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ALTERNATIVE MEASURES FOR ACID MINE DRAINAGE ABATEMENT AT NORWEGIAN MINES

Prepared for

NORWEGIAN STATE POLLUTION CONTROL AUTHORITY

By

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REPORT 64701/1
ALTERNATIVE MEASURES FOR ACID MINE DRAINAGE ABATEMENT AT NORWEGIAN MINES

EXECUTIVE SUMMARY

This report presents the author's observations following a brief review of acid mine drainage at seven representative Norwegian mine sites and recommendations on the development of appropriate abatement measures.

Norway has a history of intensive pyritic ore mining and processing for 300 years. Most of the mine sites are now abandoned and many of the remaining operations are, or will, be discontinuing in the near future. On exposure to the atmosphere, pyritic ores or wastes oxidize generating acid mine drainage (AMD). The acidic water dissolves metals, such as copper and zinc, which are carried in the seepage and run-off from the waste site into the surface waters of the surrounding environment. Concentrations of these metals reduce as dilution occurs with other surface flows and as the metals are geochemically attenuated. Close to the mine site the flows are toxic and damaging to man and the aquatic environment.

The resulting environmental damage can be quite severe. Examples are the impacts on water quality and fisheries potential in lakes such as Langvatn and Orevatn at Sulitjelma; Tunnsjoen and Store Skorovatn at Skorovas; Orvsjoen at Sekstus Gruve; Djupsjoen at Roros; and in rivers such as the Glama and Guala near Roros. Other surface water resources could be impacted near mines which are closing, if adequate long term close-out measures are not implemented, such as the Namsen river near Skorovas and Orkla river near Lokken Verk.

Norway shares the problem of AMD abatement with a number of other countries, all of which have come to realize the impact of AMD only recently. Abatement technology development is relatively new and many of the proposed measures are not adequately tested or demonstrated. Implementation of abatement measures are still substantially a process of trial and error. An example of unexpected results is provided by the abatement measures implemented at Kjoli Gruve. The need for a sound understanding of the processes of AMD; the location, condition and nature of sources; and their response to abatement measures is clearly demonstrated.

The development of any abatement plan must start with the definition of the site and AMD source characteristics; appropriate programs are described in this report. The migration route and environmental impact must be characterized and quantified in order that the consequences or benefits of alternative abatement measures can be evaluated. Alternative abatement measures should be proposed and cost/benefit analyses performed to determine what level of expenditure is justifiable and realistic. The report lists and briefly discusses alternative abatement measures and includes, by reference, methodologies that may be considered. This selection procedure is very important because of the high cost of abatement measures and the uncertainty of some of the results.

The report provides crude basic cost estimates for some elements of alternative abatement measures, based on the authors western Canadian experience, to provide an initial guide for alternative identification purposes. The costs and effectiveness of alternative abatement measures are highly site specific. Each of the seven sites visited are reviewed and comments made on the nature and sources of AMD, its migration route and impact on the surrounding environment. The potential for increases or decreases in the long term impacts are commented on. Potential alternative AMD abatement measures are mentioned, discussed and, where possible, a crude cost/benefit evaluation made. The purpose of this evaluation is not to define the most appropriate alternative (insufficient data and analyses were involved within the scope of this study), but to illustrate the types of alternatives that may be included and some of the issues that must be considered.

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ALTERNATIVE MEASURES FOR ACID MINE DRAINAGE ABATEMENT AT NORWEGIAN MINES

1.0 INTRODUCTION

1.1 Overview of Acid Mine Drainage in Norway

Norway has a history of intensive pyritic ore mining and processing for over 300 years. NIVA, 1987, lists some 39 substantial pyrite deposits as shown in Figure 1, the majority of which have been mined. Most of these mines are now abandoned.

During mining and milling, pyrite rich waste rock and mill tailings are deposited on the earths surface. Mine workings are also opened up exposing ore and waste rocks to the entry of oxygen and water. Oxidation of the exposed pyrite, by both chemical and biological processes, results in the generation of sulphuric acid. The resulting acidic waters leach heavy metals, notably copper and zinc, and other contaminants from the mine wastes and carry them away from their source of generation and into the environment.

Impacts on the environment of the acid mine drainage (AMD) can be severe. The Norwegian Institute for Water Research (NIVA) has conducted investigations and evaluations of the extent and impact of AMD from abandoned mines in Norway and the results are reported in NIVA, 1987. Table 1 provides a summary of the pollution and environmental impact assessments for the various mining areas in Norway. Severe impacts are experienced or threatened in the Orkla, Glama, Gaula and Sulitjelma water courses.

1.2 Object of This Study

The object of this study was for the author to:

- visit and inspect seven representative mine sites in Norway from which AMD is occurring;
- make a brief review of the NIVA published data on contaminant migration data from the sites; and, based on his Canadian and USA experience in AMD abatement,
- suggest typical abatement measures which he considers may be applicable;
- indicate the steps required to investigate site specific conditions, develop conceptual abatement measures, evaluate the alternatives and select and implement the most advantageous; and,
- comment on the regulatory framework under which abatement can be implemented.

1.3 Basis For This Study

This study was essentially to be an overview of Norwegian conditions, conducted during a one week visit in early September, 1987, covering the seven mine sites, with assessments and evaluations based on the authors experiance in AMD abatement in similar climatic and topographic conditions on the West coast of Canada. Time and available data were insufficient to perform an in depth evaluation on any one site. Conclusions and recommendations included in this report, written after the visit, is intended to provide guidance on the methods and procedures to be followed in developing appropriate abatement measures and not a prescription of the abatement measures.

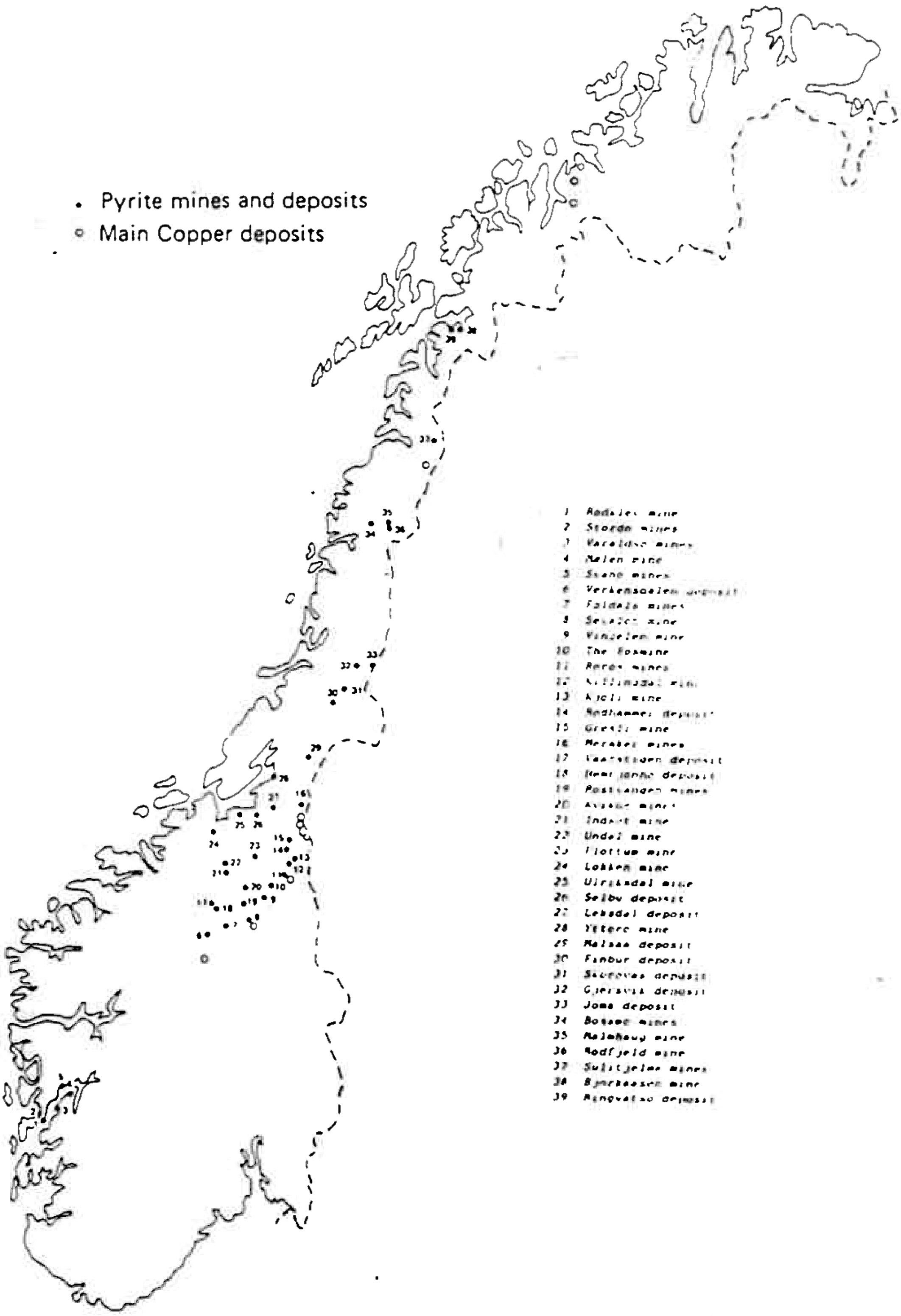


Fig. 1. Pyrite deposits in Norway as known in 1926. (Foslie 1926)

