidation of pyrite by iron (III) contained in the mine water lowers the pH
- 2) Acid mine water interacts with calcium, magnesium, and aluminum-rich minerals: This increases the pH and precipitates iron (III).
- 3) Under these conditions, copper is exchanged for iron and zinc in pyrite, as described

by the following reactions:



X-ray analysis has confirmed the formation of the above minerals.

Pumping and treatment at Løkken using the mine will continue for the next 20 years and will then be re-evaluated. The cost of treatment is NOK 250,000 (Norwegian Kroner) or about CAN\$50,000 per year. The ORKLA Group of Companies is responsible for the cost of treatment. Prior to pumping and treatment, salmon fish would not spawn in the body of water immediately downstream of the mine area, the inlet of the Orkdalsjord between the Orjkla River and Orkanger (see Fig. 4-10). They would instead spawn in the Orkla, an adjacent river. However, the situation has changed since the start of treatment because of the improvement in water quality.

4.5.4. Performance to Date

During the period of tailings deposition, the tailings pond effluent quality varied as follows: pH decreased from 10.1 to 5.3 (Fig. 4-11); sulphate remained approximately steady at 800 mg/L (Fig. 4-12); iron increased from 200 to 500 µg/L (Fig. 4-13); copper increased from 200 to 800 µg/L (Fig. 4-14); and zinc increased from 500 to 1200 µg/L (Fig. 4-15). Since deposition ended, the pH has remained between 4 and 6, sulphate has decreased to approximately 100 mg/L, copper between 0 and 600 µg/L, and zinc has increased to about 2500 µg/L. The concentrations of these species fluctuate with outflow volume. The outflow rate varies significantly over the year; the lowest flows (0.3 to 2.0 L/s) occur in July through January and the highest (average of 24.8 L/s) in April. High pH values and low species concentrations occur during periods of high flow (NIVA, 1996b).

The sulphate flux from the tailings was approximately 0.06 mg/m²/s in 1988 and 0.003 mg/m²/s in 1995. Water quality modelling conducted by NIVA predicts sulphate concentrations decreasing to 100 mg/L, and zinc concentrations decreasing to 800 µg/L (NIVA, 1993). However, zinc concentrations currently vary between 2000 and

3000 µg/L. Researchers at NIVA suggest the oxidation of thiosalts diffusing into the water cover from partly-oxidized tailings and the effects of tailings resuspension account for the discrepancy between predicted and observed zinc concentrations. As shown in Fig. 4-16, the nearly identical molar concentrations of calcium and sulphate since deposition stopped indicates that sulphuric acid generated by oxidation of pyrite or thiosulfate is neutralized by calcium carbonate or calcium hydroxide in the tailings.

4.6. STEKENJOKK, SWEDEN

4.6.1. Site Description and History

Stekenjokk is located in northern Sweden, close to the Norwegian border. Zinc and copper deposits were mined at Stekenjokk between 1976 and 1988. The site included a tailings and clarification pond 110 ha in area, waste rock dumps and a small open pit. About 4.4 million tonnes of tailings containing about 20% sulphur, mainly occurring in pyrite were deposited in the pond prior to decommissioning. The grain size of the tailings is 80% smaller than 80 µm, D_{50} is 25 µm and D_{90} is 130 µm. Permanent flooding was selected as the decommissioning option and completed in 1991. Flooding involved lowering the pond water level and resurfacing the tailings. A grid of waste rock breakwaters was constructed in the shallow pond area to limit resuspension by wave action. The breakwaters run in two perpendicular directions to divide the pond into cells. The water cover depth ranges from 0.7 to 1.0 m in the shallow portion (see Fig. 4-17) and between 5 and 6 m in the deep portion. A sand layer was also added in the shallow portion (Broman and Göransson, 1991; Broman, 1989).

4.6.1.1. Water Cover Design

The water cover design for Stekenjokk was based on 1) the selection of hydrological parameters to ensure the tailings were covered even in the event of a 1000-year drought, and 2) the use of breakwaters to minimize oxygen transport and tailings resuspension by wave action. It was estimated that the water level would drop 20 cm in the event of a drought. The Norwegian Hydrotechnical Laboratory (NHL) calculated the amount of resuspension occurring for different fetch lengths using a design wind speed of 22 m/s.

NHL also investigated the feasibility of a proposed 5 cm sand layer as a surface protection. The calculated required depth varied between 0.2 to 2.2 m for fetch lengths 0.05-1.10 km without a sand layer, and 0.2-0.93 m with a sand layer. It was decided that a sand layer should be added to the shallow section, where the minimum required water depth was 0.5 m (NHL 1989 and 1987).

The Norwegian Water Research Institute (NIVA) conducted geochemical modelling to predict metal and sulphate flux from the tailings. The prediction assumed that the tailings do not resuspend and no significant groundwater flow occurs. The results showed very small fluxes of copper, lead, and iron; zinc and sulphate fluxes were 0.0002 mg/m²/s and 0.005 mg/m²/s, respectively. NIVA calculated that adding a sand layer at the tailings-water interface would reduce zinc flux by a factor of 16 to a low value of 0.0000014 mg/m²/s (Broman, 1989).

The philosophy behind the decommissioning of the Stekenjokk tailings was as follows (Personal Communication with Per Broman):

- 1) The use of wave hydraulics to calculate the minimum depth of water necessary to prevent resuspension.
- 2) Due to the strong winds at the site, the implemented depth of water is well mixed and is found to be saturated with oxygen for the entire water column, except for a slight decrease in concentration at the bottom where the tailings appear to be consuming small amounts of oxygen.
- 3) It is recognized that suspended tailings would oxidize and that small DO gradients would produce some oxidation as well. The resulting oxidized layer will serve a 'sand' layer and prevent further oxidation and resuspension of the underlying unoxidized tailings.

4.6.2. Site Investigation

During a visit to the site in July of 1996, seepage of brownish red water at the toe of the embankment was observed. The seepage contains very few metals. A plume of remobilized metals in the embankment is believed to be the cause (P. Broman, Personal Communication).

At the shallow end of the pond, where the depth of the water cover is 20-30 cm, about 2 to 5 mm of a dark brown, fluffy deposit probably consisting of iron oxyhydroxides and organic material can be seen at the tailings interface with the waste rock (breakwaters). This fluffy deposit has also formed in the deeper sections of the pond, where regularly spaced brownish-yellow streaks of exposed tailings were observed. It is possible that waves produced during strong winds were powerful enough to mobilize the surface organic layer and expose the previously oxidized tailings underneath.

Tailings samples taken after decommissioning contain 17.5 % S, 27.6 % Fe₂O₃, Cu 1900 ppm, Pb 1600 ppm, and Zn 6600 ppm. The gangue minerals in the tailings are calcite and quartz, while the predominant sulphide minerals are pyrrhotite and pyrite. The pH in the winter is about 7.2, and increases to 7.60 from July to September. The ice cover, present from October until June, may stop the inflow of alkalinity from the surroundings. In the summer, CO₂ degassing would tend to raise alkalinity. The concentrations of all chemical species in the water cover are higher in the winter. During winter, an upward flux of Mn is observed, which precipitates as MnO₂. This precipitate absorbs Co and Cd. The ratio of suspended solids to total solids in the water cover is very low, ranging from 1 to 5 %.

4.6.3. Performance to Date

During 1992-1995, the zinc flux decreased from 0.0002 to 0.000002 mg/m²/s and sulphate flux from 0.003 to 0.0007 mg/m²/s. Zinc concentrations in the discharge had declined from 0.48 to 0.03 mg/L and sulphate from 51 to 14 mg/L by 1996. Suspended solids measured at the discharge were below the detection limit (< 5 mg/L). Analysis of 1996 water samples performed by Boliden Minerals Central Laboratory indicated the

leakage from the downstream dike contains similar loads as the pond discharge. In the pond, very low concentrations of mostly organic solids were measured for depths between 1 and 6 m. Annual measurements of the wave breakers indicate that they are not subsiding, even though they are built on tailings (Nils Eriksson, Personal Communication).

4.7. SOLBEC-CUPRA, QUÉBEC, CANADA

4.7.1. Site Description and History

The Solbec-Cupra tailings site is located near the village of Stratford, 100 km east of Sherbrooke, Québec, Canada. Tailings from nearby zinc, copper and lead mines were deposited from 1962 to 1977 over an area of 66 ha, 20 ha of which were submerged. The tailings contained 1.56% copper, 4.5% zinc, 0.69% lead, 52.11 g/t silver and 0.58 g/y gold (Vézina, 1996). The total mass of tailings was 4.16 million tonnes. By 1993, the tailings had oxidized to a depth of 0.5 m and the site effluent had a pH of 3 and contained metal concentrations of 13 mg/L Fe, 1.0 mg/L Cu and 3.0 mg/L Zn. A water cover was implemented in 1994 and the minimum water depth of 1 m was first reached in January 1996.

4.7.2. Water Cover Design

Several years of field pilot tests and laboratory experiments described in Section 5.2 and 5.7 suggested that mixing the upper 15 cm of tailings with lime and then flooding with a 1-m water cover would improve the quality of the site effluent to the pertinent water quality standards. The design required levelling the tailings to an elevation of 329 m or less, constructing two dikes and then flooding to elevation 330 m to achieve a minimum water depth of 1 m. The dikes were completed by the fall of 1994. By fall, 1995 the water level had reached an elevation of 329.35 m. The total water stored in the reservoir was 1.45 million m³, which was two million m³ less than predicted by the water balance. The deficit may include water used to saturate tailings and soil in the reservoir above the water table, the loss of water under and through the dikes, and the loss of water to vertical and lateral groundwater seepage. The water level reached 330 m in January 1996.

4.7.3. Site Investigations

Sampling of the water cover, tailings pore water, and pore waters of the underlying stratigraphic units (peat, till and bedrock) was conducted in 1995 and 1996. Additional samples were taken from the outflow and inflow waters. The water cover in the tailings basin was sampled at three different depths at each of six stations: 0.15 m below the water surface, 0.15 m above the tailings, and at mid-depth of the water cover. The concentrations of metals in the water cover were all low or below detection. Les Consultants SM of Sherbrooke is currently monitoring the site. The pH and sulphate and metal concentrations, averaged across the six stations, are presented in Table 4-1.

Table 4-1: Average pH and Metal Concentrations in the Solbec-Cupra Tailings Basin (Yanful, 1997)

Chemical Species	1995		1996		
	August	November	June	August	November
SO ₄ ²⁻ (mg/L)	240	221	137	120	146
PH	6.7	7.6	7.8	7.2	7.3
Fe (µg/L)	1840	345	389	178	391
Cu (µg/L)	10	10	<5	24	33
Zn (µg/L)	22	40	8	24.2	188

None of the parameters in Table 4-1 changed significantly with water depth. The outflow or effluent waters have a similar pH and contain metal concentrations all less than 200 µg/L. There is no significant variation in pH between the inflow water, the water cover, and the effluent. In the water cover, dissolved oxygen (DO) ranged from 9.24 to 13.00 mg/L. No trend in DO was observed with depth.

The pH and metal concentrations in the different stratigraphic units were measured at three locations where platforms containing duplicate piezometers for each layer were constructed. The results for Platform 1 are presented in Table 4-2.

Table 4-2: Pore Water Quality at Platform 1, Solbec-Cupra, June 1996
(Modified from Yanful, 1997)

Layer	SO ₄ ²⁻ (mg/L)	Fe (µg/L)	Cu (µg/L)	Zn (µg/L)	PH
Oxidized	242	103	<5	1400	7.3
Tailings	6200	257	26	3200	6.3
Unoxidized tailings	6500	247	41	2900	6.3
Peat	24	464	8	35	6.3
	2	984	10	140	6.5
Till	6	110	41	41	7.4
	18	24	10	10	7.8
Bedrock	139	318	<5	<5	8.1

Table 4-2 shows that the reproducibility of the pore water data is not good. There was also a large variation in the results from platform to platform (Yanful, 1997). For example, the concentration of iron measured in the oxidized tailings during 1995-96 ranged from below detection in one piezometer at Platform 3 to 937 mg/L at Platform 2.

Yanful (1997) used the thermodynamic equilibrium speciation model MINTEQ to

identify which minerals were controlling metal solubility in the water cover. The water cover was found to be undersaturated with respect to gypsum, which would control the solubility of calcium and sulphate. Iron was controlled by the precipitation of ferric oxyhydroxides including ferrihydrite, goethite, and lepidocrocite; as well as by jarosite.

Qualitative observations of wind/wave behaviour and resuspension were made in 1996. There are three different areas of the pond where different wave behaviour occurs. In the central part of the basin, waves have amplitudes up to 300 mm; in areas with reeds and dead tress, the wave amplitudes were less than 150 mm, while in two areas located at the south end of the basin, waves were nearly nonexistent. Observations of resuspension were made near the main dike, on the southwest side of the effluent canal. Here, the slope of the bottom is very gentle. The water is very murky up to 1 m from the shore and remains fairly turbid up to 20 m from the shore. The suspended sediments appear to be very fine, suggesting they are derived from soil and not tailings. Further work on resuspension will be conducted in Spring 1997.

4.7.4. Performance to Date

All water quality parameters in the water cover and effluent measured in 1996 conform to Directive 019. The concentration of iron in the water cover appears to decrease with time, while the concentrations of other heavy metals are steady but low. The concentration of zinc in the water cover appears to be increasing with time, though the concentration is still not approaching the regulatory limit. The impact of seepage on downstream groundwater quality remains to be assessed.

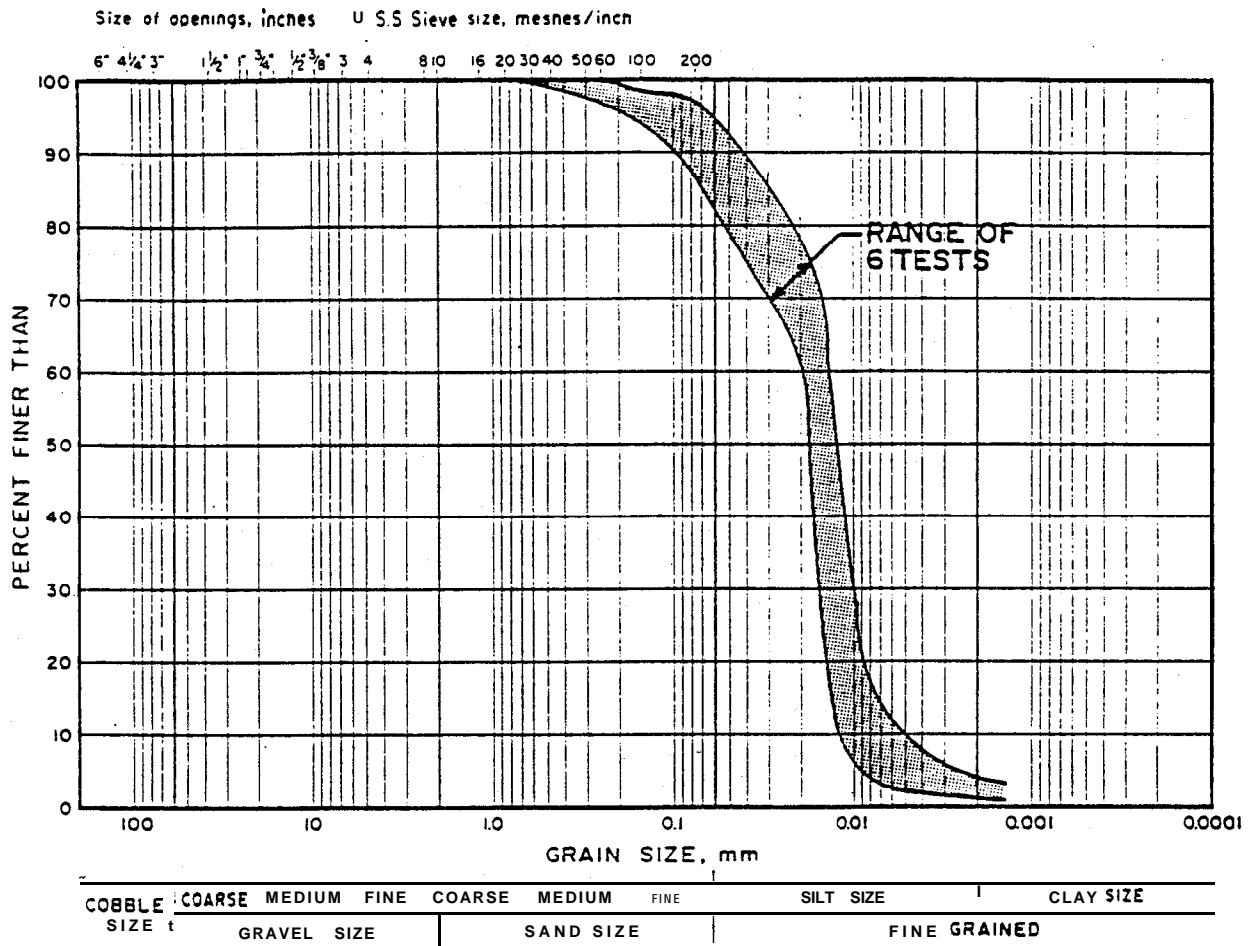


Figure 4-1

**Grain size distribution of tailings from Cell 14, Quirke Tailings Basin,
Elliot Lake, Ontario**

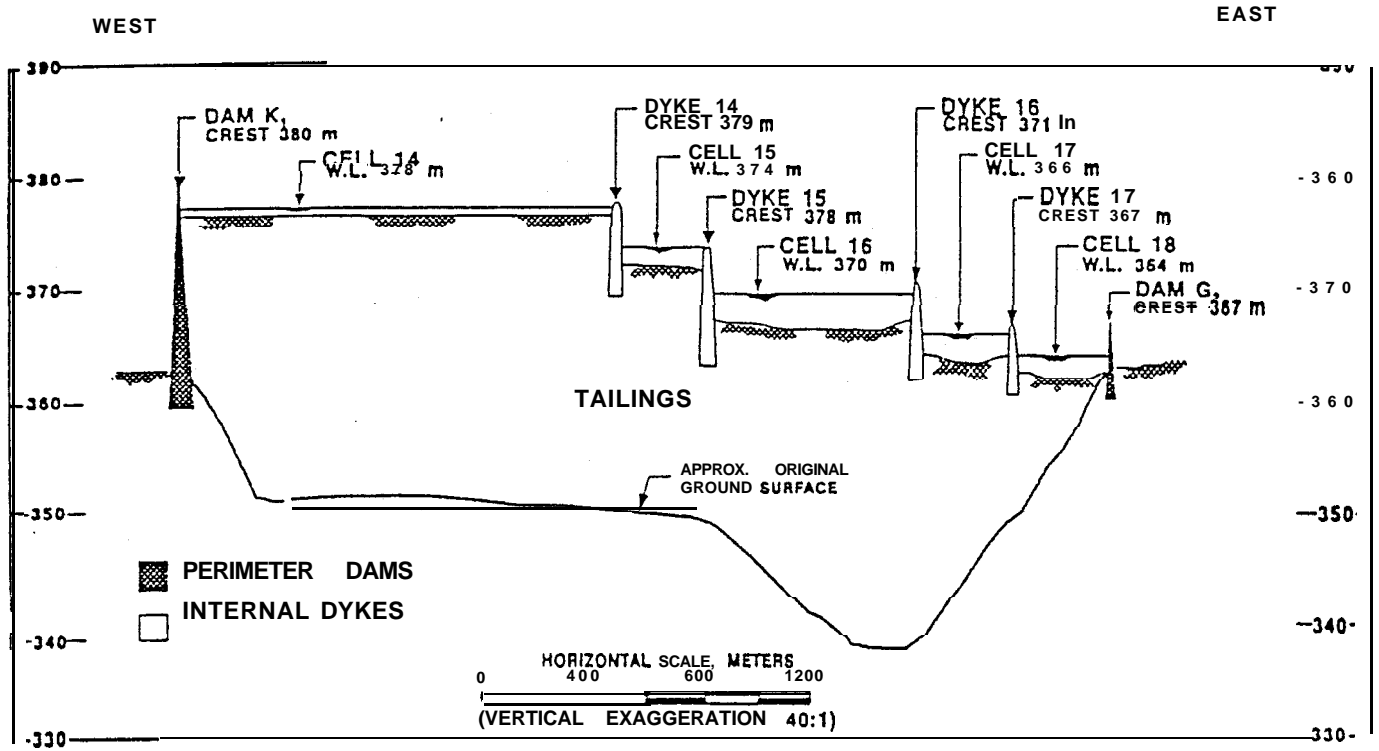


Figure 4-2

Schematic profile of flooding concept, Quirke Tailings Basin, Elliot Lake, Ontario