TABLE 1 SUMMARY OF POLLUTION AND ENVIRONMENTAL IMPACT ASSESSMENTS FOR NORWEGIAN METAL MINES (NIVA, 1987)

In the table O and X denote the following:

Effects

O Unknown, negligible
(O) Unknown (possible effects)
X Minor effects
XX Significant effects
XXX Severe effects

Inputs

O Unknown, negligible
X Moderate
XX Substantial
XXX Large

Sect. investigations	Name	Municipolitz	Draining to	Inputs	Effects	Further measures,
2.1.1	Vigsnes Copper Works	Karmøy	Lake Vigsnesvatn	XX	X	Inspection + controll programme
2.2.1	Evje Nickel Works	Evje and Hornnes	Otra	O	O	Inspection
2.2.3	Ertelien Nickel Works	Modum	Henoa/Tyrifj.	XX	X	Inspection + controll investigation
2.2.4	Langdalen	Romerike/ Modum	Skjærdalswater-course/Tyrifj.	X	X	Lower stretch of Skjærdalswatercourse should be investgated
2.2.5	Eiker Copper Works	Øvre Eiker	Drammenswater-course	X	X	
2.2.6	Modum Cobalt Works	Modum	Snarumselva-Simoa Drammensw.course	X	O	Inspection
2.2.7	Konnerud	Drammen	Drammenselva-Sandew.course	X	O	Affects ground water in the area
2.2.8	Grua	Gran	Viggaw.course	X	O	
2.2.9	Espedalen	Sør-Fron/ Gausdal	Gausa	O	O	
2.2.10	Fossgruva	Tolga-Os	Vangrøfta-Glåma	X	X	
2.2.11	Oscar II	Tolga-Os	Vangrøfta-Glåma	X	X	
2.2.12	Folldal	Folldal Alvdal	Folla-Glåma	XXX	XX	Detailed investiga-tined ... carried out 1984/85. Measures are under concerderation
2.2.14	Røstvangen etc.	Tynset	Gløta-Tunna Glåma	X	X	Effect of measures is being monitored
2.2.15	Kvikne Copper Works	Tynset	Ya-Orkla	XX	X	To be monitored of after the regulations of the Orkla w.course

TABLE 1 (cont'd.)

In the table O and X denote the following:

Effects		Inputs	
O	Unknown, negligible	O	Unknown, negligible
(O)	Unknown (possible effects)	X	Moderate
X	Minor effects	XX	Substantial
XX	Significant effects	XXX	Large
XXX	Severe effects		

Sect. investigations	Name	Municipolitz	Draining to	Inputs	Effects	Further measures.
2.3.1A	Røros Copper W. Stortvartz	Røros	Glåma	XX	X	Measures in Lake Djupsjøen
2.3.1B	Røros Copper W. Kongens	Røros	Glåma	XXX	XX	Lake Orvsjøen is has been monitored
2.3.1C	Røros Copper W. Muggruva	Røros	Gaula	X	O	
2.3.1D	Røros Copper W. Hessdalen	Holtålen	Hesja-Gaula	X	X	
2.3.2.	Kjøli	Holtålen	Gaula	XX	XXX	Furhter liming etc. in 1985?
2.3.3	Killingdal	Holtålen	Gaula	XXX	XXX	Measures in draft
2.3.5	Undal Works	Rennebu	Skauma-Orkla	X	X	
2.3.6	Løkken Works	Meldal	Raubekken-Orkla	XXX	XXX	Further measures should be drafted. The situation is being closely monitored
2.3.7	Dragset Works	Meldal	Vorma-Orkla	XX	X	The situation should be monitored by means of a controll programme
2.3.8	Høydalsgruva	Meldal	Svorka	X	O	
2.3.10	Lillefjell	Meråker	Stjørdals-watercourse	X	(X)	The Meråker area should be inspected
2.3.11/12	Gaulstad-Mokk			O	O	
2.4.1	Bossmo	Mo i Rana	Ranafjorden	X	(X)	
2.4.3	Sulitjelma	Fauske	Sulitjelma watercourse	XXX	XX	
2.4.4	Bjørkåsen	Balangen		X	O	
2.4.5	Tårstad	Evenes	Lavangseidet	O	O	
2.4.7	Bidjovagge	Kautokeino	Altawatercourse	O	O	

2.0 OVERVIEW OF AMD ABATEMENT TECHNOLOGY

In this section the mechanism of acid generation is briefly reviewed to provide a framework for the identification of alternative acid generation control options. Alternative technologies for preventing AMD from developing, from migrating to the environment or for its collection and treatment are identified and reviewed.

Our understanding of all the processes involved in acid generation and acid mine drainage is still only partial and under active development, as is the evaluation of the effectiveness of the alternative abatement measures. A considerable effort is being expended by industry, government and research institutions in countries such as the United States, Canada and Sweden on the development of appropriate technology. Rapid advancements are anticipated. Some of the evaluations made and conclusions drawn in this review are based on limited experience or data, and must be re-evaluated as additional information becomes available.

2.1 Acid Generation

Acid generation in mines and mine wastes have been described by numerous workers including Knapp, 1987; Paine, 1987; Errington and Ferguson, 1987. The following brief description of acid generation and its significance to alternative abatement measures, is extracted from a paper by the author (Robertson, 1987) which is included in the Appendix A to this report.

Acid generation is a time dependent process controlled primarily by:

- the presence and nature of reactive sulphides,
- availability of water,
- availability of oxygen (by convection and diffusion),
- bacterial action,
- temperature,
- pH or the presence of alkaline reactants.

Acid generation occurs both as a result of chemical oxidation and biological oxidation by the bacteria <u>Thiobacillus Ferrooxidans</u>. The rate of acid generation increases when both oxygen and water is freely available; as the temperature increases and as the pH decreases to about 3.5, at which bacterial acid generation is at its maximum.

If an alkali source is present (and available) in the waste in sufficient quantities, the acid products are immediately neutralized and additional acid generation is inhibited. This effectively abates AMD, despite the presence of sufficient sulphide, oxygen and water.

Acid generation prediction techniques are described by Ferguson and Erickson, 1987, a copy of which is included in the Appendix A. Concern exists that the conditions imposed during testing, to accelerate the time dependant reactions, result in unrealistically conservative results. This limitation is of less relevance in Norway where AMD is a fact at most of the sites of interest, and field measurements of AMD can be used to define the nature and extent of AMD.

2.2 AMD Control Technology

AMD control technology can be divided into three types; control of the initial acid generation; control of the migration of the acidic products; collection and treatment of the contaminated waters.