CELL 14

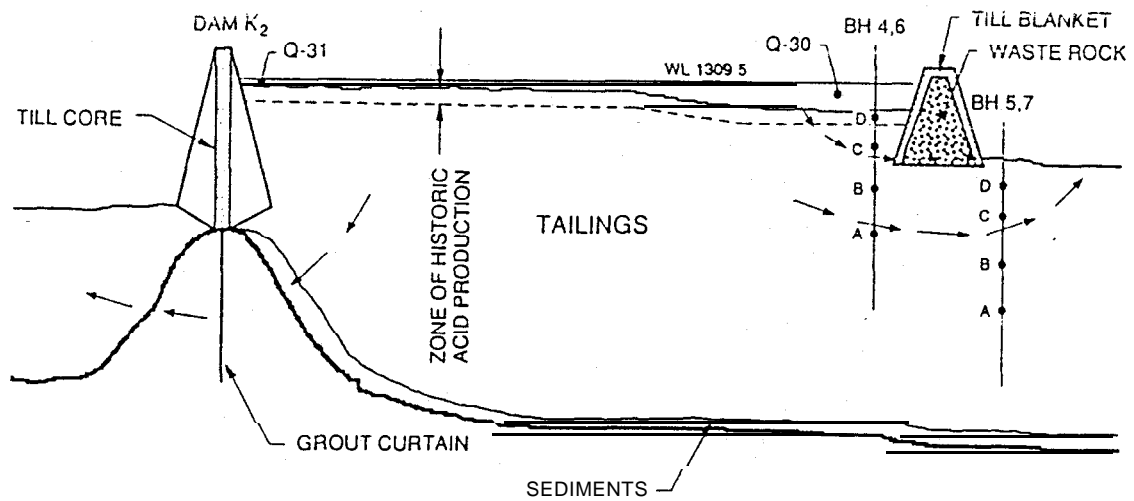


Figure 4-3

A schematic section of Cell 14, Quirke Tailings Basin, Elliot Lake, Ontario

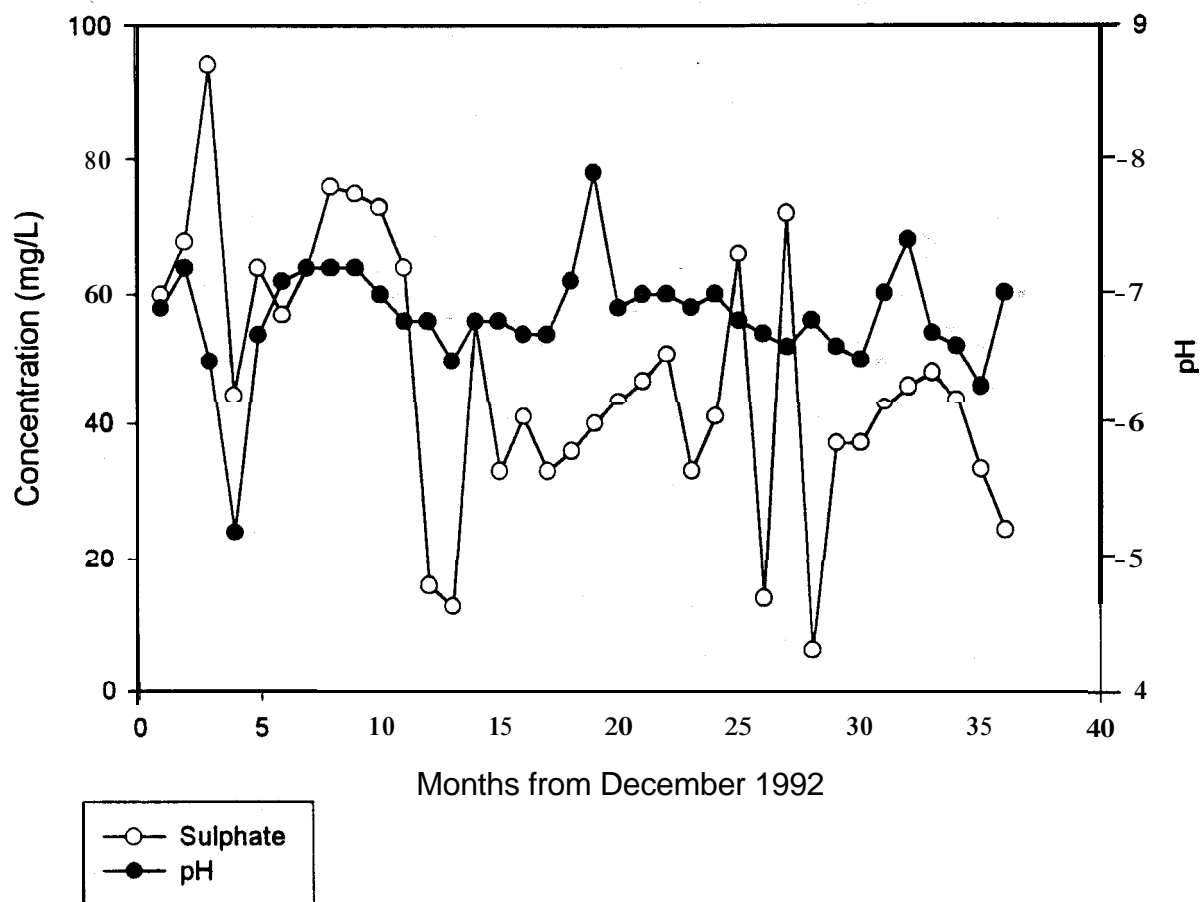


Figure 4-4

Cell 14 Quirke Tailings Basin Water Quality - Sulphate and pH

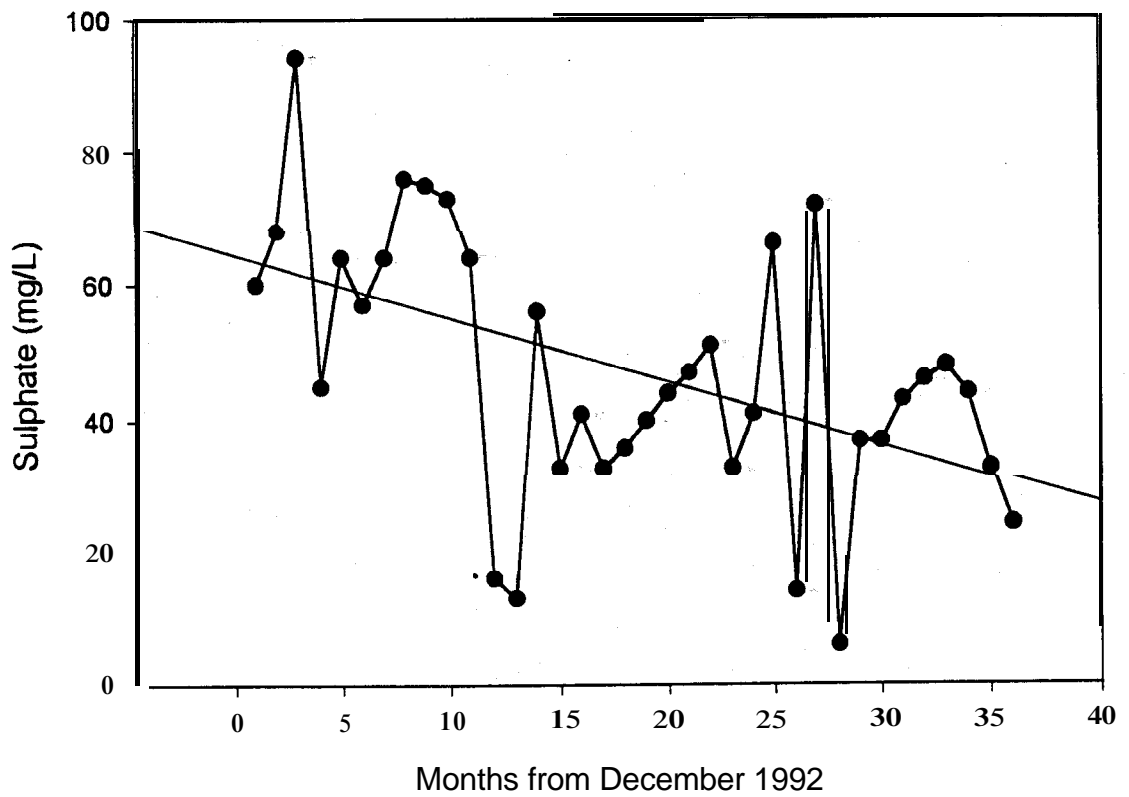


Figure 4-5

**Cell 14 Quirke Tailings Basin Water Quality
Sulphate concentration fitted with linear regression**

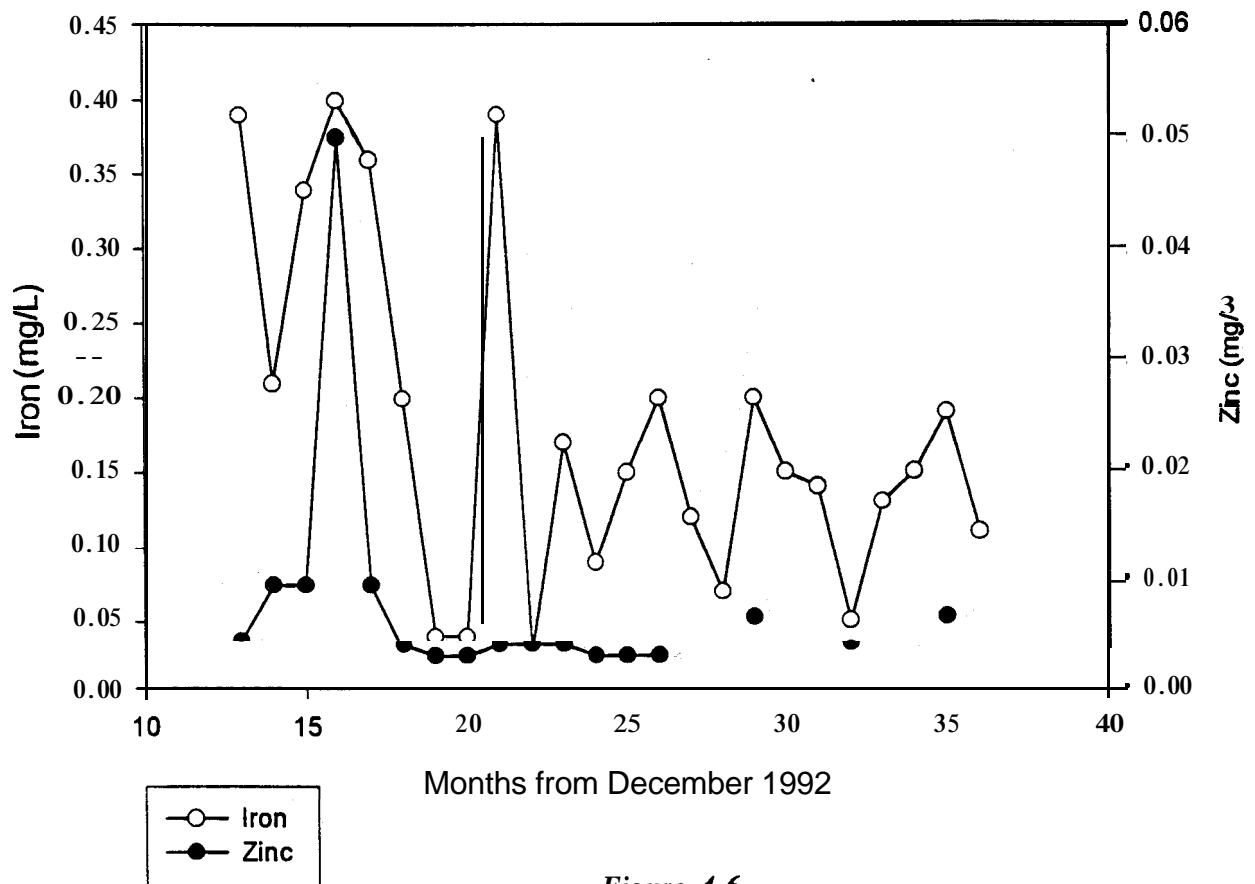


Figure 4-6

Cell 14 Quirke Tailings Basin Water Quality • Iron and Zinc

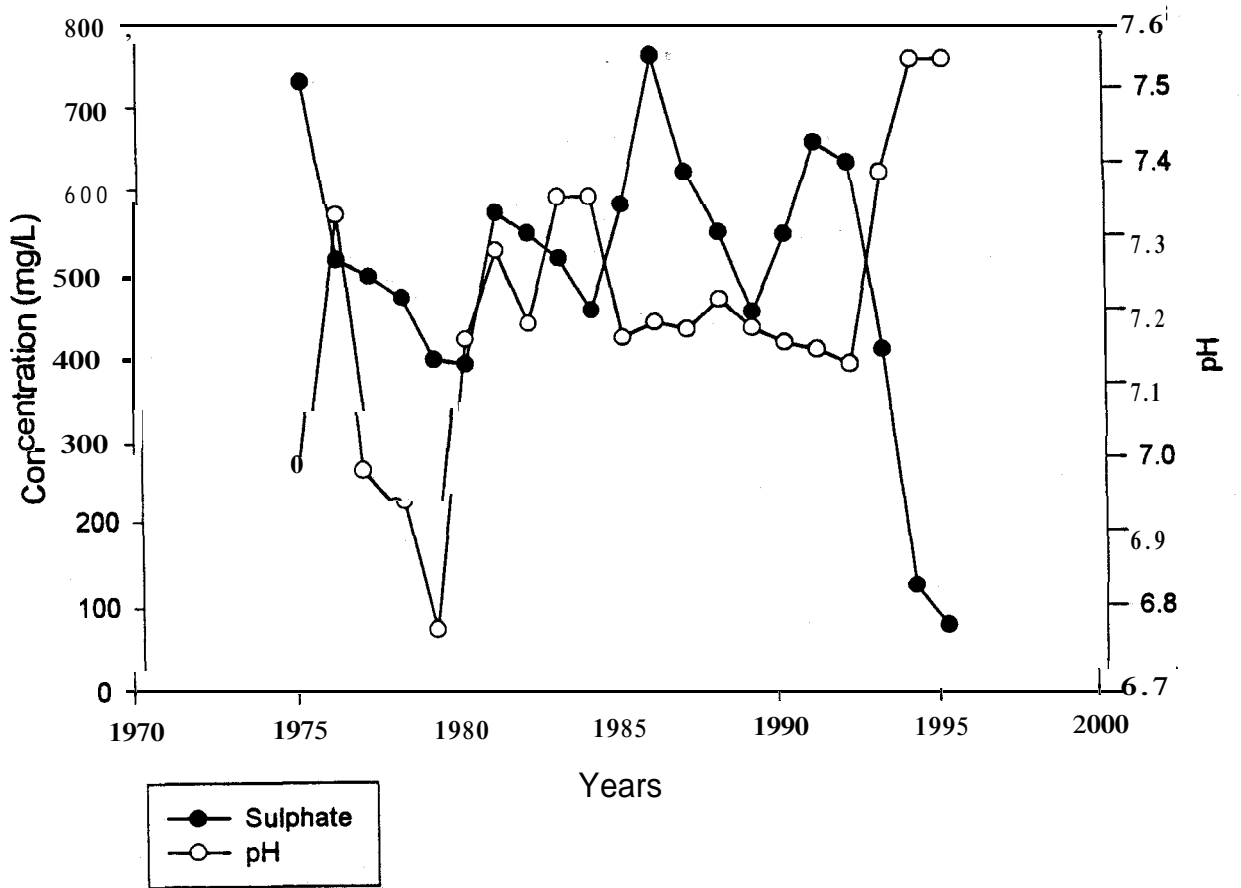


Figure 4-1

Hjerkind Pond Water Quality - Sulphate and pH

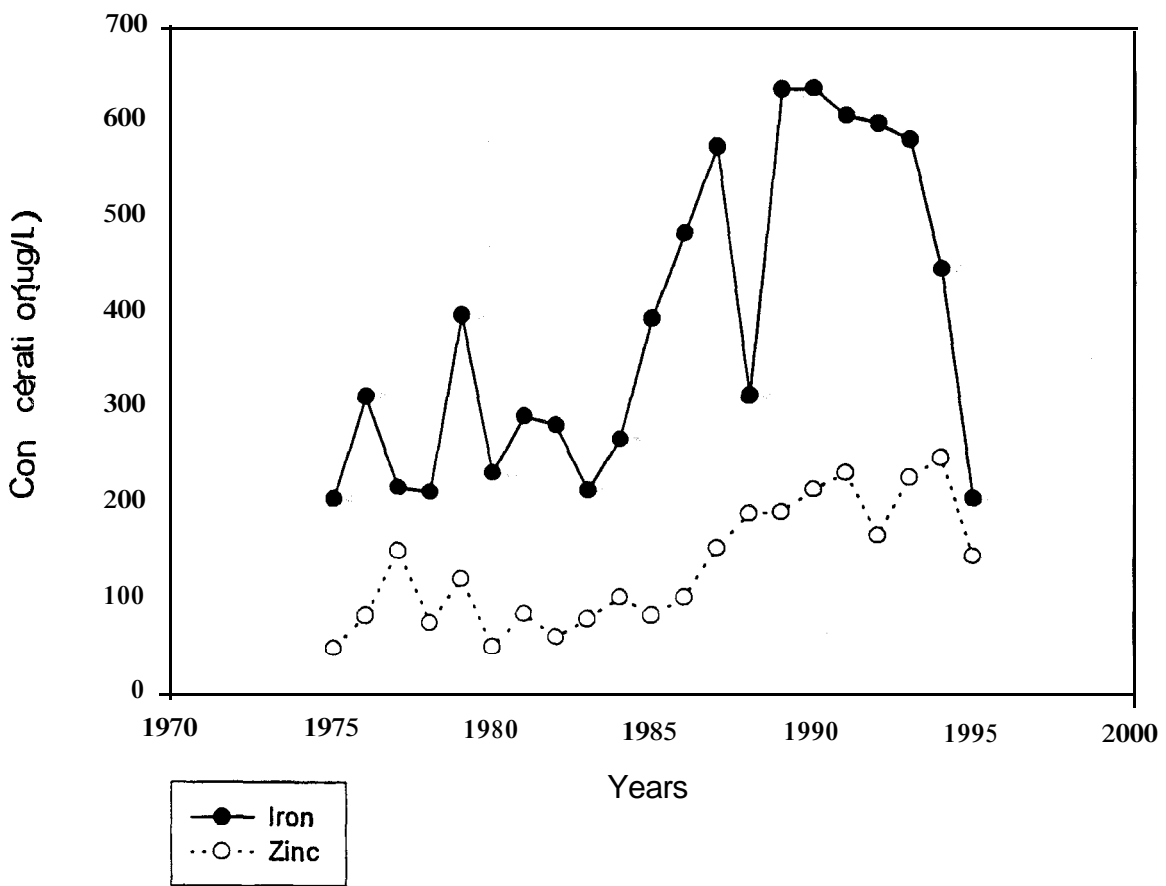


Figure 4-8

Hjerkind Pond Water Quality - Iron and Zinc

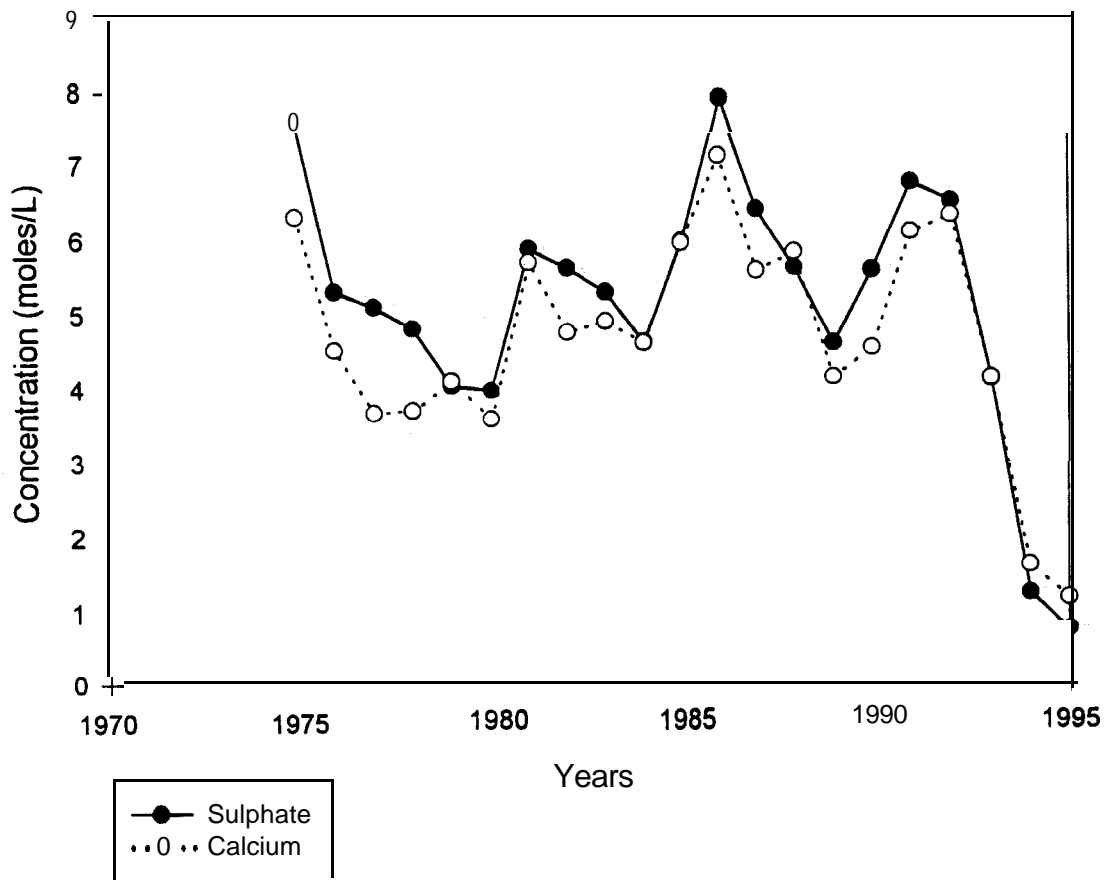


Figure 4-9

Hjerkind Pond Water Quality - Sulphate and Calcium

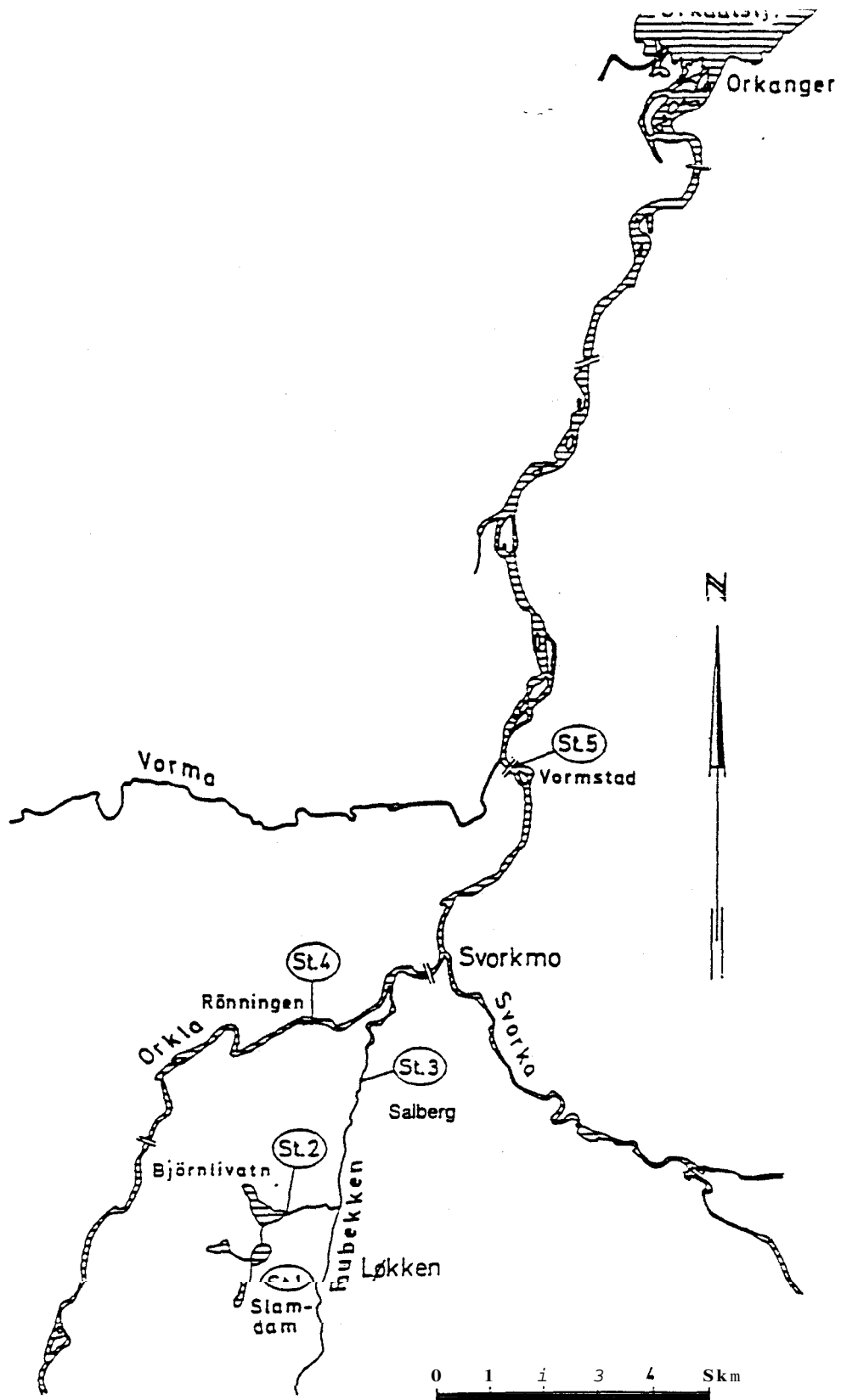


Figure 4-1 0

River system downstream of Løkken Mine, Norway

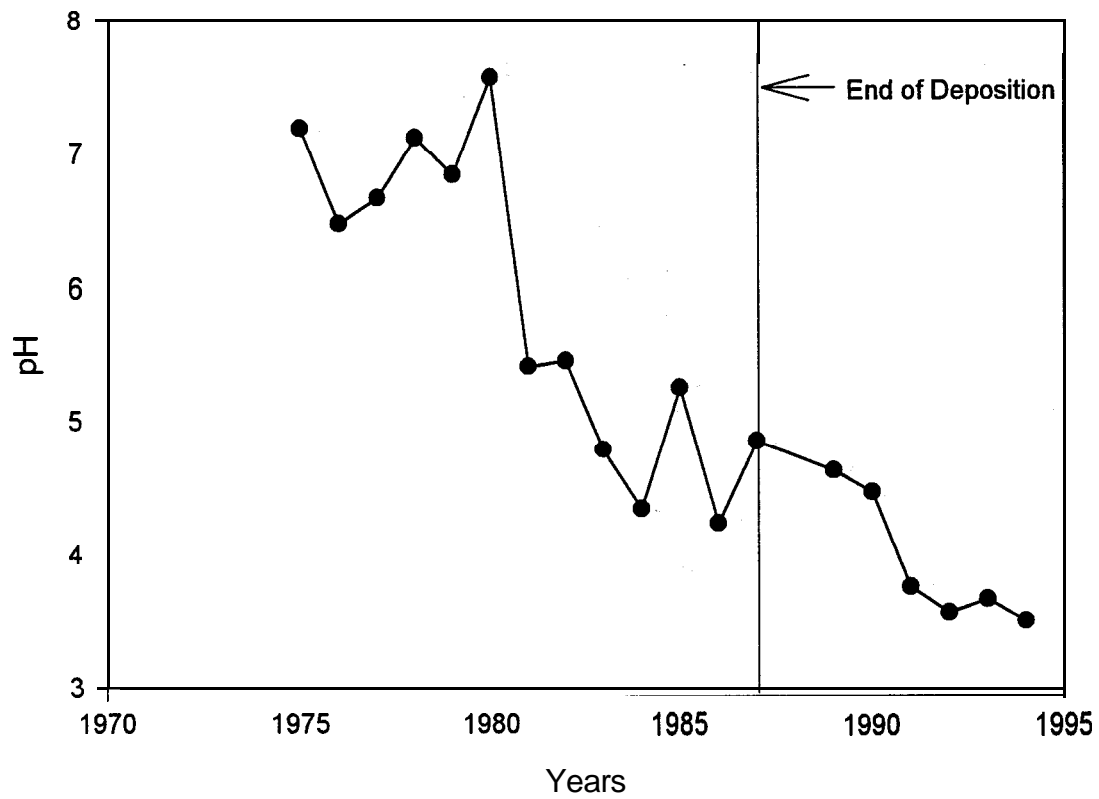


Figure 4-1 1

Lekken tailings pond pH

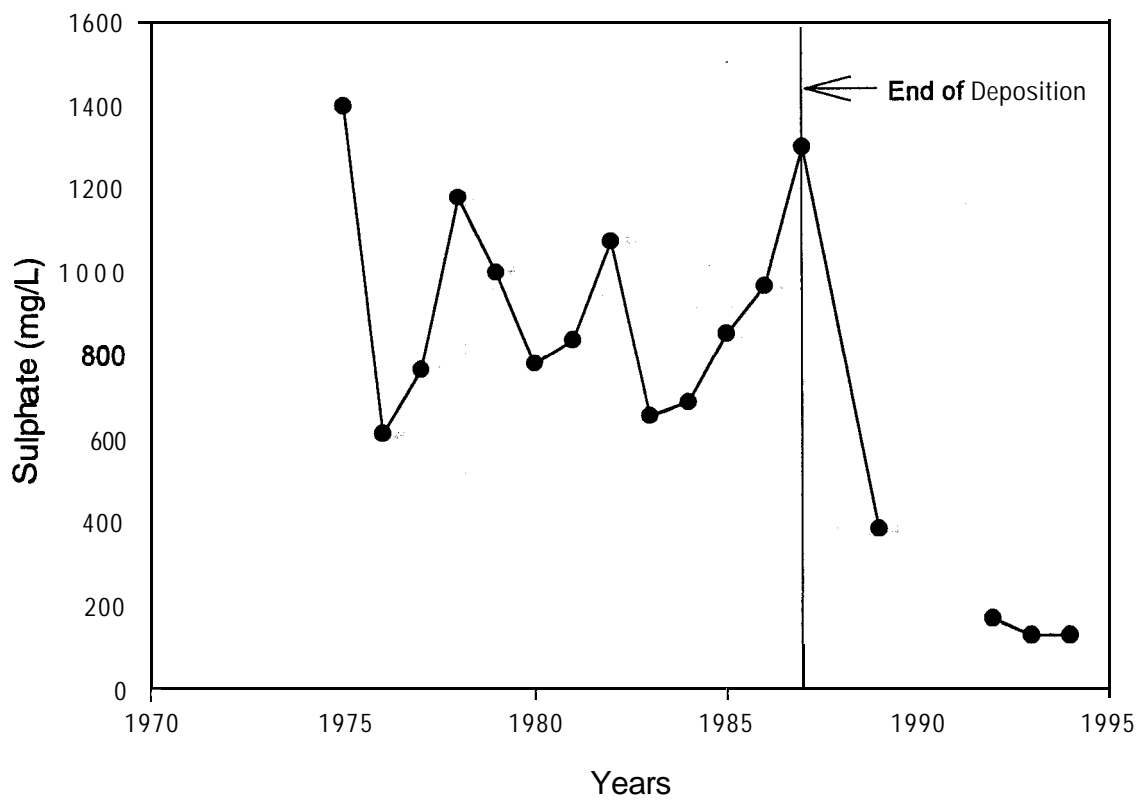


Figure 4-12

Lekken tailings pond sulphate concentrations

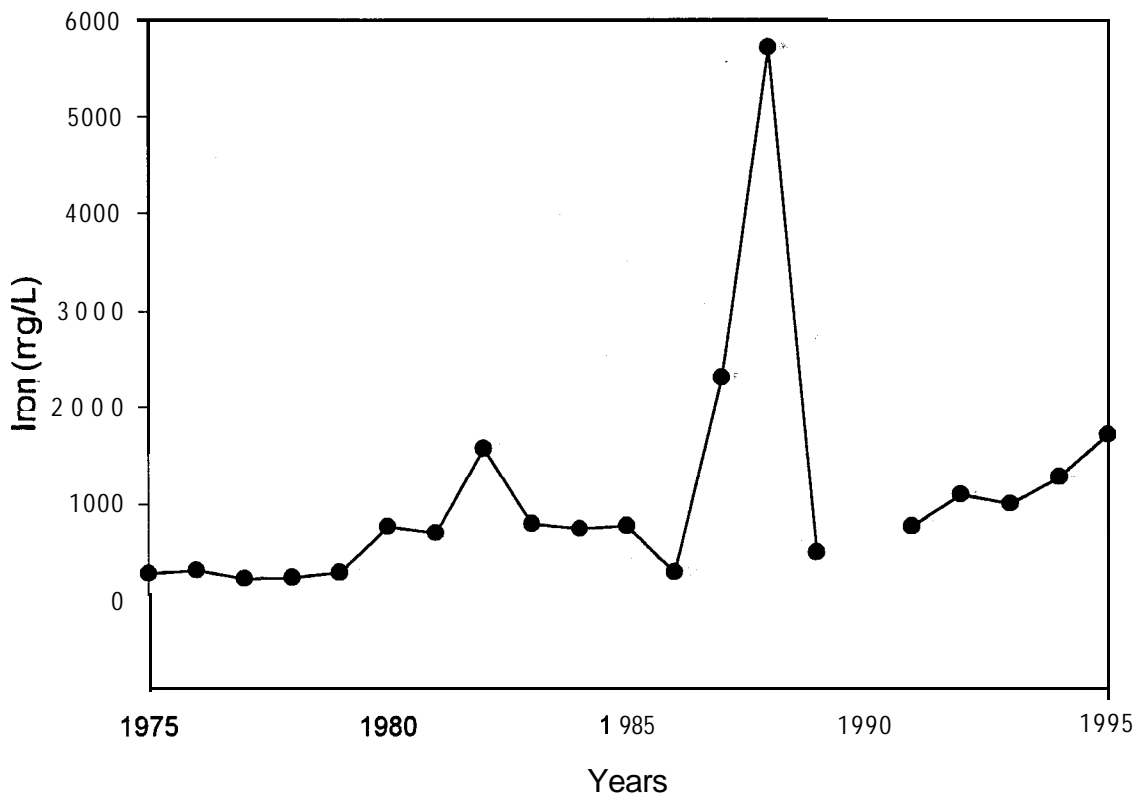


Figure 4-1 3

Lekken tailings pond iron concentrations

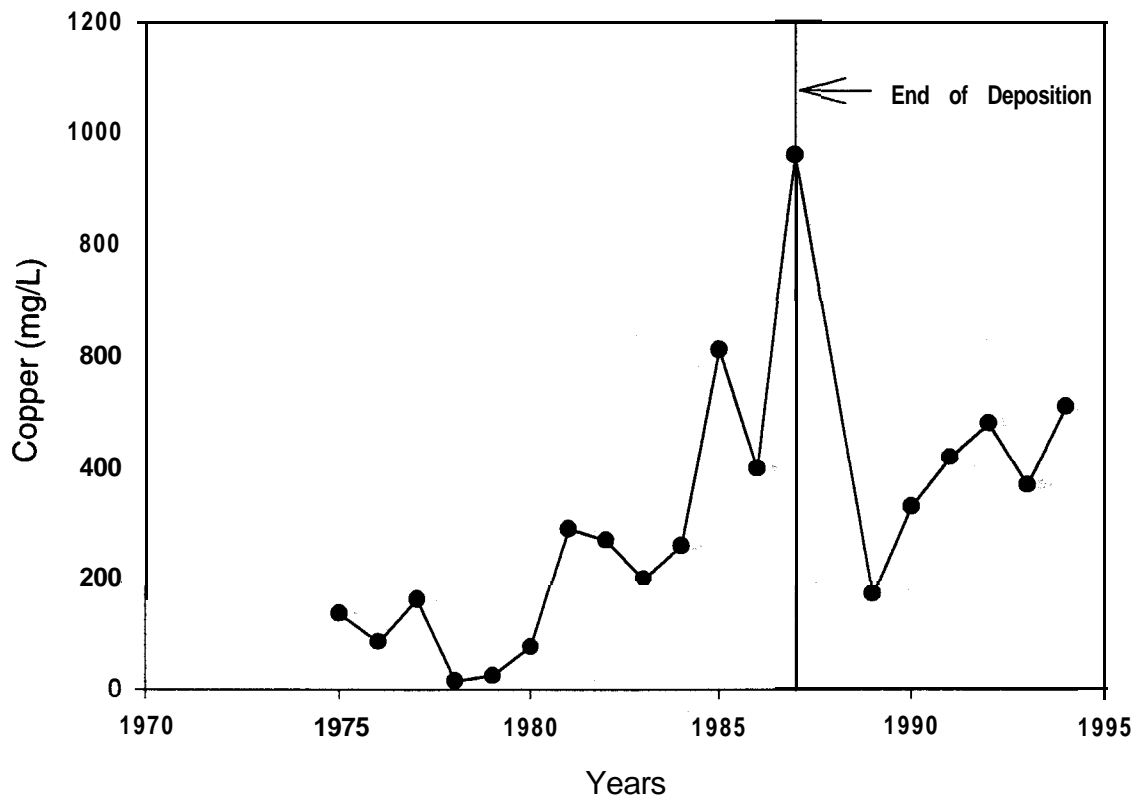


Figure 4-1 4

Lekken tailings pond copper concentrations

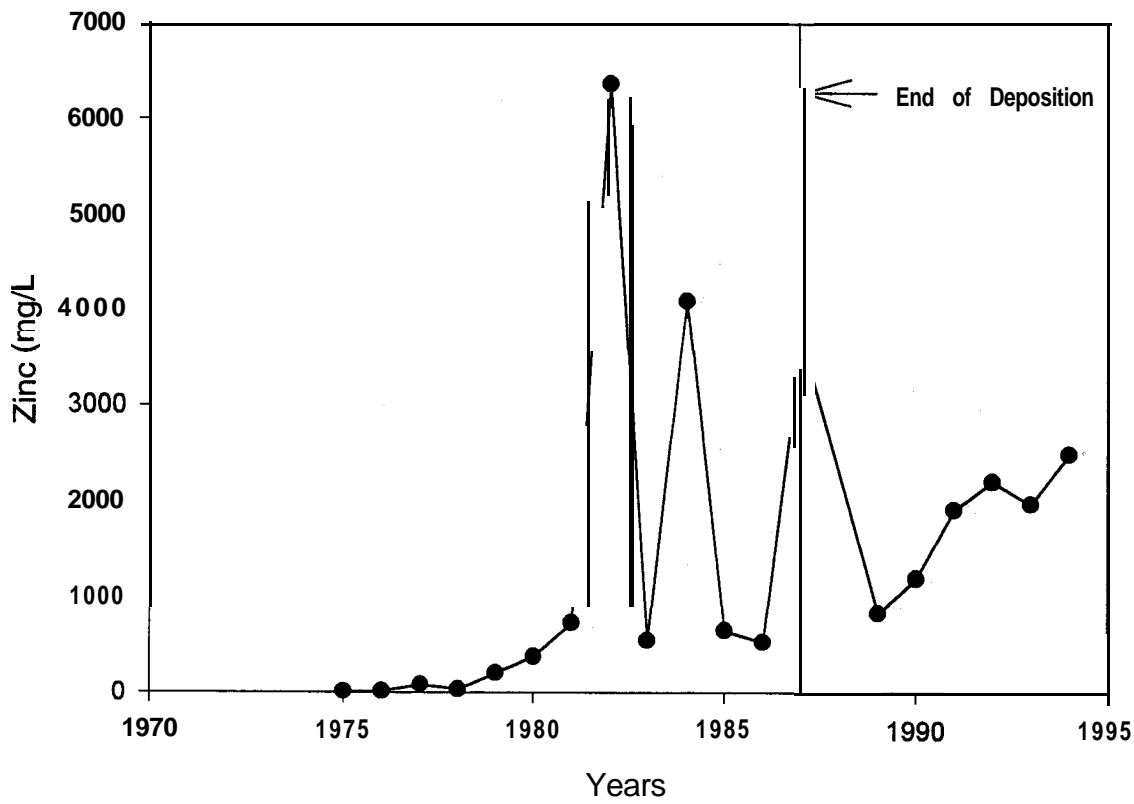


Figure 4-15

Løkken tailings pond zinc concentrations

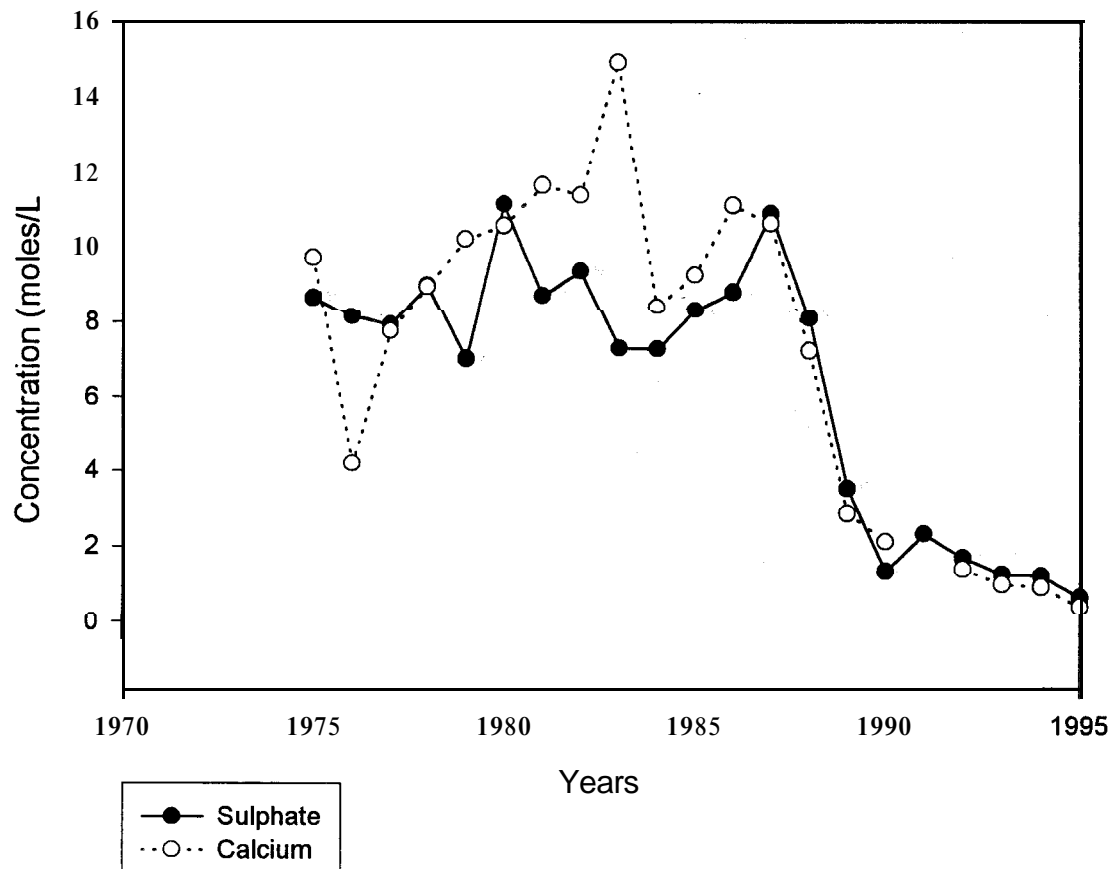


Figure 4-1 6

Lekken tailings pond sulphate and calcium concentrations

5. REVIEW OF LABORATORY AND FIELD RESEARCH ON WATER COVERS

The following chapter documents recent laboratory and field research on water covers. At the end of each section or subsection the contributions and limitations of the pertinent study are summarized.

5.1. NORANDA TECHNOLOGY CENTRE (NTC)

Noranda Technology Centre (NTC) has conducted a series of laboratory experiments to understand oxidation and metal dissolution in sulphide tailings flooded with a shallow water cover. The results of the research were reviewed and partially interpreted by Paul Simms during a visit to NTC in June-July 1996.

The NTC research includes:

1. Laboratory experiments on the performance of water covers over unoxidized and oxidized tailings under stagnant and flushing conditions and diffusion and column studies (St-Arnaud and Yanful,1994; St-Arnaud and Yanful, 1993; Noranda Technology Centre Internal Reports).
2. Pilot scale field tests to evaluate the performance of a water cover over oxidized tailings (St-Arnaud and Yanful, 1993),
3. Laboratory and field tests to evaluate the performance of biologically supported water covers (BSWC) using laboratory tests and field studies (NTC Internal Reports),
4. Modelling of long term site behaviour using the WATAIL model (NTC Internal Reports), and

5. Theoretical investigation into the effects of wind on tailings resuspension and oxygen transport (NTC Internal Reports).

5.1.1. Diffusion Cell and Column Experiments on Tailings

5.1.1.1. Diffusion Cell Tests

Diffusion tests were undertaken to evaluate the diffusion and partitioning of zinc in fresh, unoxidized Mattabi tailings. The diffusion tests involved tailings that were submerged in different aqueous solutions including a solution of zinc chloride. The solutions were stirred continuously and zinc concentrations were measured periodically throughout the test period. The vertical diffusive flux of zinc into the tailings was simulated using the computer program POLLUTE (Rowe and Booker, 1993). POLLUTE uses a semi-analytic, finite-layer technique to solve the diffusion equations describing flux and transient change in concentration at a particular depth (Fick's laws), under appropriate boundary conditions. The best-fit theoretical solution to the experimental data used an effective diffusion coefficient of $0.0086 \text{ cm}^2/\text{day}$ and a distribution coefficient of $147 \text{ cm}^3/\text{g}$ between adsorbed and soluble zinc.

5.1.1.2. Laboratory Column Experiments

Objectives

Laboratory experiments involving water-covered columns of oxidized and unoxidized tailings from the Mattabi tailings impoundment, northern Ontario, were performed. The principal aim of the experiments was to assess the relative performance of a water cover based on the following criteria:

- 1) Depth of water cover
- 2) Type of protective layer, in this case sand or peat.

The first criterion was evaluated using duplicate column tests, two with a 1-m water cover over tailings, and two others with saturated tailings but no water cover. The second criterion was evaluated using duplicate columns with a 1-m water cover combined with either a 10-cm sand or peat layer. The columns are described in Table 5-1.

5.1.1.3. *Table 5-1: NTC Column Experiments*

Column Identification	Depth of Water Cover	Surface Protection
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

Experimental Methods

Eight, 2-m high 30-cm diameter, transparent Plexiglas columns were used for these experiments. Figure 5-1 shows a typical column assembly. Each column was built with two 1-m sections. The lower sections were filled with 60 cm of relatively unoxidized tailings and 35 cm of oxidized tailings collected from the north portion of the Mattabi tailings impoundment. The tailings typically contained 33% pyrite, 1.5% sphalerite and 1% other sulphide minerals. Columns MTWC-1 and MTWC-2 were constructed first and were filled with Sturgeon Lake water (a neighbouring lake at the Mattabi site). The six other columns were constructed almost a year later and filled with distilled water.

Each column was equipped with 28 septum ports, 14 to allow sampling of pore water in the tailings, and the other 14 to monitor the quality of the water cover at different depths. There was also a bottom outlet that was closed during the static test, but open to sample the effluent when water was allowed to flow through the tailings in the columns.

Water samples were preserved and analyzed for metals and sulphur by inductively-coupled plasma spectrophotometry (ICP). Acidity was determined by titration and the pH by combination electrode and appropriate buffers.

The experiments were conducted in two phases: a static (no flow) phase and a kinetic phase, where 1 litre of distilled water was added per week to the top of each column to

simulate precipitation, and effluent was collected at the bottom. Throughout the experiments, water cover and pore water samples were collected at the septum ports. Table 5-2 presents the duration of each phase, in days, and the total volume of effluent collected for each column in pore volumes.

5.1.1.4. Table 5-2: Protocol for NTC Column Experiments

Column	Duration (days)		Pore Volumes of flow
	Static state	Kinetic state	
MTWC-1	348	605	2.43
MTWC-2	348	605	2.59
MTWC-3	183	468	2.02
MTWC-4	183	468	2.07
MTWC-5	183	468	1.98
MTWC-6	183	468	2.00
MTWC-7	227	424	1.72
MTWC-8	227	424	1.69

Key Results and Conclusions

For each column, the profiles of species concentrations observed during the static tests were used to:

- 1) estimate the total change in the dissolved masses of total iron, zinc, and sulphate, and
- 2) estimate the mass of each chemical species transferred to the water cover.**

It was found that the initial dissolved masses in Columns 1 and 2 were significantly different from the dissolved masses in Columns 3 through 8, suggesting that the tailings used in these two sets of columns were different in initial composition. The second batch of tailings used in Columns 3 to 8 was obtained from a different location at a later date, and was likely at a different stage of oxidation or of different mineralogical composition.

The transfer of metals to the water cover in Columns 1 and 2 under static conditions was significant and increased the concentrations to above permissible levels. As noted by Yanful and St-Arnaud (1993), the transfer rate of zinc was much lower than that of iron due to an apparent immobilization of zinc in the water cover. In Columns 3 through 6, containing sand and peat covers, no significant transfer of metals to the water cover was observed, except for zinc in the peat-covered columns. However, this zinc could have originated from the peat itself, which initially contained more zinc than what was transferred to the water cover.