2.2.1 Control of Acid Generation

Acid generation control measures aim at control of one or more of the factors listed in Section 2.1. Alternative control technologies are reviewed by Robertson, 1987, who concludes:

i) The removal of pyrite from the wastes cannot generally done economically.

ii) Rendering pyrite minerals inactive by developing chemical coatings holds promise but does not as yet represent viable technology.

iii) Exclusion of water to the extent that acid generation is prevented can be achieved, with medium term stability, only with synthetic membrane covers; provision for replacement allows them to be used in the long term. Soil covers are sufficiently permeable that, in high infiltration areas (such as Norway), they pass sufficient water for acid generation to continue. Soil covers may however serve to control acid migration.

iv) Exclusion of oxygen can be achieved for the long term with water covers; for the medium term with synthetic membrane covers; reduced but not excluded for the medium term with soil covers. Water covers do not require a continuously flooded cover, as a saturated layer of soil (bog or marsh) layer over the waste will effectively exclude oxygen. Soil covers have recently been evaluated on a theoretical basis by Collin, 1987; Steffen,Robertson and Kirsten, 1987, amongst others, but there is still a paucity of relevant field data or experience. If water covers are used for waste rock, the rock pile (under water) should also be covered with a thin layer of material with a low hydraulic conductivity to prevent convective water circulation in the pile, which could transport dissolved oxygen to the waste surfaces.

v) Bacterial action can not be controlled in the long term by surface spraying or admixing bactericides.

vi) Natural temperature control plays a significant part in the control of acid generation in cold climatic zones. This effect can be enhanced to increase control at some locations. Placement of a cover reduces the maximum surface temperatures. In areas of permafrost, the potential for achieving permanently frozen conditions may provide effective acid generation control. Evaluations are being made in Canada of the potential of maintaining frost in discontinuous permafrost areas by using frost inducing devices such as thermal syphons or Cryopiles.

vii) The effect of a sufficient quantity of alkali material in the waste has been previously discussed. It provides an effective control measure if intimately mixed with the waste. If the alkali is soluble then consideration must be given to time period that will be involved before the alkali is leached from the waste. Surface applications of alkali are often not effective in that insufficient alkali is dissolved and transported into the waste to achieve neutralization, or preferential flow paths form, effectively by passing much of the acid generating waste. Neutralization of acid solutions with high metal concentrations may not result in adequate reductions in the contaminant concentrations. Zinc often does not precipitate to adequately low concentrations unless the pH is increased to considerably above pH 7. Thus zoned massive sulphide wastes, with volumes of acid generating and basic wastes, may result in neutral seepage which is still fairly high in zinc. The neutralized acid products remain in the pile. Many of these products are readily soluble in acid drainage, and regeneration of AMD would result in an increased environmental impact. Redevelopment of AMD must therefore be prevented.

2.2.2 Control of AMD Migration

Acid migrates as a flow of low pH water from the waste pile to the environment. Because of differences in permeability, coarse waste dumps and tailings deposits behave differently and equivalent abatement measures have different effects.

i) Waste dumps

Coarse waste dumps have a very high permeability to both air and water. Oxygen and water are available throughout the dump, allowing acid generation and reaction products to be distributed through the dump. Precipitation and surface flows onto the dump infiltrate rapidly. Runoff is often low. Flow through the dump tends to follow preferred channels along which the acid products are regularly flushed out. There are zones in the dump which are seldom flushed but receive moisture as a result of capillary action and water vapour migration and condensation. During periods of high infiltration, new or additional flow paths develop flushing out some of the accumulated acidic products. Thus we may see an increase in the concentrations, as well as the seepage rates, in the early stages of high flow events, i.e. during the spring freshet. This results in a huge increase in the contaminant load. Disturbance of a waste pile, loaded with reactive products, changes the flow paths and results in temporary increases in the contaminant concentrations. This effect appears to be demonstrated by the Kjoli mine waste reclamation project, in Norway, (NIVA, 1979 and 1986) where consolidation and contouring of a rock waste pile resulted in a huge increase in the contaminant load in seepage waters.

ii) Tailings impoundments

In contrast to waste rock dumps, infiltration and flow of air and water through tailings impoundments is much more uniform on a micro scale but restricted by the relatively low permeability of the tailings. This flow restriction results in the development of zones of oxidation and contaminant migration within the tailings as described by Robertson, 1987, and illustrated in Figure 2.

Acid generation in the tailings generally commences after the last of the tailings have been discharged to the surface of the impoundment. Prior to this, acid generation is effectively prevented by the neutralization which occurs each time a new layer of tailings is deposited (effectively alkali control). Once acid generation starts it may be many tens of years before the acid product front passes through the tailings and foundation soils to reach a surface discharge location. Surface flows across the tailings, on the other hand, may produce AMD shortly after tailings discharge stops. This is of significance to some of the Norwegian deposits where modern fine grained tailings impoundments have only been constructed since the process of flotation has been implemented, and tailings placement has only recently (in the last decade) been discontinued.

It may appear that there are two approaches to the control of acid migration: (i) control of seepage entry to the waste using diversion ditches and low permeability covers and (ii) control of seepage away from the waste using liners or cut-off walls. In practice approach (ii) can not function in the long term since infiltrating water which can not escape via seepage must collect until available storage is filled and then discharge over the lip of the liner or cut off wall. Thus the only long term abatement is infiltration control and the prevention of groundwater entry into the waste.

Diversion of surface flows, including ditching to divert surface flow from reaching the wastes, and contouring to promote surface water run off are always cost effective measures to reduce contaminant migration flows. The long term stability of such diversions, subject to sedimentation, vegetation and debris blockage, ice blockage and erosion is a concern. Surface contouring is only effective if the waste pile or tailings are not subject to substantial erosion, consolidation settlement or frost heave. These aspects are reviewed in Steffen, Robertson and Kirsten, 1987.

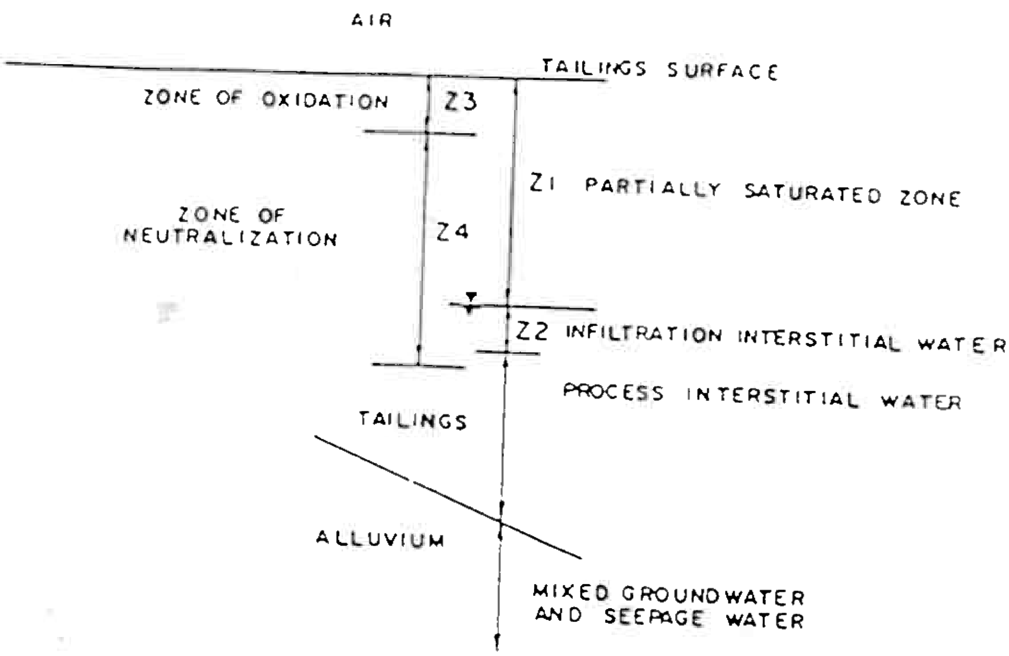


FIGURE 2(a) WATER QUALITY MODEL FOR TAILINGS COLUMN

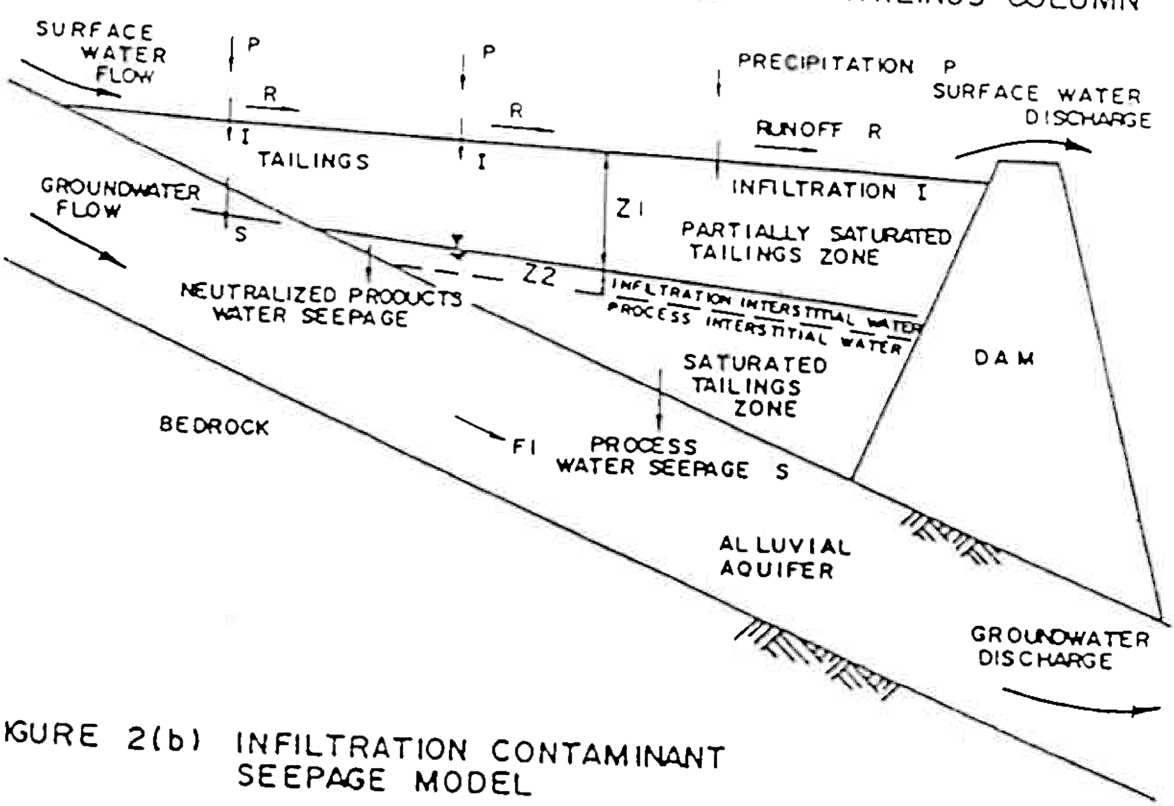


FIGURE 2(b) INFILTRATION CONTAMINANT
SEEPAGE MODEL

Surface infiltration can be appreciably reduced by the placement of low permeability soil covers. However soil covers have a finite permeability which may be increased in the long term by erosion and frost action. Water infiltration through such a cover is not dependant only on the permeability but also on the water balance that develops in the cover as a result of run off, infiltration, storage and evaporative and transpiration losses. This water balance is drastically affected by factors such as ponding on the waste surface, presence of suction breaking layers etc. and cover layers can be designed to reduce infiltration. Work on cover design to reduce infiltration is described in Rasmuson and Erickson, 1986, and Steffen, Robertson and Kirsten, 1987. Research work, to establish the long term effectiveness of such designs, is still very limited.

Synthetic membrane covers can and have been used to reduce infiltration to very low values. Relevant technology and materials are developing rapidly as a result of extensive use of synthetic membranes for isolation of other wastes elsewhere in the world. The effective life of a suitable synthetic membrane (such as 2 mm high density polyethelene, HDPE), adequately covered to prevent physical damage, should be at least 100 years. The present value of providing for the replacement of such a cover every 50 to 100 years is a relatively small portion of the initial installation cost.

Groundwater leaching can be very difficult to prevent if the site has not been selected to avoid it. Cut-off walls may be considered.

2.2.3 AMD Collection and Treatment

The collection and treatment of AMD has been the most widely applied abatement measure to date. A wide number of papers are available on the topic (Vachon et al, 1987). Collection of the surface flows is usually fairly readily achieved. The collection of subsurface flows requires the installation of collection trenches, or wells, or cut-off walls to force the groundwater flow to the surface where it can be collected. Most collection systems require long term maintenance.

Treatment may be grouped into active and passive systems.

i) Active systems

Active systems require the continuous operation or intervention by man. Chemical treatment, in a water treatment plant, is an example.

Chemical treatment involves the addition of an alkali (usually lime or quick lime), and the settling of the resulting sludges in a settling pond. Major concerns relating to this abatement measure are the need for long-term treatment, the quantities of sludges produced and the requirement for the long-term stabilization of the sludges.

The prime advantage of chemical treatment is that it involves known and tried technology, and continuous monitoring enables corrective action to be taken to ensure that discharge standards are met. Major disadvantages are the ongoing requirement for active care and the associated costs; a high risk of equipment or power failure resulting in untreated discharges and the need to provide safe long term storage for the accumulated sludges, which may be more difficult than the initial containment of the pyritic wastes.

ii) Passive systems

Passive systems are intended to operate without frequent intervention by man. These include:

a) Passing AMD through limestone trenches or channels. The large volumes of sludge produced results in rapid coating of the alkaline materials and the filling of the voids between them. Thus the trenches have a short effective life and cannot be considered in the long-term.

b) Natural or developed wetlands which rely on both geochemical and biological effects to achieve treatment and water purification (Kalin and Van Everdingen, 1987). While this method still has a number of uncertainties it does offer hope for a long-term passive treatment system, particularly as a polishing step in improving water quality.

2.3 Control and Abatement

Control of AMD is the objective in all containment and reclamation designs. Control implies that AMD can be constrained, by specific actions by man, to some desirable level consistent with cost and environmental impact.

Abatement is the term used to describe those actions which will reduce AMD and reduce the environmental impact. It is often not possible to pre-determine quantitatively the effect of an abatement action; or the effect may be less than that needed to achieve control. At many abandoned mine sites the cost of controlling AMD is prohibitive but cost effective abatement measures can be applied to significantly reduce AMD.

3.0 REVIEW OF REGULATORY ENVIRONMENT FOR AMD ABATEMENT AND CONTROL

In this section some of the factors important to the implementation of regulatory control and abatement of AMD are reviewed.

3.1 Introduction

Concern regarding the environmental impacts of AMD has developed primarily in last decade in concert with the increased public concern with environmental impacts in general. In the United States the extent of the impacts of AMD have been determined substantially as a result of investigation and research conducted under the Comprehensive Environmental Response, Compensation and Liability Act of 1980, and the subsequent Superfund Amendments and Re-Authorization Act of 1986. Substantially as a result of Government acceptance of the need to develop and implement remedial actions there has been a change in focus in the design and implementation of abatement measures. More funds have been available to evaluate the impacts at sites abandoned by those originally responsible for mining. Government interests have also focused more on achieving clean-up and long term controls which will ensure that future intervention is eliminated or minimized.

Concerns have developed over the same period in other countries such as Canada, Sweden and Norway. The mining industry shares this concern and has taken the initiative at many locations in the development and implementation of abatement measures at operating or newly abandoned sites. The technology of AMD and its abatement is complex and will require the combined efforts of Industry and Government to develop and test appropriate solutions. In Canada a combined Industry and Government committee has been formed to arrange funding and coordinate research into AMD and its abatement (Reactive Acid Tailings Studies committee, Ferguson, 1987).

3.2 Protection of Water Quality

The major impact of AMD is on the receiving surface waters. Protection of the surface waters requires the definition of a set of desirable water quality criteria. The desirable criteria should depend on the quality of the receiving waters (if not already impacted by the AMD) and the resource value of that water. Streams which sustain large sport and commercial fisheries should have different criteria to those with low natural quality which are unproductive or little used.

The regulatory approach to new mines is often to require the use of best available technology with some minimum criterion which must be met. For abandoned mines with AMD these minimum criteria may be technically impossible or exorbitantly costly to achieve. Under these circumstances some form of cost benefit evaluation of the alternative abatement measures makes considerable sense.

Included for reference purposes in Appendix B are the first pages of a selection of North American Guidelines and Standards:

CANADA:

i) Federal;
   - Canadian Water Quality Guidelines, March 1987, Chapter 3 Freshwater Aquatic Life.
   - Environment Canada, Metal Mining Liquid Effluent Regulations and Guidelines.

ii) British Columbia, Provincial;
   - Pollution Control Objectives for The Mining, Smelting and Related Industries, Table V Objectives for the discharges of final effluents to marine and fresh waters.

UNITED STATES FEDERAL:

- Environmental Protection Agency; Water Quality Criteria Documents - 1980.
- EPA; Ambient Water Quality Criteria for Copper - 1984.
- EPA; Water Quality Criteria; Notice of final ambient water quality criteria documents - 1985.

In Norway, as in Canada, the surface waters in rivers tend to be of very high quality, and sports fishing and commercial salmon fishing are often effected by AMD. Aquatic water standards may therefore be applicable. Selection of an appropriate standard is site dependant but a logical selection procedure for abatement level may follow the reasoning of the following example.