During the kinetic phase of each column test, effluent concentrations of total iron, ferric iron, zinc, and sulphate were obtained. The effluent concentration profiles were modelled using the program POLLUTE (Rowe and Booker, 1993) to solve the following dispersion-advection equation:

$$(5-1) \quad n \frac{\partial c}{\partial t} = nD \frac{\partial^2 c}{\partial z^2} - nv \frac{\partial c}{\partial z} - \rho K_d - n\lambda c$$

where

n is the porosity,

c is the concentration,

D is the diffusion coefficient of a contaminant through porewater,

v is the seepage velocity,

ρ is the tailings density

K_d is the partitioning coefficient,

and λ is a first-order decay constant, used to model chemical or biological processes removing contaminants from solution.

POLLUTE uses a semi-analytic finite-layer method to solve the above equations, given pertinent boundary condition.

For Columns 1 and 2, effluent iron and zinc concentrations were successfully modelled. Iron concentrations could be modelled with no sorption and no decay. To model zinc, it

was necessary to set λ to a very high value in the oxidized tailings (upper 35cm of tailings). This suggests instantaneous zinc precipitation or co-precipitation. This is not unreasonable, as mass balance calculations later showed that flushing removed only 85 to 87% of the dissolved mass of zinc at the beginning of the kinetic phase. POLLUTE predicted that the effluent concentrations of iron would decrease to a permissible value (< 1 mg/L) after 6.2 pore volumes of flow and zinc to permissible value (< 0.5 mg/L) after 2.3 pore volumes.

The model could not adequately describe the tail end of the sulphate concentration profiles. While the model predicted that sulphate concentrations would decrease to zero, the measured concentrations only decreased to a constant nonzero value, which may reflect equilibrium between mass removal by flushing and mineral dissolution. POLLUTE does not model mineral dissolution.

Steady-state concentrations were observed for iron, zinc, and sulphate in Columns 3 to 8. Two explanations may explain the difference in behaviour between Columns 1 and 2, and Columns 3 through 8. Firstly, Columns 3 to 8 contained different types and quantities of secondary minerals. Secondly, the use of sand and peat layers may limit initial mineral dissolution during the static test since ion solubility limits are reached more quickly if ions are unable to diffuse into the water cover.

It was shown that the rate of precipitation/dissolution of iron in the sand, peat, and uncovered columns varied with flow rate (Fig. 5-2). The dissolution rate was calculated by comparing the total dissolved mass of iron, the sum of the masses of iron removed in the effluent and iron present in the pore water, at different times. Varying the flow rate would produce different effluent concentrations at the same number of pore volumes.

The total mass of sulphate, iron, and zinc removed by flushing was highest in Columns 5 and 6 (peat layer), and similar in the sand protected and uncovered columns (Fig. 5-3). A comparison of the final effluent concentrations in Fig 5-4 shows the concentrations of total iron and sulphate from Columns 5 and 6 (peat layer) are twice as high as those from

Columns 7 and 8 (uncovered) and six times as large in columns 3 and 4 (sand layer). This shows that, with time, the peat columns will generate the highest concentration of contaminants, the uncovered columns will generate the second highest concentration, and the sand-covered columns the third highest. The large differences in initial pore water chemistry between Columns 1 and 2 and Columns 3 to 8 make comparisons difficult.