Possible standards to consider:

Level 1. Meet Canadian Water Quality Guidelines for Freshwater Aquatic Life (CWQG-FWL).
Level 2. Meet CWQG-FWL for dissolved or acid extractable metals.
Level 3. Meet no observed effect level for commercially important species.
Level 4. Meet no acute toxicity effect for commercially important species.

Example of levels for copper:

- copper toxicity to Atlantic Salmon
- water hardness = 50 micro g/l
- LC50 for Atlantic Salmon = 197 micro g/l 0.000197

Level 1 - Total copper	= 0.002 mg/l
Level 2 - Dissolved copper	= 0.002 mg/l
Level 3 - T. or D. copper	= 0.035 mg/l
Level 4 - T. or D. copper	= 0.098 mg/l

A maximum total copper concentration of 0.005 mg/l, in the receiving waters of a salmon river, was recently set by the British Columbia Ministry of Environment and Parks as the objective for abatement of AMD from an abandoned copper mine.

All of these standards may be very difficult to meet in instances of severe AMD to sensitive receiving waters. A further reduction in objectives may have to be considered on economic grounds.

3.3 Confidence in Control Technology

Experience with the implementation of control measures for abandonment and their long term effectiveness is embryonic: there is not a clear understanding of what is effective under any given set of circumstances. Development of an AMD control or abatement plan involves the consideration and evaluation of technology and methods which are still under development and test. There is considerable doubt as to our ability to accurately determine the effectiveness of some of the proposed measures. The cost of the 'secure' measures are often prohibitively large and we are behooved to carefully evaluate the lower cost alternatives. With each alternative there is an associated risk of failure, which is substantially dependant on the reliability of the technology involved. The following review discusses some of these uncertainties.

3.4 AMD with Time

Acid generation is an evolutionary process which may take many decades to develop and evolve.

Experience in British Columbia has shown that in some dumps, with a moderate neutralizing capacity, it may take ten or more years before an acid generating waste develops its potential to the extent that marcasite

AMD becomes significant. Where the pyrite coats the fracture surfaces and is of a type which oxidizes rapidly (marcosite) acid generation can be much more rapid and may reduce after a time as available pyrite is spent or becomes coated with oxidation products. As coarse waste rock weathers and breaks up, new surfaces and pyrite are exposed. Thus a slaking waste may exhibit an increase in acid generation with time.

The rate of acid generation in fine grained tailings are limited by the rate of oxygen diffusion and water infiltration and seepage. While the surface run off becomes acid very quickly, the initial AMD seepage is neutralized along the seepage path. As discussed by Robertson, 1987, it may take decades for the full AMD potential from a tailings impoundment to establish.

Over the very long term there must be a reduction in the AMD as the available pyrite is spent. The total acid generating potential of a one million tonne waste deposit containing 15% sulphides is about 450 000 tonnes or 1 500 tonnes per year for 300 years; sufficient to be considered an infinite source.

3.5 Long Term Stability of Control Measures

It is desirable that abatement or control measures should function as intended inperpetuity. Unfortunately this is seldom achievable. All geotechnical structures are subjected to disruptive forces. These may be divided into two classes; extreme events and perpetual forces.

Extreme events such as floods, earthquakes, tornadoes and fires subject the reclamation measures to forces for which they may not have been designed. Where tailings dams are constructed in water courses or have associated diversion structures, the flood hazard may be high. Robertson and Clifton, 1987, review requirements for the long term stabilization of tailings impoundments.

Erosion by wind and water, frost action, root action and burrowing animals are examples of the perpetual forces. Acting over the long term of interest, these forces wear away the covers and embankments constructed by man.

3.6 Maintenance

With out some maintenance, most geotechnical structures will fail in time. Ditches, for example become blocked by sediment, debris, vegetation or ice. Overflowing of the ditch may result in scour and the formation of a new flow channel with associated erosion. While maintenance may be infrequent, perhaps only every few tens of years, it is an essential part of the long term security of waste disposal systems. Since the mining companies that undertake the mining and reclamation may have a limited functional life, the onus of maintenance will accrue to the State. It is considered appropriate that this be recognized and that the appropriate State Regulatory Authority make arrangements for the provision of such maintenance.

3.7 Responsibility for Abatement Costs

Abatement costs can be high. In many instances the entity responsible for the placement of the wastes is no longer in existence and the onus of clean-up accrues to the State.

In other instances the properties have been operated and possibly prepared for abandonment in accordance with the laws of the State at the time. Such properties may have passed on to new owners. Owners of such properties find themselves in possession of a property requiring abatement actions for which they do not feel responsible, have not anticipated in their financial arrangements and may not be capable of undertaking. Operating mines wishing to close and abandon find themselves in an essentially similar situation where they may be required to implement abatement measures which were never part of the States requirements at the time of mine planning and financing.

While the responsibility for abatement costs are a matter for the State to decide, it is the writers opinion that the cost of implementing abatement measures required as a result of a change in State requirements, should be born by the State. The minerals recovered in generating the wastes were used for the benefit of the citizens of the State, provided employment and taxes. The liability and benefits, as they were understood, were accepted by all beneficiaries at the time of project initiation. It is appropriate that they all share in the incremental costs of AMD abatement required as a result of recognition of an increased environmental impact and the desire to abate the impact.

4.0 EVALUATION AND SELECTION OF ABATEMENT MEASURES

This section lists and discusses the major activities required to develop, evaluate and select an appropriate set of abatement measures for a particular site at which AMD is occurring. The required activities comprise:

i) Determining the physical characteristics of the site.
ii) Determining the location, physical and chemical characteristics of the sources of AMD.
iii) Determining the nature and extent of contaminant migration resulting from the AMD.
iv) Description of the environment and the impact of the AMD.
v) Conceptual design of abatement alternatives.
vi) Assessments of the costs and benefits.

4.1 Characterization of Site

Prior to starting any site specific investigation and abatement measures design it is first necessary to prepare the data base on the physical characteristics of the site. This includes:

- Topographic maps of the area at sufficient scale to define the location of all major physical features on the site such as buildings, structures, dams, waste piles, pits, mine entries, water courses etc. Use of air photo interpretation is of considerable benefit.
- For underground mines; mine maps, locations of all mine development, ore remnants, openings, water inflow records, support and subsidence records.
- Geological maps which define the underlying soils and rock materials, and geohydrology.
- Climatic data, including precipitation, evaporation, temperature and wind velocity data.
- Location, history and nature of mining activities, and all waste deposits.
- Location and nature of naturally available construction materials (clays for covers etc.) and alternative disposal sites (lakes, mine workings or impoundment sites for under water disposal etc.).

4.2 Characterization of AMD Sources

All wastes are not acid generating, and acid generating wastes have different degrees of severity. The acid generating characteristics of the various sources on a site should be determined as well as the AMD that is occurring from each source.

i) Determination of sources

Sources are determined by conducting seep and run-off surveys. It is recommended that at least four seep surveys be conducted during the course of the first year to determine the temporal variation of the flow rates and concentrations of the various seeps. Such a set of surveys will serve to demonstrate both the location and size of the AMD sources on the site (hence those that require abatement). It may be found that waste rock from certain parts of the mine or from a particular mining period are not of concern. It will indicate the contaminant load variation during the year. This provides a better understanding of the sources and mechanism of AMD at the site as well as average and maximum load estimates. Two of the surveys should be done early and during the spring freshet when concentrations and loadings may be greatest.

Seep surveys are conducted by walking and inspecting the site at likely seep locations. Seep locations should be marked and recorded for future reference, flow measurements made using a flow meter, bucket or estimate, as is appropriate, and water samples collected. Samples should be tested in the field for pH and conductivity and in the laboratory for copper, zinc, iron, sulphate, acidity and calcium. Occasionally a full suite of 26 elements may be done on larger seeps. The seep surveys should be plotted on a map showing locations, flow rates, concentrations, loadings and the overall flow pattern.

Evaluation of the conservation of flow volumes and loadings will indicate where sources and sinks are located.

The use of thermography (thermal photography) and various forms of remote sensing has been used with varying success to define zones of activity in the waste dumps.

ii) Characterization of each source

Each major source is surveyed to determine its size.

Mine maps, where available, are required for the definition of mine workings. The location and nature of wastes placed in the mine, ore exposures, remnants, all openings and water quality should be determined. Ore and wall rock samples are tested (if the information is not available) as per the waste rock samples discussed below.