The same ratios of iron to sulphate are observed in the effluent concentrations in the sand, peat, and uncovered columns, showing that the same proportions of minerals were dissolving in each column. The higher rate of contaminant release in the peat columns cannot be attributed to high flow rate and/or pH; it may be caused by enhanced mineral dissolution by organic acids. The high steady-state effluent concentrations of the uncovered tailings may be indirectly caused by the oxidation of ferrous to ferric iron at the surface of the tailings and the subsequent precipitation of ferric hydroxides. This could be inferred from the formation of red-orange precipitates at the surface and in the oxidized tailings. The drop in pH caused by this reaction will further change the equilibrium between dissolution and flushing. Calculations based on oxygen supply show that additional pyrite oxidation could not account for the difference in equilibrium concentrations between the sand-covered and saturated tailings.

Redox potential, pH and dissolved oxygen (DO) profiles were measured at the end of the static test. They showed that the use of the peat layer lowered the concentration of oxygen in the tailings (1-2 mg/L) below the concentration measured in the sand-covered tailings (2-3 mg/L). The data also showed that the DO concentrations in the tailings in Columns 7 and 8 were similar to the concentrations in the tailings in Columns 1 and 2 (2-3 mg/L). This similarity could be attributed to oxygen consumption through the oxidation of ferrous to ferric iron, and probably the reduction of oxygen diffusion by surface ferric oxyhydroxide deposits. The results also show that the pH in the water cover in Columns 1 and 2 (3.5) was lower than in the tailings (5.5). The use of interception layers kept the pH of the water covers at neutral levels.

At the conclusion of the tests, final pore water concentrations were obtained from Column 1, before dismantling. Tailings were squeezed for pore water extraction and were also sequentially leached with water and acid (pH 3.5). During the kinetic tests, 13 % of the mass of soluble zinc initially present was not removed by flushing. This mass was, however, removed by squeezing and leaching operations, suggesting that zinc was removed during the experiment by the formation of meta-stable solids.

Implications

The use of sand as a protective layer greatly improves the quality of the water cover. Peat, on the other hand, appears to increase the rate and amount of metal leaching.

It is difficult to compare the water covered tailings with the saturated tailings because of the difference in their initial composition. However, since the oxidation of ferrous to ferric iron and the subsequent precipitation of ferric hydroxide decreases pH and increases effluent metal concentrations, the use of a water cover would be better than simply saturating the tailings, as visual evidence suggests less ferric hydroxide precipitation occurs.

5.1.1.5. Field Test Cell

The test cell, constructed on the Mattabi tailings impoundment, is a 70 x 70-m square area surrounded by an embankment, 1.9 m on the north side and 3.8 m on the south side. The cell was constructed in October 1991 and 20 to 50 cm of precipitation accumulated in it by July 1992. The tailings used in the column tests were taken from the same tailings impoundment on which the test cell was constructed. Ferric oxyhydroxides were also observed to form in the test cell. Zinc and iron concentrations in the water cover were respectively 50 and 10 times lower than in the laboratory columns, but sulphate concentrations were similar. The lower zinc and iron concentrations may be explained by 1) downward pore water flow, estimated to be 2 cm per year, and 2) coprecipitation of zinc with iron oxyhydroxides, which was shown by analysis of the precipitate.

5.1.1.6. Summary

The foregoing evidence from laboratory and field experiments indicate re-dissolution of soluble phases upon flooding and the subsequent diffusion of metals to the water cover may necessitate long-term treatment of seepage and effluent from tailings ponds. The column experiments show that the use of protective layers at the tailings/water interface greatly improves the quality of the water cover. With respect to flushing, the use of a peat layer apparently increases the quantity and rate of metal leaching, so sand is the preferred protection layer. It is difficult to compare the water-covered tailings and the saturated tailings because of the difference in initial composition of the tailings used in each column. However, if the oxidation of ferrous to ferric iron and subsequent precipitation of ferric hydroxide does indeed increase the steady-state effluent concentrations, then the use of a water cover is preferred since visual evidence shows that less ferric hydroxide precipitation occurs.

It should be noted that the NTC study did not simulate wind-derived wave activity or the gradual burial of the tailings by organic sediments. Though the kinetic phase of the column experiments is useful for assessing the rate of metal leaching in the tailings, the magnitude of the flow used is unrealistic. Many tailings ponds may also be constructed on top of alkaline soils, where the mobility of metals migrating out of the tailings would be considerably reduced. Thus, the detrimental effects of using a peat interception layer may have been exaggerated in the NTC study.

5.1.2. Biologically Supported Water Covers

The objective of a BSWC (biologically supported water cover), consisting of plants transplanted on the surface of tailings, is to promote the buildup of organic sediments. The organic layer would tend to stop oxygen penetration into the tailings and minimize metal transfer to the water cover through adsorption and mineral precipitation caused by reducing conditions and microbiological activity. The use of plants on the surface of the tailings may promote natural treatment of water covers by biosorption, microbiological reduction and organic complexation of metals, as well as prevent tailings resuspension.

5.1.2.1. Laboratory Tests

NTC also conducted in-house laboratory tests to determine the oxygen-consuming and metal-absorbing capacities of different organic systems. Plexiglas reactors, 5 L each, were converted to diffusion cells by adding a PVC screen across each reactor, above which was placed 5 cm of the organic material. The organic material was either 1) a sand as a control, or 2) a mixture of algae and duckweed, or 3) a mixture of algae and duckweed inoculated with sulphate-reducing bacteria. The reactors were flooded with water.

In the oxygen consumption tests, water was added to the top of the reactor and was withdrawn from a port 1 cm above the base, giving a flow rate of 0.05 mL/min. While tests using organic material showed dissolved oxygen (DO) concentrations decreasing to zero after 1 pore volume, concentrations observed in the sand test (the control) fluctuated between 4 and 7 mg/L.

The metal flux tests involved placing a solution containing 50 mg/L of zinc below the PVC screen and stirring continuously. Zinc concentrations above the sand and organic material were measured with time. The concentrations measured above the sand layer were found to be significantly higher than those measured above the organic layer (5 to 10 mg/L for sand and less than 1 mg/L for the organic material).

Plant growth tests were also conducted in the laboratory to determine the effectiveness of plant transplantation on Waite Amulet oxidized tailings, using the following substrates:

- 1) Soil as a control
- 2) Tailings as a control for water balance
- 3) Organic test, 20 cm of tailings with 10 cm of soil on top
- 4) Inorganic test, tailings mixed with limestone.

Each substrate was 30 cm thick and had 30 cm of water cover. Before planting, all tanks were neutralized to pH 7 and amended with fertilizer. Lighting was maintained for 10-14 hr/day.

The plants tested were water plantain, cordate pickeralweed, burweed coontail, and water lily. Cordate pickeralweed showed no growth in any of the tests, including the controls, showing that the right conditions for growth of this plant were not present. All plants grew better in the tailings-soil mix than in the tailings-limestone mix.

Analysis of the water cover quality showed similar trends in soil, organic, and inorganic substrate tests for zinc and aluminum. Zinc concentrations decreased to 0.03 mg/L and aluminum concentrations decreased to 0.5 mg/L. Initial sulphate concentrations varied significantly between the tests: sulphate decreased continuously in the soil and organic tests, but remained the same in the inorganic and tailings tests. Total iron concentrations dropped below 1 mg/L in the soil and inorganic tests, while the organic tests showed a drop to below 0.1 mg/L. All tests, except those involving the tailings, showed a declining trend in iron concentrations.

The soil, organic, and inorganic tests all showed higher concentrations of TOC (total organic carbon) and TP (total phosphorus) than the tailings test. TOC in the soil, inorganic and organic tests constantly decreased, while the tailings test showed an increase in the first half of the test, and then a subsequent decrease. TP showed a similar behaviour.

Total Kjeldahl nitrogen (TKN) behaved unusually in the inorganic tests. It initially decreased to very low concentrations (< 0.5 mg/L) and then increased to higher concentrations in the soil and organic tests six months later.

A model was developed to determine the depth of organic material necessary to consume all oxygen above the tailings and the time needed to accumulate this depth. Assuming that oxygen transport is by diffusion only, a net buildup of 4.7 mm of organic sediment is required, which will accumulate within 3 to 5 years after the establishment of a BSWC.

Field Trials

Field tests were carried out at the Heath Steele tailings pond to examine the feasibility of a BSWC. Plant species for transplantation were selected from the nearest alkaline lake. Preliminary field trials, where native species were planted in small test plots, were successful: after 1 month, the density of plant species observed in the test plots was similar to the species density in the donor lake. Therefore, the implementation of full-scale field testing went ahead. The goal of the tests was to assess the effects of different parameters on plant growth. These included:

- 1) Wind and wave action (two separate sites, where wind conditions were different, were used),
- 2) Type of sediment (the tailings pond contained two types of sediment, sludge and tailings),
- 3) Water depth (three depths of 1, 3 and 5 m)
- 4) Plant species (pondweed and milfoil), and
- 5) Plot shape (square or rectangular).

The two sites were located at the East and West sides of the pond. Each site included a group of test plots. Each plot tested a different combination of the above parameters and enclosed eight square or rectangular “sandwiches”, as shown in Fig. 5-5. A sandwich was constructed by enclosing plants and substrates in a folded piece of chicken wire. Each sandwich had a constant surface area of 1 m². In all, 22 plots were installed in the Heath Steele pond. One plot was installed in the donor lake as a control.

Preliminary observations in October 1995 showed that the plant conditions in the Heath Steele and control plots were similar. However, rooting outside the sandwiches had not been observed at either location. This may have been due to the time of planting (July - August).

Of the plots planted with pondweed, the East tailings plot with a 1-m water cover had the highest cell density, which was higher than in the control plot. All the other Heath Steele plots had lower densities than the control. Cell density ranged from 64.3 % (East tailings)

to 20.3 % (West Sludge, 1 m water cover) of the density observed in the donor lake. For the milfoil plots, the cell densities ranged from 100% of the donor lake (Control) to 58.4% (East Sludge site). The plots with the 3-m water cover had significantly higher species density than those with 1-m water cover. The East Sludge plot, which was more exposed to wind, suprisingly performed better than the West Sludge plot (Fig. 5-6).

Comparing the water quality of Heath Steele and the donor lake, the pH in Heath Steele was higher than in the donor lake (9-10 compared to 8) as were the concentrations of sulphur, calcium, and sodium. TOC and TP concentrations were similar in both.

5.1.2.2. Summary

The feasibility of a BSWC and its ability to reduce metal flux from tailings into a water cover were demonstrated in the laboratory. Preliminary results from the field trials suggest that a water cover depth greater than 1 m will improve plant growth. Further research is needed to determine the reproducibility of this result and the impact of the specific effects of greater water depth, such as decreased water velocity at the bottom and decreased oxygen transport, on the performance of a BSWC.

5.1.2.3. Site Modelling

NTC modelled two tailings sites, Mattabi and Heath Steele, using WATAIL, a computer program developed at the University of Waterloo for comparing site decommissioning options. The WATAIL program is described in Section 5.9.

Modelling results indicated that additional flooding of the Mattabi tailings would only marginally improve the effluent quality even over a period of 100 to 200 years, since approximately 50% of the site is already flooded and the depth of the unsaturated zone of tailings is small (less than 0.6 metres). The effect of flooding the tailings at earlier times was also examined: the earlier flooding occurred, the lower the mass of oxidation products, and the greater the improvement in effluent quality. The longer flooding was delayed, the less advantageous it became.

Three options were examined for the Heath Steele tailings:

- 1) no flooding and no lime addition
- 2) no flooding with lime addition
- 3) flooding with lime addition

Predictions show that in Option 1, the reactive minerals near the surface are depleted in 20 years and acid production stabilizes (limited by the increase in the oxygen transport path), giving a slightly acidic pH (~ 6). In Option 2 the pH initially rises, but drops to the same value as in Option 1 with time. For Option 3, pH initially drops to 5.5 in 25 years, but rises rapidly and stabilizes at 8.5.

The above results show that the only advantage of lime addition without flooding is to delay the onset of acid (low pH) conditions. The use of a water cover was found to greatly improve water quality.

5.1.2.4. Summary

The modelling showed that in general, the effect of flooding is dependent on specific site conditions and the time of flooding. However, as discussed in Section 5.9, the WATAIL model is limited and difficult to verify either experimentally or in the field.

5.1.3. Wind

Work done by NTC on wind includes a literature review on wind effects on oxygen transport and particle resuspension and an estimate of the amount of resuspension for typical wind speeds in the province of Québec. The work is based on the method of Rodney and Stefan (1987), described in Section 2.2.4. The estimate shows that, for a 1 m tailings pond and a wind speed of 3 m/s (Québec average), tailings particles of density 2600 kg/m³ and less than 30 µm in diameter will go into suspension. This implies, for example, that 60% of typical Heath Steele tailings would be resuspended in a 1-m water cover and 30% in a 5-m water cover. Although these estimates are conservative, since they ignore tailings cohesion, they seem to indicate that resuspension could be a significant problem. The model formulation ignores the contribution of return currents to particle resuspension. Return currents may increase the water particle velocity at the bed.

An investigation of bubble entrainment showed that it is not a plausible source of oxygen transportation, as the large wave amplitudes needed to transport bubbles to the bottom of the water cover, would rarely occur and only for short periods of time.

The effect of wind-induced turbulence on acid production in unoxidized tailings under water cover was estimated using Morin's method (Section 3.1). Published equations for oxygen transport by turbulence were used to estimate a vertical turbulent mixing coefficient K that replaces the molecular diffusion coefficient in Morin's equations (Bank and Herrera, 1977). From correlation work on natural streams, K is calculated as:

$$(5-2) \quad K = \frac{K_L D}{a_0}$$

where K_L is air-water dissolved oxygen transfer coefficient, D is the water depth, and a_0 is an empirical constant, with a value of 0.053 for the above study. K_L is calculated by:

$$(5-3) \quad K_L = K_0 - \frac{K_o K_r}{0.0246} + K_r$$

where

K_0 is the wind oxygen transfer coefficient:

$$(5-4) \quad K_0 = 10^{-6}(8.43 U^{1/2} - 3.67 U + 0.43 U^2)$$

and K_r is the oxygen transfer coefficient due to rain given by:

$$(5-5) \quad K_r = (6.76 \times 10^{-5} + 2.91 \times 10^{-9} z)r^{1.26}$$

Acid production was calculated for a hypothetical unoxidized waste rock pile for different water depths using the turbulent coefficient K as calculated above, and by assuming oxygen transport by molecular diffusion alone using an average pond oxygen concentration. The predicted acid production was found to increase with increasing wind

speed and pond depth. Acid production estimates that considered turbulent oxygen transport were found to be up to 30% greater than those based on molecular diffusion alone. However, when the reactive minerals at the surface are exhausted, the oxygen supply for acid production at greater depth will be controlled by molecular diffusion through the oxidized layer. It follows that when using a protective cover over tailings, such as sand, the transport of oxygen to the tailings will be dominated by molecular diffusion.

Predictions using the formulations of Bank (1975) and Bank and Herrera(1977) show that turbulent diffusion processes will cause oxygen saturation in large bodies of water, regardless of depth, if no reaction is present to consume oxygen.

5.1.3.1. Summary

The prediction of resuspension suggests it could pose a significant problem in tailings ponds. The significance of the 30% increase in oxygen transport due to turbulence may, however, be questionable as the resulting oxidation rate is still extremely low when compared to the oxidation rate in uncovered tailings.

5.2. SOLBEC TAILINGS FIELD TRIALS

Field experiments were undertaken by the Centre de recherches minérales (CRM), Quebec Ministry of Natural Resources, Canada to investigate the impact of tailings flooding at the Solbec site near Solbec Cupra, Québec. There were four phases in this project. In the first phase, six columns were assembled in the laboratory and three cylindrical plastic basins were constructed in the field, each with one piezometer. This phase involved maintaining a water level of 1 m over each column and basin, and periodically sampling the water cover for the measurement of pH, temperature, and redox potential. In the second phase, the above-mentioned field program was extended to include the installation of a new 3 x 3-m basin with two piezometers. For the third phase, a new cylindrical basin was constructed, with ten piezometers in place, to sample both surface and pore waters. Water samples were also analyzed for conductivity and dissolved oxygen (DO). The purpose of the fourth phase was to evaluate the effects of

lime addition to previously oxidized tailings and measure the neutralizing effects in the underlying layers. A new basin with eight piezometers was constructed for this purpose. The new basin was installed in an area where the tailings had not been disturbed in the previous phases. The basin was driven into the tailings to a depth of 1.27 m, and had a height of 1.35 m above the tailings. The amount of limestone added was estimated based on the sulphur content of the tailings. The limestone was mixed with the upper 15 cm of oxidized tailings.

The stratigraphy of the Solbec site consists of a surface layer of oxidized tailings, of yellowish colour and varying thickness due to expansion cracks caused by surface drying of the tailings, a layer of unoxidized tailings, dark to steel grey in colour and approximately 95 cm thick, resting on top of a 25 cm thick layer of peat, overlying the local till.

Water for all the basins was supplied from a large plastic tank that was replenished periodically with tap water. Three control piezometers were installed in the tailings outside the experimental basins. Starting in summer of 1992 and ending in fall 1993, sampling was bimonthly and included analyses for water level, electrical conductivity, redox, pH, dissolved oxygen, and iron.

The water cover sampling results are presented below.

pH

In all basins, the pH ranged from neutral to slightly alkaline (~ 8). The pH was slightly acidic (6-7) at the beginning of each season. In the 1991 and 1992 basins, the pH followed trends for the supply water whose pH varied between 7 and 10 in 1993.

Redox

In all basins, the redox correlated highly with the inflow water varying from 0-100 mV in 1990 and 1991 to 300- 400 in 1992 and 100 -300 in 1993.

Electrical Conductivity

This parameter also had similar ranges (0.10 -0.40 mhos/cm) in the cover water and supply water.

Dissolved Oxygen

Ranged from 6-17 mg/L, in both the supply and cover water.

Iron

Always less than 0.5 mg/L, compared to measurements of 30 mg/L from the tailings effluent.

Alkalinity

Variations corresponded with those in the supply water.

Hardness

Values were slightly above the supply water value. The hardness of the water cover ranged from 55 to 186 mg/L CaCO₃ ; the measured hardness of the supply water was between 50 and 96 mg/L CaCO₃.

Calcium, Potassium, and Magnesium

The concentrations of these species were always low in both the water cover and supply water. Calcium concentrations in the supply water and in most of the basins ranged from 10 to 40 mg/L. In the 1992 basin the calcium concentrations were higher and up to 65 mg/L.

Sulphate

Sulphate concentrations in the water cover were up to 10 times those of the supply water, as shown in Table 5-3.