Waste rock deposits are sampled both from the surface and at depth and tested to determine;

- pH of interstitial water, if any, or of a de-ionized water wash
- ph of paste of waste rock with a small quantity of de-ionized water
- acid and base potential
- for some of the samples copper, zinc, total sulphides and sulphates and other constituents as appropriate.

Tailings solids and interstitial pore fluid samples are taken at various depths on profiles in the tailings impoundment at locations representative of the beach area (coarse tailings) and the pond area (slimes). Tailings interstitial water is separated from the solids using centrifuge separation or washing techniques. The interstitial water is tested for pH and conductivity in the field and in the laboratory for copper, zinc, iron, sulphate, calcium, and acidity. These tests provide a profile of the both the acid generating products and the neutralization products to determine the various zones defined in Figure 2. Some of the solids samples are tested for the same constituents as the waste rock.

Both rock waste and tailings may be subjected to kinematic tests to determine the nature and rate of acid generation evolution.

Based on this characterization the current AMD yield from each source must be estimated plus an assessment of how this is going to change with time. For example, in a tailings impoundment, it may be concluded that the surface yield of contaminants has fully developed but that the yield to the seepage flow is still developing and will result in increased loadings at some time in the future. To estimate such future yields it may be necessary to perform modelling of the acid generation and migration such as described by Jayes et al., 1984, and Davis and Richie, 1986, for waste dumps and SENES, 1986, for tailings impoundments.

4.3 Characterization of Contaminant Migration

Much of the investigative work done to date on the impacts of AMD from Norwegian mines has been characterization of the contaminant migration in the surface flows away from the sources and into the environment. The object of such characterization is to define the flows and loadings of contaminants being carried away from the site. Sampling and testing should be done at a sufficient number of times during the year to define the temporal variation of the loadings and concentrations. Care should be taken to define the time and conditions of maximum concentrations in the most important receiving waters. Monitoring should be continued for as long as it is necessary to determine in if the contaminant plume is increasing, stabilizing or reducing. If an increase in the AMD release rates, or further development of the contaminant plume, is anticipated then the downstream flow path should be sampled, tested and described to a sufficient extent to enable the contaminant plume development to be evaluated by modelling. This will require the determination of the quality and buffering capacity of all downstream diluting flows and the flow channel environment (natural AMD abatement in bogs, flows though alkali soils or rock channels etc).

In addition to the surface flow investigations currently being done some consideration should be given to subsurface flows. Where waste deposits are underlain by aquifers the migration route to the environment may be along these aquifers. During the initial stages of plume development the buffering capacity along such a route may abate AMD and reduce the contaminant loadings to surface waters. As the buffering capacity of the flow path becomes exhausted the contaminant yields may increase. Such increases should be anticipated if the long term impacts of the AMD is to be evaluated. Sampling and testing of the subsurface flow path may include test pitting or drilling to sample the soil strata and laboratory tests to determine the neutralizing and buffering capacity of the soils to the AMD, including column leach tests to determine coefficients of retardation, Kd's. It is also necessary to define the physical parameters which control flow such as the hydraulic conductivity, gradient, and storage along the flow path. The resulting information may be used in a contaminant migration prediction model to a level of sophistication to suit the concerns associated with that particular flow path.

4.4 Characterization of Environment and Impacts

The economic and other benefits of AMD abatement can only be assessed if the baseline environmental conditions and the impacts of AMD are clearly defined. The NIVA investigations to date have concentrated effectively on the definition of baseline surface water quality and the impacts of AMD on these. These characterization studies should be extended to define the baseline aquatic life and impact on the aquatic life as well as the other environmental impacts such as effects on consumptive use by man, sports and recreational value impacts etc.

4.5 Conceptual Design of Abatement Alternatives

Conceptual designs are prepared for all potentially viable abatement options in order that the potential effectiveness, impacts, and costs can be estimated with a view to making a rational and defensible selection of the most cost effective and appropriate option.

Alternatives which should be considered include, but are not necessarily limited to:

- Diversion of surface water
- Conditioning of wastes
    - pyrite removal
    - dewatering
    - placement methods
    - solidification
- Covers
    - vegetation (in combination with others)
    - soil, till or clay
    - synthetic membranes
    - water
    - saturated soil or bog
- Base addition
    - mixing with waste
    - surface application
- Alkali trenches
    - up stream
    - down stream

- Bactericide
  - mixing with waste
  - surface application
- Collection and treatment
  - chemical treatment
  - peat or woodwaste treatment
- Wetland treatment

A brief review, and references to further research, of these alternatives are provided in Robertson, 1987, in Appendix A. Many of the alternatives may be rejected, as not effective or applicable to a particular site, after a cursory consideration. The remainder must be evaluated in sufficient depth to enable a valid and defensible alternatives design, costing and benefits assessment to be developed.

4.6 Assessment of Costs and Benefits

4.6.1 Assessment methodology

Since we often lack reliable, accurate technology for the prediction of acid generation and the effectiveness of abatement measures, much of the assessment methodology is of necessity based on judgement rather than deterministic analyses. In a recent study for the abatement of AMD from a waste dump (Steffen, Robertson and Kirsten, 1987a) an assessment method was used which involved consideration of each of the following factors:

- practicality of implementing the measure
- durability (life expectancy) of the measure
- security or risk of failure
- environmental impact of constructing the measure
- construction complexity and duration
- inspection and maintenance requirements
- effectiveness in reducing AMD

A point score and weighting factor was assigned to each of the above factors to derive a total points ranking of the measures. A cost estimate was prepared for each measure. By dividing the cost by the total points achieved by each measure it was possible to develop a cost/points benefit ranking of the measures. This process assisted considerably in deciding which of the measures could be selected and, where appropriate, combined with others, to achieve the most cost effective set of measures.

A more definitive cost/benefit evaluation involves the determination of the present value of both the abatement measures costs and the financial value of the benefits gained in environmental clean-up. The value of environmental improvements are often difficult to quantify in a manner that enjoys general agreement. Steffen Robertson and Kirsten, 1987a, calculated the present value of the improvement in commercial and sport fisheries associated with the clean-up of the AMD from a particular mine waste site and found it to exceed the present value of the clean-up costs by a few times. The site they considered was in an area of relatively high population density and they did not include tourism, recreational value or any other values in the present value calculations. Such evaluations are extremely site specific and allow the level of abatement to be adjusted to the usage and benefit to man of the impacted resources.

4.6.2 Costs of abatement measures

In this sub-section approximate costs are estimated for some alternative abatement measures as they may apply to some 'generic' site. The rates, in Canadian dollars, are based on the writers experience in western Canada and must be adjusted to suit Norwegian conditions and particularly the site specific conditions on any project. They are provided to enable a first order estimate to be made of possible costs when making a preliminary evaluation of alternative abatement measures.

Diversion of surface water:

This is usually a low cost effective measure which should be considered as part of any abatement scheme. Trenching to achieve water diversion is dependant on the flows to be diverted, the need for rip-rap erosion protection, the slope of the terrain in which the diversion trench is formed and the need for blasting in the excavations. Costs can typically range from $10 per meter for small unprotected trenches in soils on gentle slopes to $100 or more per meter for ditches on sloping terrain requiring some blasting and rip-rap lining.

Conditioning of wastes:

Costs are highly site specific and it is not realistic to quote typical costs. Conditioning is unlikely to be practical or economic unless it is part of the mineral extraction process.

Covers:

Vegetation: Costs depend on the ability of the acid generating wastes or cover materials to support vegetation. Direct vegetation costs may range from $1000 to $2000 per ha. depending on the amount of waste or soil preparation and fertilizer that may be required. Additional costs are accrued when it is necessary to install drainage, contour and surface the wastes, improve access over soft or boggy areas and repeat lime or fertilizer applications. With such measures costs may increase to $5000 per ha. These costs do not include for any soil cover placement.

Soil, till or clay: The cost of cover placement is determined largely by the availability of suitable cover materials in close proximity to the waste site (within one or two km.), and the trafficability conditions to and on the waste. Under ideal conditions a 0.6 m cover (minimum recommended by the author) can be placed for about $2 per sq. m. This may increase to about $4 per sq. m for more difficult placement conditions (such as winter placement on soft tailings) with moderate haul distances. To this must be added the costs of waste pile surface contouring, access preparation, reclamation of the borrow areas and the cover surface. Where the cover is a complex layered cover (with clay layers and granular suction barriers) the costs will increase depending on the number and thicknesses of the layers and sources of materials. Rip-rap materials may be required on sloping surfaces to protect against erosion at a cost of between $3 per sq.m, if available as borrow locally, and $6 per sq.m if it has to be specially quarried. Vegetation may be sufficient erosion protection on gentle slopes.