**Table 5-3: Sulphate Concentrations in Basins and Supply Water, Solbec-Cupra
(mg/L)**

Sampling Date	Basin 89-1	Basin 89-2	Basin 89-3	Basin 91	Basin 92	Supply water
28/10/92	23	32	17	27	49	n.m.
06/7/93	26	47	44	7.2	78	8.2
03/8/93	16	16	21	36	50	6.7
30/9/93	13	14	19	30	30	8.4

Lead and zinc concentrations in the cover water were slightly higher than in the supply water, but still low. Other metals including Cu, Cr, Hg, As, and Ni were all at low concentrations in both the cover and supply water.

Generally, high metal concentrations and low pH values were measured at the beginning of each season, when the water cover had just been established. With the exception of the low pH measurements observed during spring, all measured parameters conformed to effluent discharge standards.

The results of pore water sampling are presented below:

pH

In the 1989 basins, the pH in the oxidized tailings pore water rose from approximately 3 to 6 in three years. The pH of the oxidized tailings in the 1992 basin rose from 5 to 7 within the first year. The low pH values in the tailings, measured from piezometers within and outside the basins, steadily increased with depth to higher than 8, showing that the deeper tailings have a large buffering capacity.

Redox

In the 1989 basins the redox potential in the oxidized tailings rose from 0 to 300 mV by 1993. In the 1992 basin, the redox potential has, to date, varied between 100 and

200 mV. The alkalinity measured in the oxidized tailings mixed with lime was comparable to that of the unoxidized tailings and was significantly higher than in the undisturbed oxidized tailings, as expected. Sulphate concentrations in the oxidized tailings pore water in the 1992 basins were up to 2900 mg/L and were approximately four times as high as those measured in the 1989 basins. Sulphate concentrations at lower depths beneath the basin were similar to those observed in the unoxidized tailings. The concentrations were, however, considerably lower (less than 100 mg/L) outside the basins.

Calcium and magnesium in the oxidized tailings were higher in the 1992 basin than in the other basins. The heavy metals with significant concentrations were copper, zinc, and iron. The 1992 basin reported the lowest copper and iron concentrations and the highest zinc concentration (24.4 mg/L). Heavy metal concentrations in the tailings outside the basin were generally much lower than below the basins.

5.2.1. Summary

Many measured parameters in the water cover and supply water were similar. The small scale of the experiment probably caused this. In a large tailings pond, these parameters could be expected to change significantly from inlet to outlet. The downward flow within the basin, caused by the difference in hydraulic head between the basin and the rest of the tailings, would tend to flush chemical species downwards and improve the water quality of the cover and the upper tailings and worsen the quality at lower levels. Another shortcoming of the study is that it does not simulate wind/wave action present in a real tailings pond.

The project report suggests the high sulphate concentrations observed in both the water cover and in the oxidized tailings, especially in the “neutralized” basin, can be explained by the dissolution of gypsum, and the diffusion of sulphate from the pore water to the water cover. Using the supplied water quality data, we performed MINTEQ analysis to

show that the pore water is saturated with gypsum. The addition of lime has a definite effect in reducing the solubility of metals, except zinc. As noted in the report, the solubility of zinc remains high at high pH values.

Although the report concluded that flooding prevents significant further oxidation of the unoxidized tailings, it is clear from the results that additional measures, such as lime addition, must be implemented to limit secondary mineral dissolution in the oxidized tailings at the Solbec site.

5.3. CANMET ELLIOT LAKE LABORATORY

The CANMET Laboratory at Elliot Lake conducted three different types of experiments on flooded tailings: lysimeter tests and two sets of column tests.

5.3.1. Lysimeter Tests

In these experiments, pyritic uranium tailings were placed in a Plexiglas tank 1.22 m long, 0.3 m wide, and 0.61 m high. The tailings were placed on a bed of coarse polyethylene beads, to ensure homogeneous vertical flow through the tailings when water samples were withdrawn from the bottom of the tank. Two mixing chambers were set up in either end of the tank to homogenize the water cover. Water was passed through holed screens to establish laminar flow. Sampling ports were installed at 0.025, 0.15 and 0.275 m below the surface of the tailings.

Five tanks were in place at the time of this review. The first tank was constructed in 1983. Davé and Lim (1983) present results observed during 300 days of operation. Flow variations (0-300 mL/min) and column height (0.055-0.178 m) did not produce significant changes in sulphate flux from the tailings (92.9 to 98.4 mg/m²/hour). The flux, J_s , was calculated from the measured, steady-state sulphate concentration in the water cover:

$$(5-4) \quad J_s = CQ / \text{AREA}$$

where C is the steady state concentration in the water cover, mg/L

Q is the flow, L/s

and AREA is the cross-sectional area in m² of the tailings surface at the interface.

During the first 300 days, all fluxes including those of radium, calcium and total dissolved solids decreased rapidly. Dissolution of soluble minerals and oxidation products at the surface of the tailings resulted in significant mass flux transport compared to diffusion from the tailings below. At the time of this review the tank had been practically overgrown with algae and had developed a visible oxidation zone 2-4 cm in thickness. Personal communication with T.P. Lim indicates bacteria are reducing sulphate to sulphide as H₂S. Iron is then reprecipitated as pyrite. Very small quantities of iron-oxidizing bacteria are present near the tailings surface, and ferrous ion oxidation appears to be in equilibrium with ferric ion hydrolysis. Deposition of organic sediments in the tailings substrate has partially reduced the previously established anoxia.

In 1990, a second tank was constructed that contained previously oxidized tailings. Here, a significant accumulation of organic sediments was observed, along with a visible brownish-orange (characteristic colour of iron oxyhydroxide precipitates) oxidation zone less than 1 cm thick. Three other lysimeter tests were started in 1994, one with fine tailings, the other coarse tailings, while the third contained tailings washed with water to remove gypsum. While green algae is growing in the first two tanks, the third tank shows only slight growth of black organic sediments, as the washing process may have removed all nutrients. The oxidation zone is thickest at the top of the coarse tailings and very thin at the top of the washed tailings - all zones were less than 1 cm in thickness.

5.3.2. Leaching Column Experiments

The column experiments described by Davé and Vivjurka (1994) are continuing. The results indicate a water cover reduces acid drainage in coarse tailings by a factor of 3000.

5.3.3. MEND/INCO Water Cover Experiments

New column experiments simulating water covers over a variety of sulphide-bearing materials, were commenced in 1995. The columns are 1.2 m high and 0.3 x 0.3 m in area. A total of 10 sets consisting of 30 columns have been installed, with each set containing a variety of tailings, waste rock and cover materials such as municipal compost and low sulphur tailings, as shown in Table 5-4.

Table 5-4: Test Material and Water Cover Characteristics of New Column Experiments, CANMET Elliot Lake Laboratory

Test Materials	Water Cover
1 m oxidized waste rock from Mine Doyon	0.5 m and 1 m stagnant, 1 m with mixing, 1 m with porewater flow
1 m oxidized tailings	0.5 m and 1 m stagnant, 1 m with mixing, 1 m with porewater flow
1 m unoxidized tailings	0.5m and 1 m stagnant, 1 m with mixing, 1 m with porewater flow
0.5 m oxidized tailings bottom, 0.5 m low-sulphide tailings top	0.5m and 1 m stagnant, 1 m with mixing, 1 m with porewater flow
0.5 m oxidized tailings bottom, 0.5 m compost top	0.5m and 1 m stagnant, 1 m with mixing, 1 m with porewater flow
0.7 m pyrrhotite tailings bottom, 0.3 m low-sulphide tailings top	0.5m and 1 m stagnant, 1 m with mixing, 1 m with porewater flow

0.35 m compost on top of 0.35 m oxidized tailings on top of 0.35 m unoxidized tailings	0.5m and 1 m stagnant, 1 m with mixing, 1 m with porewater flow
0.5 m unoxidized tailings on top of 0.5 m oxidized tailings	0.5m and 1 m stagnant, 1 m with mixing, 1 m with porewater flow
0.35 m low-sulphide tailings on top of 0.35 m oxidized tailings on top of 0.35 m unoxidized tailings	0.5m and 1 m stagnant, 1 m with mixing, 1 m with porewater flow
1 m unoxidized tailings (fine grained)	0.5m and 1 m stagnant, 1 m with mixing, 1 m with porewater flow

The experiments are run for at least three months at a time with various depths of water cover under stagnant or mixed conditions. A pump mixes the water cover from bottom to top. Future experiments will have pore water flow; presently none do. All tests use untreated lake water to eliminate excess alkalinity and free chlorine that can influence microbial activity.

Water is sampled at specific intervals across the water column and tailings profile using suction. Samples are taken from top to bottom to avoid sample displacement by drawdown. The samples are analyzed for dissolved oxygen (DO), metals, acidity, alkalinity, redox, and pH.

Early results of oxygen profiles, after two months of testing, are reported here. In the stagnant water columns, a diffusion/consumption profile developed near the tailings-water cover interface. The DO concentrations in the water column declined from approximately 10 to 7 mg/L, possibly due to a temperature change and/or consumption of

oxygen by organic matter throughout the water column. The DO concentrations in the tailings declined at the tailings-water interface to less than 0.5 mg/L. In well mixed water covers, DO concentrations were nearly constant, with a very steep drop at the tailings-water interface. The DO concentration in the 1-m covers was generally about 0.5 mg/L higher than in the 0.5 m water cover.

5.3.4. Summary

The CANMET research lends support to the use of shallow water covers for permanent tailings decommissioning. The establishment of reducing conditions, the apparent halt of the migration of the oxidation zone, and the reduction of metal fluxes, are all promising indications of acceptable water cover performance. The following issues, however, need further investigation:

- 1) The rate of downward migration of the oxidation zone, if any,
- 2) The proper conditions for vegetative growth, and
- 3) Disturbance of the oxidized layer and organic cover as a result of wave-induced motion and resuspension of fine tailings.

5.4. MCGILL GEOTECHNICAL RESEARCH CENTRE (GRC)

The objective of the McGill University GRC study (MEND Project 2.13.2b, 1993) was to predict the height of water necessary to avoid resuspension of tailings from the Solbec site. Experimentation involved placing a 20 cm-thick layer of tailings with light compaction, in a 1.730 x 0.105 x 0.300 m deep Lucite tank. The tank was filled with tap water to a height of 214 mm. Waves were created mechanically by immersing a cylinder to a depth of 25 cm at 2 cycles/s. The water level in the tank was lowered until resuspension of the particles occurred in each test. The tests were repeated with a 13-mm sand layer over the tailings.

For uncovered tailings, particle resuspension occurred at a wave height to water depth ratio of 0.2129. This ratio was employed to determine the necessary water depth at the actual site, by first calculating a critical wave depth from wind speed and fetch length data. The wave depth was calculated using the semi-empirical Sverdup-Munk-

Bretschneider method described in Section 2.2.3. The fetch length used in the calculations was 1 km, which is the length of the whole site, and therefore an overestimate.

There are several possible sources of inaccuracy in this approach. The criterion for resuspension of particles is the maximum water velocity at the surface of the tailings. This is not caused alone by wave action, but also by wind-induced return currents that are especially significant where there are large wind fetches. Additionally, in small tanks, standing waves have a significant effect on the bed velocity, and the absence of a shore results in less attenuation of wave energy than present in a tailings pond (Davé, 1997). Therefore, the phenomena controlling resuspension in a small tank are different from those in a large tailings pond. The correlation between wave height and water depth obtained in the experiments were only for a specific wave height (25 cm) and a specific wave period (0.5 s). In fact, for large bodies of water, the significant wave period is of the order of 10 to 15 seconds. If the wave depth and period are different at the site, the above correlation may not hold. Lastly, the grain size of the tailings used in the experiment were not representative of the Solbec tailings (Vezina, 1997).

5.5. UNITED STATES BUREAU OF MINES

W.W. White (III), formerly of the United States Bureau of Mines (USBM) and E.M. Trujillo and C. Lin at the University of Utah have been developing a three-dimensional geochemical model to predict the generation of AMD by calibrating it to humidity cell test data (USBM 1994,1995,1996). The model successfully predicts the transition from abiotic to biotic pyrite oxidation, jarosite precipitation, and growth of iron-oxidizing bacteria. The model employs shrinking-core theory in its formulation of the sulphide oxidation process (for example, see Davis and Ritchie, 1986).

Of particular interest to the present review are the results of flooded-cell testing of samples of waste-rock, one containing 24% framboidal pyrite (Sample a), and the other 6% euhedral pyrite with coarser particles and, therefore, a smaller surface area

(Sample b). The coarser sample (Sample b) produced more sulphate than the framboidal sample. The depth of the oxidation zone was less than 1 cm in the finer-grained waste rock, but was deeper in the coarser sample. This suggests that the rate of oxidation is controlled by the vertical transport of oxygen in the tailings; the euhedral waste-rock has a higher porosity than the framboidal sample. The geochemical model could not simulate these results, perhaps because shrinking-core theory assumes the vertical ingress of oxygen through the pore spaces into mine wastes is fast compared to oxygen transport within a particle. However, in tailings, vertical ingress of oxygen may control the rate of oxidation, not intraparticle diffusion.

In comparison to standard humidity cell tests, flooding reduced the cumulative sulphate release rate in the waste rock by 80% over a 44-week test period. However, some amount of pyrite oxidation appeared to have occurred in the flooded waste rock, as indicated by small increases in the cumulative sulphate release rate and small decreases in pH.

5.6. ECHO BAY MINE

The US Environmental Protection Agency (EPA) evaluated the potential environmental impact of a proposal to deposit tailings in a man-made reservoir (EPA, 1994). The proposed disposal was part of a plan to reopen the Alaska-Juneau gold mine near Juneau. The plan called for subaqueous disposal of tailings, after flotation and cyanide leach treatment, by a barge-mounted pipe into an artificial impoundment, where the minimum water level would be 6.1 m (20 ft). The impoundment would cover 420 acres and requires the construction of a 105.2-m (345-ft.) dam. Contaminant transport modelling performed by EPA showed that the concentrations of total suspended solids, cyanide, arsenic, and copper in the discharged surface water from the impoundment would likely exceed regulatory standards. The EPA also stated that high levels of metals in the pore water of the tailings would bioaccumulate and likely put wildlife at substantial risk. The plan was eventually rejected.

Water quality modelling of the impoundment was completed using two models, WASP4 and CE-QUAL-W2, both developed by the United States Army Corps of Engineers. The WASP4 model is a two-dimensional time-dependent water quality model that predicts total suspended solids, cyanide, and metal concentrations in a grid. WASP4 models the turbulent mixing processes characteristic of the impoundment, settling of tailings, and leaching and partitioning of metals. The model assumes that when the tailings are first deposited by barge, they are distributed instantaneously over the water-column as a function of mixing-energy. The concentrations of solids and metals in each grid are then calculated from leaching and adsorption rates, settling rates, and mixing in the vertical plane. The pore water concentrations of metals in the tailings or sediments are also calculated, taking into account leaching, adsorption, and molecular diffusion.

Figure 5-7 illustrates the speciation sub-model, where metals are dissolved in a first-order reaction with a constant leaching rate and are adsorbed according to a partitioning coefficient. For the project in question, the leaching rate constant was assumed to be the same for all metals and was taken from iron leaching tests. Partitioning coefficients were obtained from laboratory tests.

Parameters governing mixing energy and settling and resuspension of particles were based on estimates provided by experts, rather than on experimental or analytical methods. The results of modelling were sensitive to variability in these parameters. To estimate the effluent water quality, the amount of resuspension was set to zero, since it was assumed that any significant amount of resuspension occurring in the reservoir would resettle by the time the flow reached the outlet. Settling rates were estimated based on Stokes Law, which is generally not applicable to dense suspensions of flocculated particles. Tailings oxidation was not modelled, since acid- base accounting test results showed that the tailings have enough natural alkalinity to offset acid generation.

The model CE-QUAL-W2 can handle more temporal and spatial variability of hydrological input than the WASP4 model. It was used to predict the total suspended

solids of the effluent from the reservoir. Both these models were also used to examine the water quality of the downstream water bodies.

5.6.1. Summary

Although the models used appear very sophisticated, they have the following deficiencies:

- 1) They rely heavily on non-physically based parameters that require extensive laboratory work to determine.
- 1) Modelling of particle resuspension is based on an arbitrary choice of Stoke's velocity for the settling of a single particle.
- 2) Suspended solids and metal concentrations are the only water quality indicators predicted.
- 3) The geochemical modelling is very simple, with a single first order leaching rate applied to all metals, and a partitioning coefficient obtained by averaging of results from several laboratory tests. The water quality predictions are pH- They rely heavily on non-physically based parameters that require independent.
- 4) Acid generation is not modelled.

5.7. LAVAL UNIVERSITY MICROBIOLOGICAL EXPERIMENTS ON FLOODED TAILINGS

Water cover research at Laval University (Rapport NEDEM 2.13.2c, 1994) includes microbiological experiments on flooded tailings from the Solbec tailings impoundment and estimates of the amount of lime required to neutralize the acidity produced by dissolution of minerals in previously oxidized tailings.

The microbiological study consists of column tests on undisturbed fresh tailings from the site, undisturbed fresh tailings covered with lime, undisturbed fresh tailings covered with crushed limestone, and one column of oxidized tailings with no neutralizing material. The first three columns were covered with 1 m of water, and the last column with 1.5 m of water. The depth of tailings in each column was approximately 50 cm. In the first three

columns, 30 mL samples were taken from 80 cm above the tailings, 10 cm above the tailings, and from the bottom of the tailings twice a month. In the fourth column, 20 mL samples were extracted from five different levels in the water column, and four different depths in the tailings, every two weeks. The water level in each column was refilled after each sampling campaign. The temperature of the columns was maintained between 22 and 25°C.

Bacteria were isolated from samples of oxidized and unoxidized tailings and from the supernatant above the tailings, using various growth media. The cultures were inoculated with liquid samples taken directly from the site, collected from supernatants of the column experiments, and a nutritional solution. There were no acidophilic bacteria in the submerged unoxidized tailings. The surface oxidized tailings were heavily colonized with autotrophic, heterotrophic and acidophilic bacteria. These include *Thiobacillus ferrooxidans*, *Thiobacillus thiooxidans*, *Thiobacillus acidophilus*, and *Thiobacillus thioparus*. *T. ferrooxidans* rapidly oxidizes iron sulphates, though their preferred substrate is pyrite. Each of these groups of bacteria contributes to the oxidation of ferrous iron and sulphur, produced by the chemical oxidation of pyrite by ferric ions.

T. Thioparus bacteria can function at a neutral pH and catalyze or initiate acidification and facilitate the subsequent growth of *T. Ferrooxidans* and *T. Thioxidans* bacteria, which then accelerate pyrite oxidation. These bacteria are all strict aerobes. Since the majority of the metabolic, oxidative consumption of substrates requires the presence of molecular oxygen, all factors limiting oxygen supply influences the viability of the bacteria.

In order to establish suitable conditions for pyrite consumption, there must be adequate oxygen supply, optimal pH, a lack of organic material, and a supply of CO₂ as the only substrate. In addition, heterogenic bacteria must be present to consume organic material toxic to *T. ferrooxidans*, and autotrophs should be present to fix nitrogen.

In fresh tailings columns the water cover remained neutral or alkaline whether or not a neutralizing material was added. Limestone (CaCO_3) addition actually increased the acidity of the flow through the bottom of the tailings.

Standard methods to evaluate the acid-generating potential of soils, including rapid titration, buffer, and acid-exchange methods were used on a variety of tailings samples. These results were compared with 37-week incubation studies, consisting of flooding tailings in small cells (23 cm diameter and 25 cm depth), in which lime was added. It was found that the standard methods could not accurately predict the amount of CaCO_3 needed to neutralize the accumulated acidity in the incubated tailings.

5.7.1. Summary

Using the incubation experiments, it was estimated that the required lime application at the Solbec Cupra site would be 118 tonnes/hectare to a depth of 15 cm. Flooding of unoxidized tailings prevented their colonization by iron-oxidizing bacteria. On the other hand, flooded oxidized tailings remained heavily colonized by these bacteria.

5.8. WATAIL

WATAIL is a two-dimensional model developed by the Centre for Groundwater Research at the University of Waterloo to evaluate the loading of oxidation products in a tailings drainage basin (Scharer et al., 1993). The program predicts sulphate and iron concentrations and pH in groundwater and surface water. Tailings impoundments are simulated as a network of nodes, each node consisting of a homogeneous mass of tailings, characterized by pyrite content, average grain size, depth of deposit, and depth to water table. Both groundwater and surface water flows between the nodes are modelled. Pyrite oxidation is the first process the program models within a time step. The rate of pyrite oxidation is taken to be a function of particle surface area, controlled by the rate of oxygen transport into the tailings. The oxidation is written as:

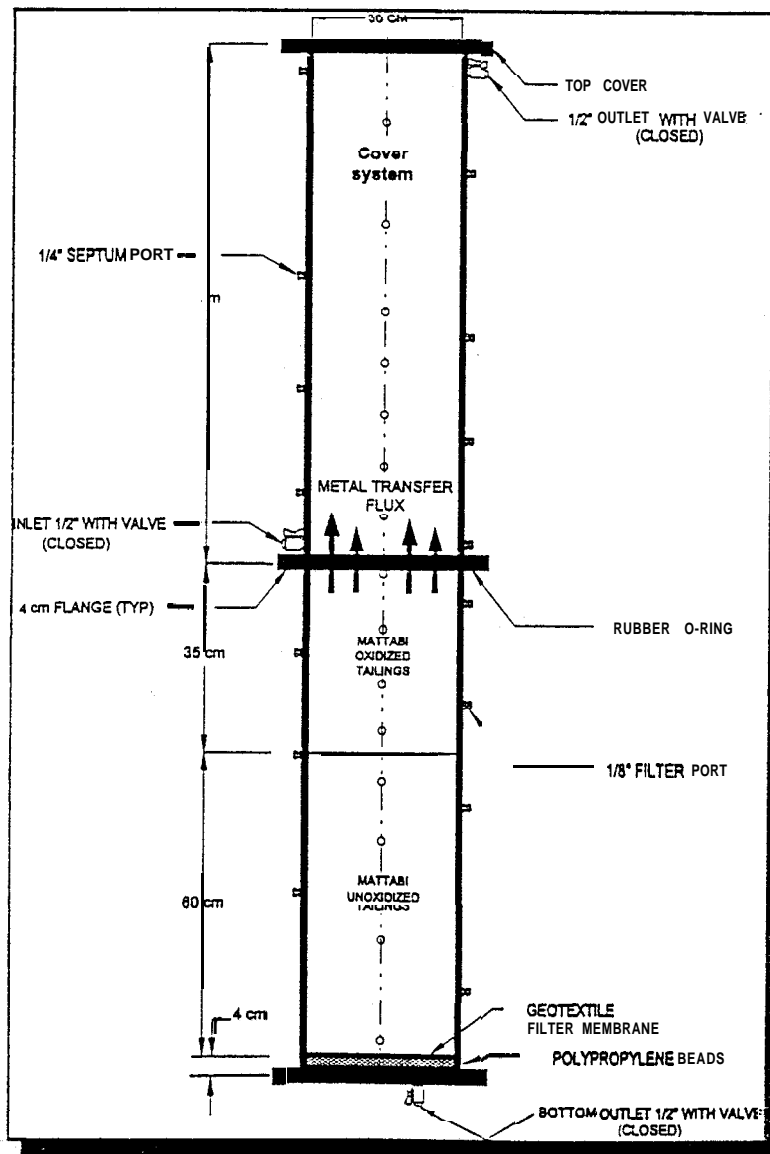
$$(5-5) \quad (K_{\text{biotic}} + K_{\text{abiotic}}) [\text{Total Surface Area}] / [\text{Bulk density of the tailings}]$$

where the abiotic and biotic rates of pyrite oxidation are functions of oxygen concentration. The sulphate production rate is determined using the equation proposed by Scharer (1991) [see Section 3.2]. This equation is essentially a function of the effective diffusion coefficient of oxygen in the tailings; it does not consider the effect of pyrite consumption at the surface, the subsequent increase in the path for oxygen from the tailings surface to unreacted pyrite, and therefore the decrease in the oxidation rate.

The precipitation and dissolution of secondary minerals, such as gypsum, siderite, aluminum hydroxide, ferric hydroxides and jarosite, are modelled with aqueous equilibrium reactions, i.e. these reactions proceed instantaneously to maintain equilibrium, regardless of the actual kinetics of mineral precipitation or dissolution.

Each node has a parameter for depth of water column, for example, the depth of water cover. This parameter affects the volume of water in the node and hence chemical concentrations. A default diffusion coefficient of $3.3 \times 10^{-11} \text{ m}^2/\text{s}$ is assigned for every ponded node.

WATAIL does not consider wind effects on oxygen transport or the effect of particle resuspension on oxidation rate. As a result of the many uncertainties involved in its features, we do not recommend it for quantitative predictions. It may, however, be used as a tool for comparing different management options.



Laboratory column

Figure 5-1

A schematic of a typical instrumented laboratory column used in Matabi tailings experiments at NTC

Dissolution Rate vs Flowrate

.. MTWC-5

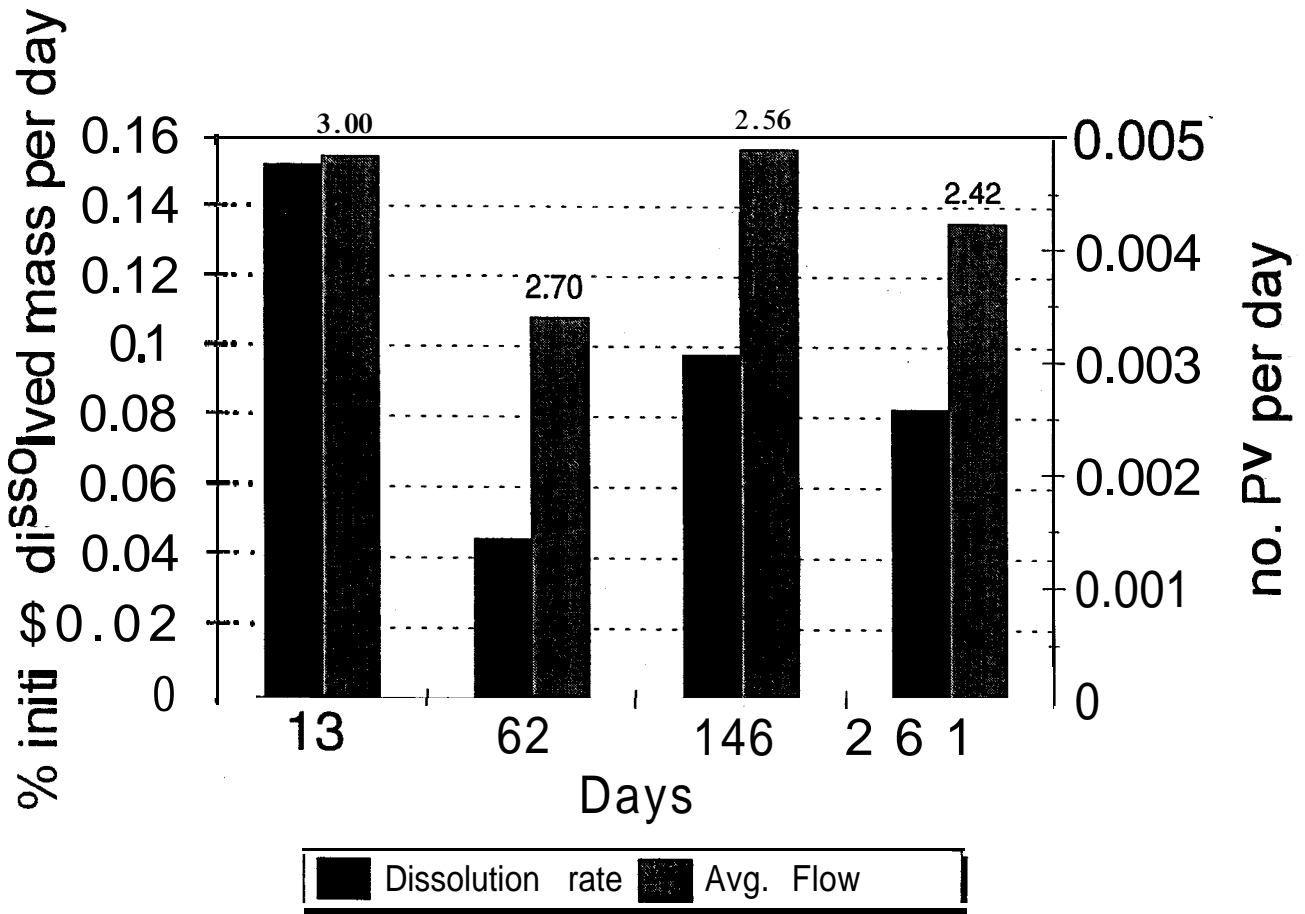


Figure 5-2

Flow rate and solids dissolution rate, as a percentage of initially dissolved mass during kinetic testing of Mattabi tailings

Effluent Loads as % Initially Dissolved Mass

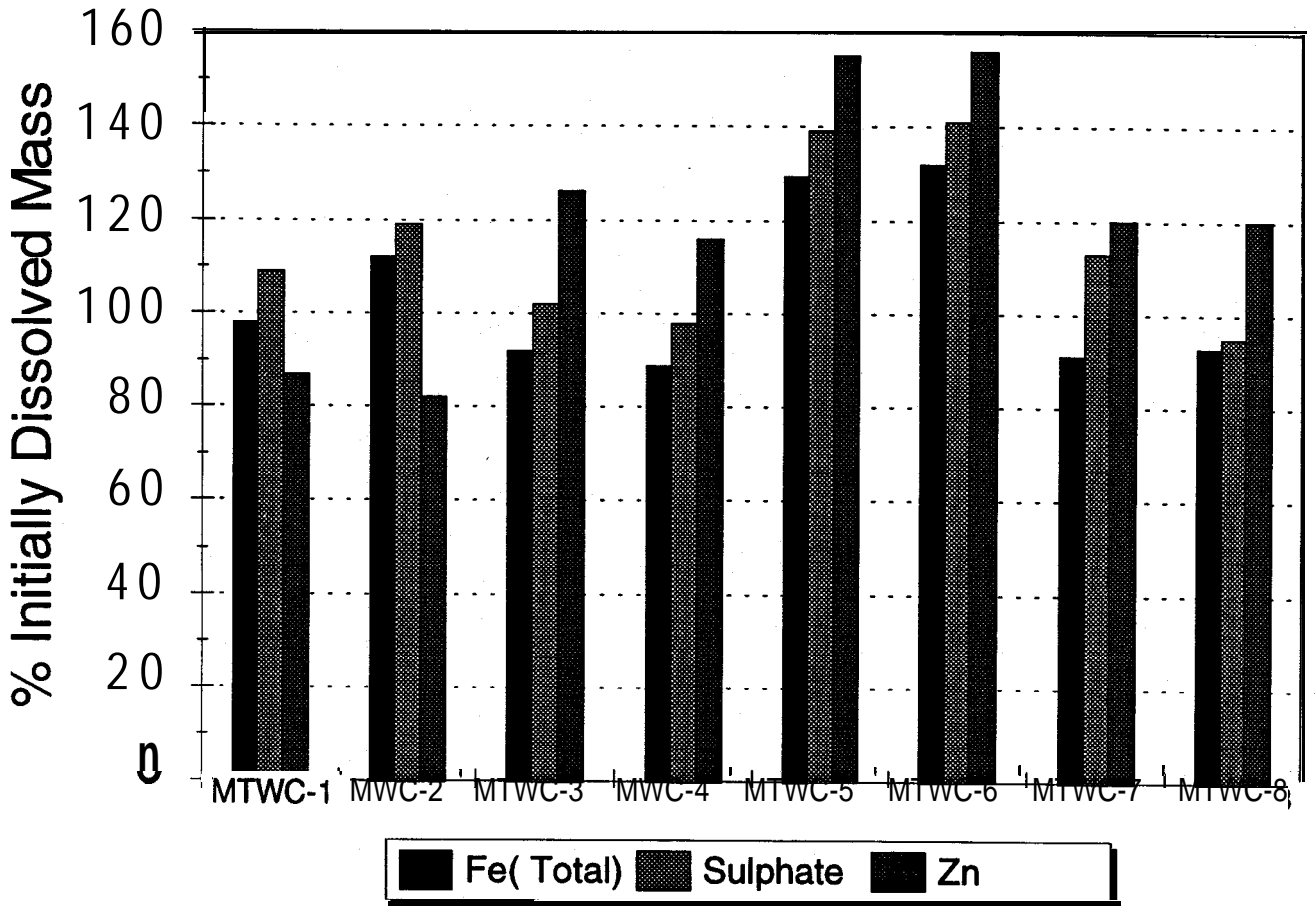


Figure 5-3

Mass of sulphate and metals removed from Mattabi tailings
as a percentage of initially dissolved mass

Steady-State Effluent Concentrations

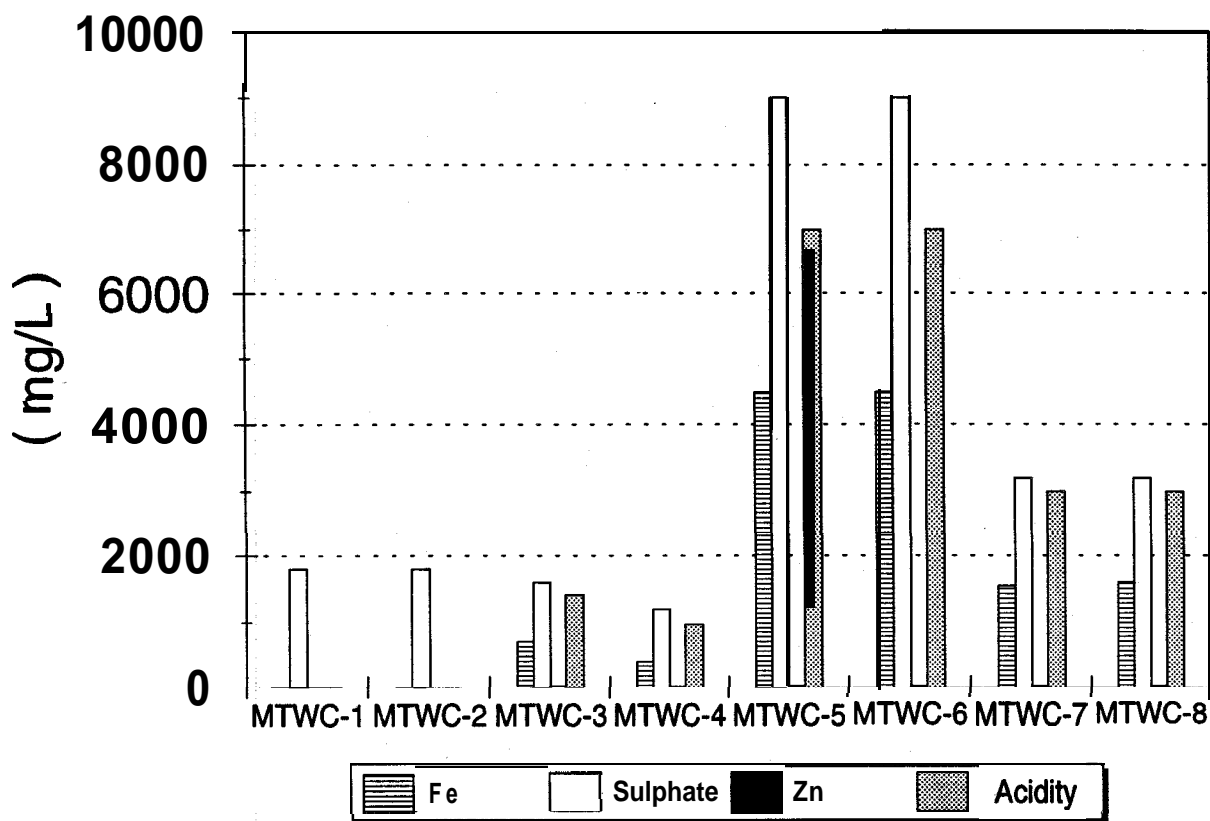
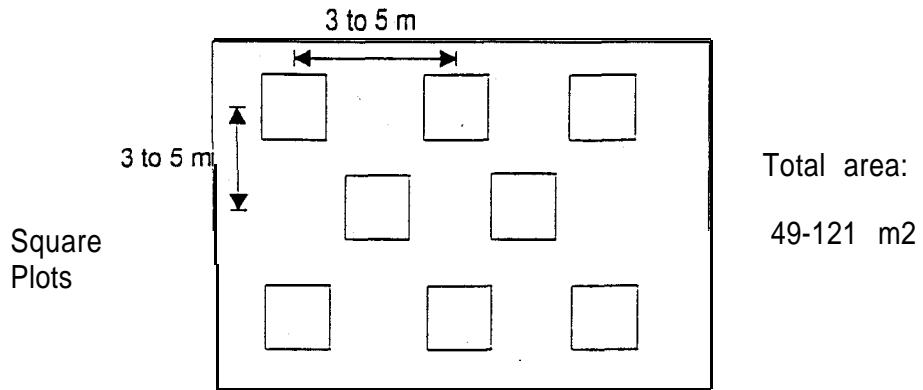


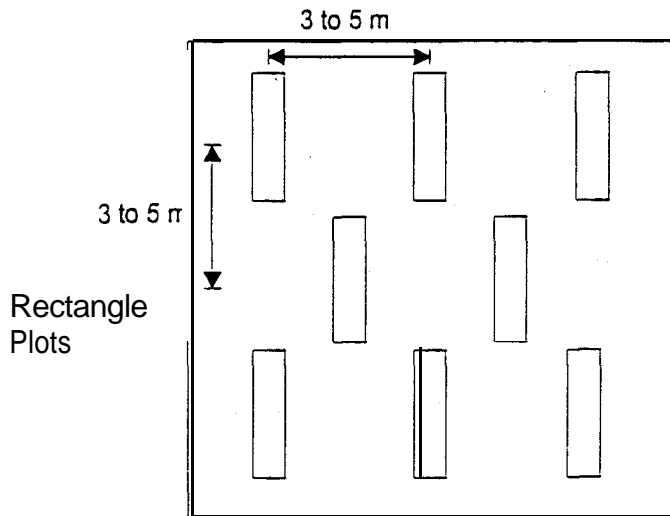
Figure 5-4

Steady-state concentrations of ions in the effluent from NTC Mattabi tailings columns observed at the end of the kinetic test

Schematics of Individual Plots



8 square sandwiches
1 X 1 m



8 rectangular sandwiches
0.33 X 3 m

Figure 5-5

Layout of plots for field trials at establishing a Biologically Supported Water Cover at Heath Steele tailings pond

Periphyton density and Number of **Taxa** for Pondweed and Milfoii at Different Sites

Station	Pondweed			Milfoii		
	Density.		Number of taxa	Density		Number of taxa
	cells/mL	% of Tet. Lake		cells/mL	% of Tet. Lake	
Tetagouche Lake	1077		17	705		14
Control	648	60.2	16	711	100.9	17
ET	693	64.3	16	-		
ES	639	59.3	13	412	58.4	14
WT	612	56.8	17	442	62.7	12
WS1	219	20.3	12	488	69.2	14
WS3	543	50.4	12	592	84.0	15
WS5	537	49.9	16			

- ET: East site, in tailings
- ES: East site, in sludge
- WT: West site, in tailings
- WS 1: West site, in sludge at 1 m depth
- WS3: West site, in sludge at 3 m depth
- WS5: West site, in sludge at 5 m depth

Figure 5-6

Growth data on plant colonization attempts at Heath Steele and the donor lake

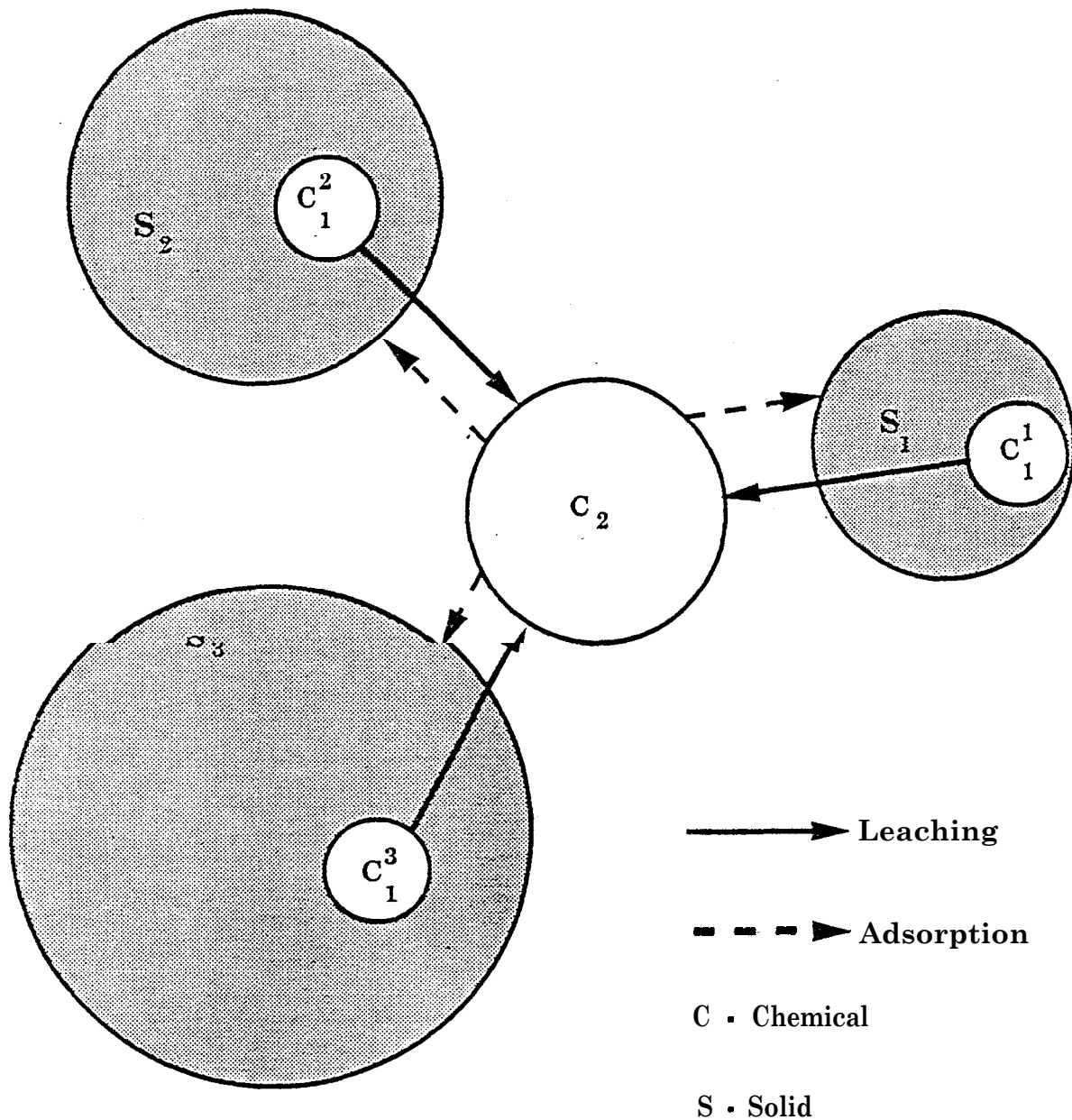


Figure 5-7

Conceptual model of geochemical processes for metals in the proposed Alaska-Juneau Mine tailings pond

6. OVERALL SUMMARY

6.1. DESIGN AND PERFORMANCE OF FLOODED TAILINGS SITES

The most important design parameter for a water cover is the depth of water, which will have a significant impact on cost. For most of the sites reviewed, the criteria used to determine the required depth of water cover were:

- 1) Maintaining moisture saturation of the tailings in the event of a drought
- 2) Avoiding tailings resuspension due to wind

Not all the sites were, however, designed or constructed using the above criteria. In some cases there is no mention of how the depth of water was selected.

The resuspension calculations examined in the review made many assumptions regarding the nature of tailings and ignores, for example, their cohesive nature. As the grain size of particles decreases, inter-particle attraction becomes important and begins to influence the criterion for resuspension. If only 10% of a soil is of clay size, the clay properties will completely dominate the sedimentation behaviour of the soil. Tailings may behave similarly. Results of resuspension calculations have not been verified by field investigation, with the exception of measurements at the Equity Silver site in British Columbia, Canada. Even this study is limited, as the surface observations of bedforms were only made at two different depths. In addition, the thixotropic nature of tailings may prevent the formation of bedforms during resuspension. The force required to resuspend the smallest tailings particle is much smaller than the force required to develop bedforms. Results of ongoing experimental work at The University of Western Ontario suggest that the lightest tailings particles go easily into suspension and will remain in suspension days after wave activity has ceased without the formation of bedforms.

No predictive water quality models examined in the review consider the depth of water cover in determining the flux of chemical species from tailings. In most cases, long-term

field data are not available for verifying predictions, although at the Løkken site in Norway, the predictions have been already shown to be unconservative. The depth of water may have a significant impact on water quality in several ways. Wind-induced stresses at the bed may prevent or delay the creation of anoxic conditions by disturbing organic matter or iron oxyhydroxide deposits at the tailings-water interface. Resuspension may increase the initial rate of acid generation in unoxidized tailings, as the gross rate of reaction will not be limited by diffusion of oxygen through interstitial water. Normally, oxidation creates a surface deposit or coating of reaction products, which can then limit further diffusion of oxygen to the unreacted iron sulphides. Resuspension may abrade this coating and expose fresh tailings surface for further oxidation. For previously oxidized tailings, resuspension would increase the initial rate of dissolution of secondary minerals and raise metal concentrations in the water cover. Even in the absence of resuspension, mixing of the water cover by wind will effect metal release by altering the rate of metal diffusion into the water cover. Without mixing, the concentration of metals above the tailings-water interface will build up, lowering the concentration gradient across the interface. Mixing will, however, disperse metals throughout the water cover, resulting in lower concentrations above the interface, and therefore a higher gradient and a faster rate of diffusion across the interface.

The minimum depth of water, the composition of the tailings, the hydrological regime, and the concentration of metals in the water cover of the sites reviewed, are presented in Table 6-1.

Table 6-1 indicates the Løkken pond has a relatively low pH and high concentrations of iron and zinc. These tailings were probably oxidized prior to flooding, and the low acidity is believed to be due to the release of stored oxidation products, such as ferrous iron and thiosalts to the water cover, which will in turn oxidize. The leaching of zinc may be accelerated by the increased acidity generated by this additional oxidation. The low pH of the water cover increases the solubility of ferric ion, which may cause additional iron sulphide oxidation.

The Equity Silver site on the other hand has a low sulphur content and has the greatest minimum and median depths of water. The geochemical investigation of the site revealed that, as with subaqueous disposal of tailings in lakes, the mobility of many metals is not controlled by oxygen, and that tailings may act as a metal source or sink under reducing conditions.

It is apparent from Table 6-1 that the hydrology of each site is critical in determining its performance, as metal concentrations in the water cover are controlled by dilution from precipitation and by inflow and outflow. The flux of metals out of the tailings is effected by downward flow from the cover into the tailings. Downward flow may cause transport of metals off-site through groundwater flow; however, the metal mobility may be reduced by the natural alkalinity of the soil surrounding the tailings impoundment.

Table 6-1 Characteristics of Reviewed Water Cover Sites

Site	Minimum and Median Depth of Water	Tailings Composition	Site Hydrology	Metal Concentrations in the Water Cover	Notes
Equity Silver	1.5 m, 4.3 m	S 2-10 % Fe 4.8-11 % Cu 0.11 % Zn 0.06 % As 0.5-1.5 %	No surface inflow or outflow	Fe ~ 0 Cu ~ 0 Zn < 0.5 mg/L As ~ 0 pH 7-8	
Quirke Cell 14	0.6 m	Pyrite 5.3%	average outflow 40 L/s, significant downward seepage into pore water	Fe 0.2 mg/L Zn 0.01 mg/L pH 6.5 - 7.5	Oxidation products flushed down into pore water
Panel Wetlands	0.3 m	As above	Large inflow and outflow	Fe 0.4 mg/L pH 6.5 - 9.8	Vegetation at low water levels
Solbec – Cupra	1 m	Cu 1.56 % Zn 4.5 %	Outflow level only reached in Jan 1996	Fe < 0.4 mg/L Cu < 0.4 mg/L Zn < 0.2 mg/L	
Hjerkinn	1 m	S 5 - 18 % Fe 17.5 % Cu 0.26% Zn 0.4 %	Outflow 70-450 L/s	Fe 0.2 mg/L Zn 0.147 mg/L	Moles of sulphate and calcium the same
Løkken	1 m	S 40 % Cu 0.32 % Zn 0.32 %	Outflow 0.3-24.8 L/s	Cu < 0.6 mg/L Zn 2.5 mg/L pH 4-6	As above
Stekenjokk	0.7 m	S 20 % Cu 0.19 % Zn 0.66 %		Zn 0.03 mg/L pH 7-7.6	Constructed breakwaters and sand layer

Other observations made at the various sites in this review include:

A dramatic increase in pH during the summer, which may be either caused by microfauna activity, or by CO₂ degassing, or by an increase in alkalinity from surface runoff into the tailings ponds, or a combination of these factors.

Gypsum dissolution controls sulphate concentrations in water covers at several field sites.

- Surface layers consisting of orange material (iron oxyhydroxides) and organic matter have been observed to form at tailings-water interfaces at various sites. These surface layers provide some resistance to resuspension and may hinder further oxygen transfer.

Innovative water covers designs found from the review include the use of a very shallow water cover to encourage the growth of vegetation (Quirke Lake, Ontario), and the use of breakwaters and sand/ rock layers to prevent resuspension (Stekenjokk, Sweden).

6.2. LABORATORY RESEARCH ON WATER COVERS

In column and lysimeter experiments on both oxidized and unoxidized tailings, flooding was found to result in the formation of a thin surface coating of brownish-orange iron oxyhydroxides which, in previously unoxidized tailings, marks the extent of oxidation vertically into the tailings. The finite thickness of the oxidation zone apparently occurs because of the attainment of steady-state conditions between pyrite oxidation and sulphate reduction to sulphur. With time, the growth of organic deposits at the surface occurs, which further impedes upward metal flux to the water cover. The Noranda Technology Centre is developing a method to determine how long organic buildup takes, and how much organic matter is required to consume all the oxygen transported to the bottom of a water cover.

Results of column experiments on oxidized tailings show that flooding causes the dissolution of oxidation products and the subsequent diffusion of metals into the water cover, often resulting in metal concentrations higher than permissible levels. The dissolution of some oxidation products, or other minerals, may be slow and contribute to high metal concentrations in the water cover for many years. The use of a sand or a peat protective cover at the tailings-water interface reduces metal fluxes into the water cover to negligible levels. However, metal flux into the water cover may resume when the absorptive capacity of the protective cover is reached. The use of a peat layer creates highly anoxic and reducing conditions in the tailings, and also significantly increases the mobility of metals in the tailings pore water. The increase in metal mobility may be partially caused by the complexing of metals by organic compounds originating from the peat, which would limit adsorption and precipitation. Peat also tends to generate small acidity.

Laboratory experiments have demonstrated the effectiveness of a Biologically Supported Water Cover (BSWC). A BSWC involves colonizing flooded tailings with plants to facilitate organic buildup for limiting resuspension and metal flux through biological adsorption. Preliminary results suggest that its effectiveness in the field is dependent on

the depth of water and wind conditions.

The results of laboratory tests also show that coarser tailings under a water cover initially generate higher amounts of acid drainage than finer tailings. This could be due to the initially faster rate of vertical oxygen transport in coarser tailings.

The addition of lime to previously unoxidized tailings prior to flooding inhibits the activity of iron-oxidizing bacteria by keeping pH high. Lime addition to oxidized tailings increases pH and limits the mobility of most metals in the water cover and pore waters. Mixing the top portion of tailings with lime was found to be more effective than simply adding lime to the surface. Traditional methods used to estimate the amount of neutralizing material required, such as acid-exchange methods, were inaccurate when applied to long-term (37 weeks) experiments on flooded tailings.

7. CONCLUSIONS AND RECOMMENDATIONS FOR FURTHER RESEARCH

It is apparent from laboratory and field studies, that flooding tailings is the most successful method presently known for preventing and controlling ARD. However, in many cases the water cover discharge will require treatment to meet regulatory standards, and it is not currently possible to accurately predict the required amount and length of treatment. State of the art predictive contaminant models ignore many important phenomena in water covers, such as secondary mineral dissolution, metal release independent of oxidation, and resuspension, to name a few. Though it is hypothesized that a diagenetic environment for some tailings minerals will be established with the accumulation of organic matter, it is uncertain how long it will take to develop and whether it will occur under all conditions. Furthermore, some metals may continue to be released to the water cover in such an environment.

The effect of water depth on the performance of the water cover is not precisely known, as the effects of wind, including tailings resuspension, on the performance of water covers have not been adequately investigated. If significant additional metal release results from resuspension, a more fundamental approach to modelling resuspension will be required. If the additional metal release due to resuspension is insignificant, the only criterion may be to limit suspended solids in the water cover effluent.

When previously oxidized tailings are flooded large fluxes of metals into the water cover may occur. In addition, oxidation products may diffuse into the water cover, dissolve, and significantly increase the acidity of the tailings pond. The use of protective covers such as sand or peat, the mixing of alkaline material into the upper tailings to maintain a neutral pH, and the implementation of Biologically Supported Water Covers (BSWC) are all potentially effective methods to reduce contamination of the water cover. Protective covers and a BSWC

have the added advantage of preventing tailings resuspension. Uncertainties associated with each method, however, remain. Since the dissolution of oxidation products may continue for a considerable length of time, the absorptive capacity of a protective cover may be exceeded, and significant metal flux into the water cover may resume. Maintaining a neutral pH does not limit the mobility of all metals, and the difference in dissolution rates of alkaline and other minerals makes predicting the necessary amount of neutralizing material difficult. A suitable environment for the implementation of a BSWC may therefore not exist at all sites.

In order to determine the criteria for a minimum water depth, further understanding of wind effects, including resuspension, is required. Simple experiments may be conducted to assess if significant additional metal release results from resuspension. Examination and characterization of the beds of tailings ponds at different sites with varying depths, such as was done at Equity Silver, should also be continued. Resuspension should be evaluated in the field to verify and improve existing models. The development of accurate models may require fundamental research into the thixotropic behaviour of tailings.

Research into protective covers and BSWCs should continue. The determination of the absorptive capacity of the covers may be obtained using methods discussed in the review. The time for which significant mineral dissolution and metal release continues must be determined to evaluate the feasibility of protective covers.

The hypothesis that the burial of tailings with organic matter will lead to the establishment of a diagenetic environment in flooded tailings, must be confirmed by field observations. The evolution of tailings geochemistry at sites may be evaluated, as was done at Equity Silver, by monitoring the concentration of metals in the pore water and in the water cover, and by analyzing solid tailings samples. The development of vegetation and other aquatic life at water cover sites should also be studied. Particular attention should be paid to the effect of water depth on geochemical and biological development.

Though water covers are a promising technology, fundamental understanding of many phenomena influencing their performance is unknown, and the minimum depth of water required to minimize metal release cannot presently be determined with confidence.

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