Synthetic membrane: To ensure durability it is necessary to use thick membranes (2 mm say) with good chemical stability characteristics (such as HDPE), perform adequate bed preparation and place a cover (0.6 m minimum recommended) to provide physical protection. In Canada the supply and installation of 2 mm HDPE sheeting at remote mine sites in the typical quantities costs about $14 per sq.m. The cost of bed preparation and cover placement will be about $6 per sq.m and to this must be added the costs of waste pile contouring and revegetation.

Water cover: Costs and practicality are extremely site dependant and it is not possible to quote typical costs. It is seldom possible to achieve a water cover unless the disposal site was designed for one in the first place. If a natural water body, such as a lake, is available to which the wastes can be removed then the cost of the water cover option is proportional to the amount of waste to be moved. Waste removal, with a haul of less than one km, will typically cost about $2.50 per cu.m. Additional haul will be at about 50c per cu.m per km. To this must be added the cost of lake water quality control during waste placement, the cost of a soil layer on the submerged waste and reclamation of the area from which waste is removed.

Saturated soil or bog layers: Such layers can be economically achieved if the waste pile is suitably shaped, a water source is available and suitable construction soils are available on the site. It is not possible to quote typical costs.

Base addition:

The cost of base addition is primarily dependant on the availability of a cheap source of suitable base material close to the site, such as a limestone source. The quantity of base required is dependant on the potential acidity of the waste and can differ by and order of magnitude or more. Intimate mixing is usually required for this abatement measure to be effective in the long term. The cost of excavation of an existing waste deposit, base addition and replacement is likely to be at least $3 per cu.m. plus the cost of the base material delivered to site.

Alkali trenches:

The cost of the alkali trenches depends on the type and amount of alkali and the frequency with which it will have to be replaced. The present value of future replacements, as well as the cost af disposal of the spent alkali, should be included in the overall cost.

Bactericide:

Comments on costs are essentially similar to those for base addition. The costs of intimate mixing will be similar but the effective life is shorter.

Collection and treatment:

The cost of collection and chemical treatment is primarily dependant on the maximum flow volumes that have to be treated and the level to which treatment must be taken. Other abatement cost comparisons (Steffen, Robertson and Kirsten,1987a, amongst others) have shown that the present value of the long term costs of treatment can be comparable or less than alternatives which achieve clean-up. These comparisons generally do not take into consideration the long term problem of disposal of the resulting toxic sludges.

Wetland treatment:

Wetland treatment is being successfully implemented for AMD from coal mine in the eastern USA. The author is not aware of successful applications to AMD from metal mines in cold climate areas and is unable to comment on short or long term costs.

5.0 REVIEW OF AMD AND ABATEMENT AT SELECTED NORWEGIAN SITES

This section offers a brief review of the observations made during the visits to selected AMD sites in Norway with the object of illustrating some of the alternative AMD abatement measures that may be considered at those sites. It is necessary to stress the short duration of the visits, limited data availability and cursory nature of the conceptual abatement measures development and evaluation. A detailed site and AMD sources characterization study, contaminant migration and environmental impact assessment, alternative abatement measures design and evaluation is required before any specific set of abatement measures can be selected.

5.1 Storvartsgruva

5.1.1 AMD Impact

The location of the Storvarts mine is shown on Figures 3 and 4. Started in approximately 1644, mining continued until 1972.

Mining was from underground workings with a total production of more than 1 million tons. Initial mining involved the hand picking of high grade ore for direct roasting, resulting in surface piles and underground fill of relatively rich (-15%) sulphide rich waste rock. The mines have been allowed to flood but the level and quality of the mine water is not known. It is not known if flooding has reached its maximum height and if discharge or seepage can be anticipated. These aspects should be determined if the long term impact of the mines is to be determined.

The waste rock piles are scattered around the site. Some of the waste rocks (greenstones) are slaking and have weathered to soils. Other wastes, particularly the low grade sulphide rich ore/waste appears to be weather resistant. The weathering resistant waste appears to be clean and may have yielded the majority of the acidity available from surface oxidation. Vegetation is growing well on these old coarse rock piles and this may indicate that acid generation, at least near the surface, is now low and leached out. The slaking wastes appear to be oxidizing throughout and are a potential long term source of AMD. AMD seeps were apparent from the base of some of the waste piles and accumulations of AMD precipitants were apparent along the flow routes. Waste rock had been used for road and platform construction, further adding to the distribution of the wastes. Definition of the location and nature of the AMD generating wastes will require a detailed survey of the site and sampling of the various waste piles. A comprehensive seep survey, conducted during the height of the spring freshet, will indicate the sources of significant surface drainage of AMD. No data currently exists which defines the AMD sources and loadings attributable to these waste dumps.

During this century a modern milling and flotation plant was constructed and the ore milled and floated to extract the sulphides. The resulting tailings have a lower pyrite content but are clearly oxidizing and acid producing. They were discharged to a broad valley which drains via a stream to Djupsjoen Lake. Erosion of the tailings washed tailings down the stream forming a small tailings delta (approximately 2 ha above current lake level) in the lake. The stream channel was not inspected but undoubtedly tailings deposits must occur along it. The lake has a poorly maintained dam at its outlet (NIVA, 1980) and this presumably indicates that the level of the lake is artificially raised and that the lake deposit of tailings may be considerably greater than that observed. The current quality of the lake water is not known to the author but it is understood that it is improving and that fishing has improved in the lake.

To prevent further erosion of the main tailings deposit (more than 20 ha) the State constructed a earth embankment in about 1982. Water stored behind this embankment floods a small portion of the tailings. Measurements of the water quality from the overflow from this dam indicates that the water quality has a typical range as follows:

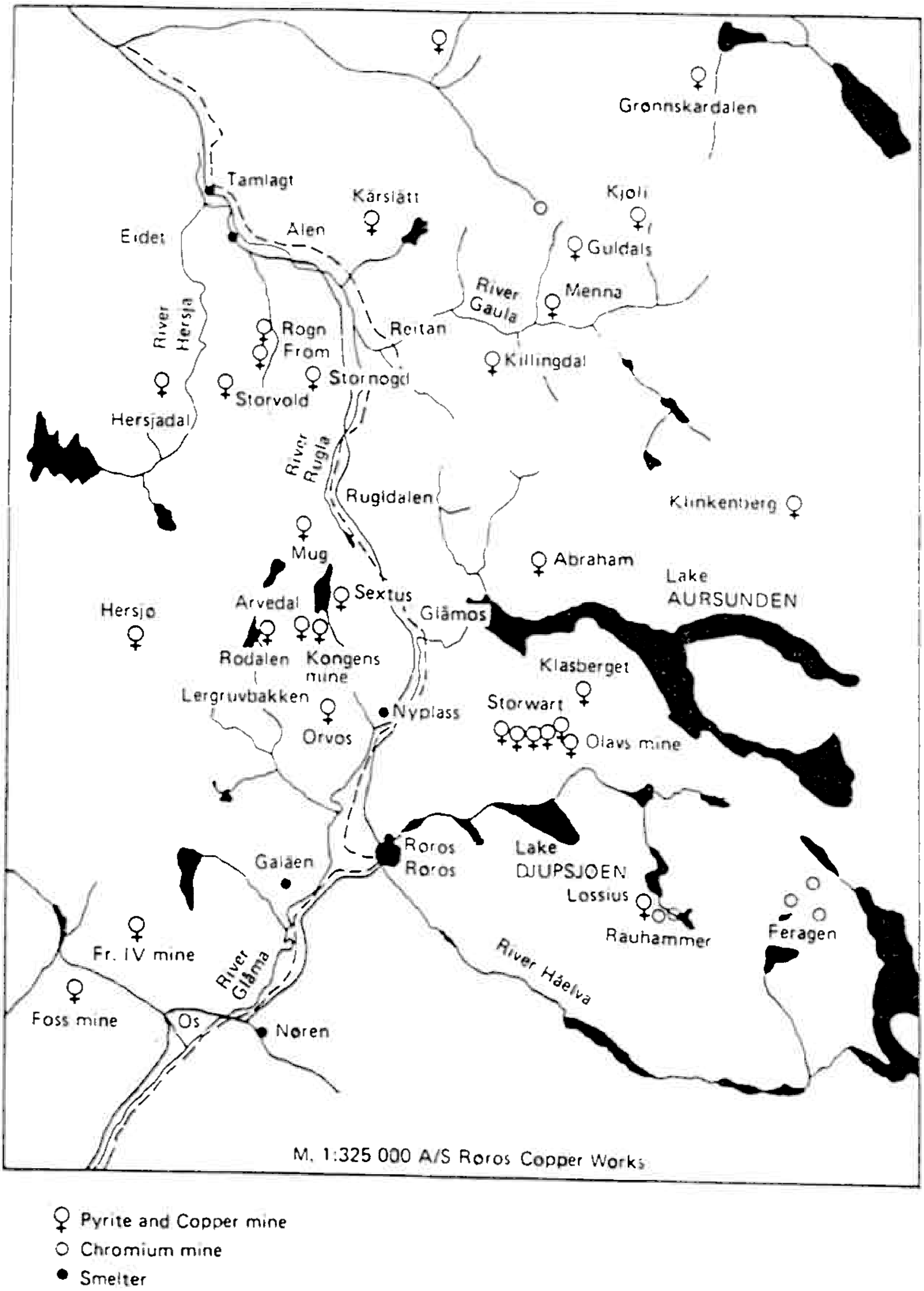


Fig. 3 Abandoned mines in the Røros area (after NIVA, 1987)

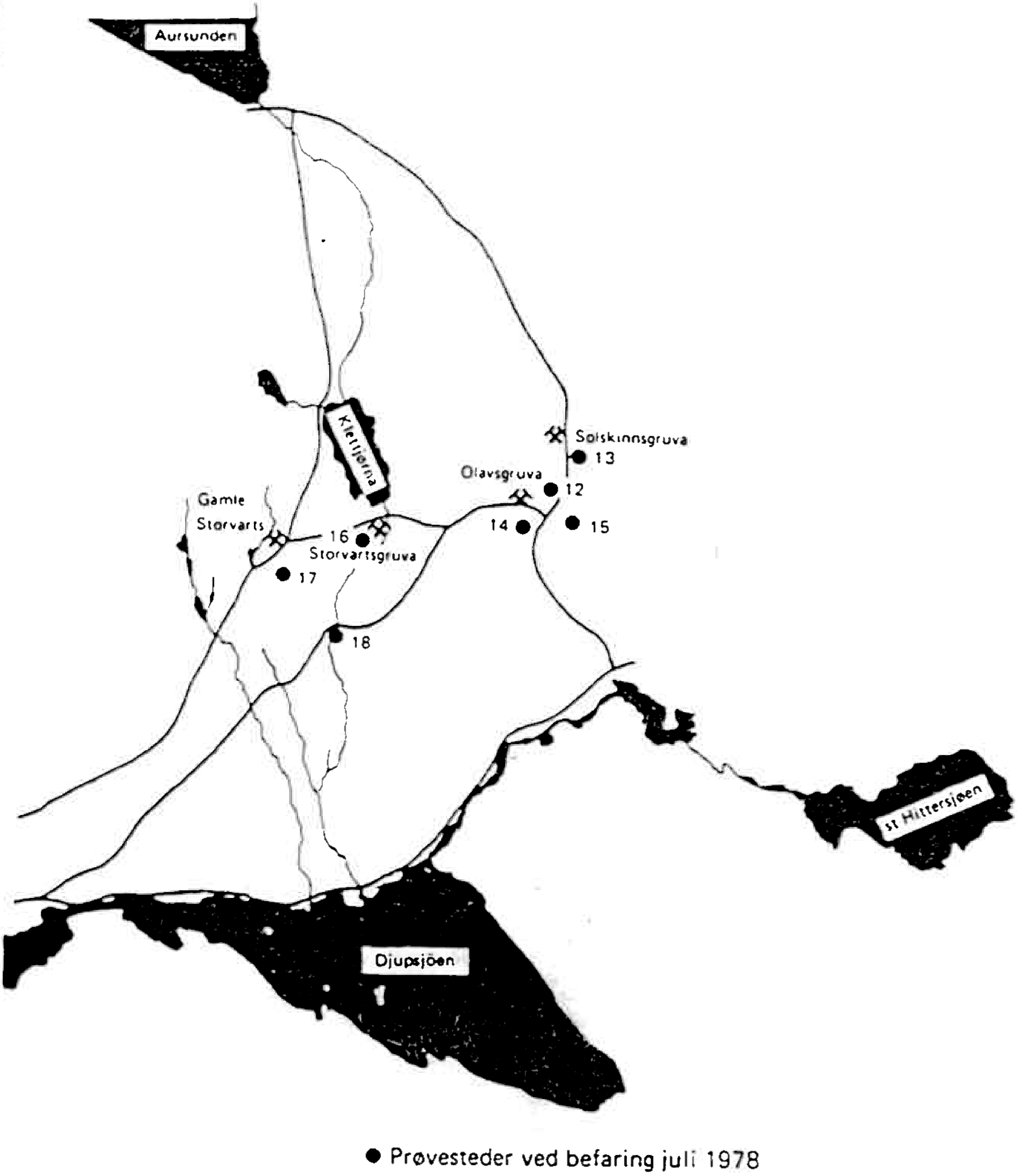


Fig. 4 Location of Storvartsgruva (after NIVA, 1980)

Copper	1-8 mg/l
Zinc	5-17 mg/l
pH	3.4-5

This results in about the following annual loadings:

Copper	1.6 tonnes/year
Zinc	7.2 "
Iron	12.4 "
Cadmium	12.7 Kg/year
Sulphate	155 tonnes/year

It was observed that there was substantial leakage through the earth embankment which discharged from the downstream drain. The extent to which sub-surface seepage was occurring is not known. These latter flows may be a significant load not measured at the dam overflow. AMD yield from the deposits along the stream and in the delta are not known.

The AMD yield from this property is expected to have reached its maximum, except possibly from the following sources:

- mine drainage, should it occur,
- additional exposure of the beach tailings if the lake level drops,
- additional groundwater seepage yields as the acid generation front develops in the tailings deposit.

These potential sources for increased AMD should be investigated. Should it be established that they are unlikely to occur then abatement may be planned to address current yields. The impact (July 1978) on the waters of Djupsjoen lake, as measured at the outlet, is such that the resulting concentrations are:

Copper	0.055 mg/l
Zinc	0.22 mg/l

Comparative values at Hitterelva, near Roros are

Copper	0.08 mg/l
Zinc	0.16 mg/l

The background values are not known to the author but it appears that the decrease in water quality is significant and may have a material impact on the aquatic environment. The drainage from this mining area flows into the Glama where its impact combines with that from other mining areas.

5.1.2 Alternative Abatement Options

Alternative abatement options can only be proposed with confidence once the steps outlined in sections 4.1 to 4.4 of this report have been completed. Based on a preliminary evaluation the following may be applicable.

A. Mine drainage: No action required.

B. Waste dumps:

Determine the AMD releases from the various dump locations from a detailed seep survey. Consider the following abatement measures for those dumps responsible for a significant percentage of the AMD yield:

B i) Divert surface flows around dumps to reduce leaching.

B ii) Cover dumps in-situ, with as little disturbance as possible, with a till or low permeability soil cover. Contour cover to promote drainage and establish vegetation to control erosion. Evaluate infiltration to asses reduction of AMD migration.

B iii) As an alternative to ii) consider consolidating all waste in a single pile (required if all waste is contributing to AMD) contouring and covering with a synthetic membrane, 0.6 m till cover and revegetating. Reclaim disturbed areas. This should provide an effective control to both acid generation and migration. Establish a fund to provide for the long term replacement of the membrane.

C. Tailings:

At tailings impoundment:

C i) Minimize contaminant migration by a) diverting current flows across the impoundment, round the impoundment, b) improving surface contouring to reduce ponding and hence infiltration.

C ii) Install a 0.6 m till cover over the exposed tailings and revegetation.

C iii) As for ii) but first install a synthetic membrane cover.

C iv) As an alternative to ii) install a water cover by increasing the size of the present embankment and dredging tailings from higher elevations into the impoundment to achieve a 1 m water cover. Water treatment will be required during the dredging period to control surface water discharges.

Tailings along stream:

C v) If these are yielding significant AMD they may be removed to the tailings impoundment. Placement in Djupsjoen would contaminate the lake heavily during placement.

Tailings in delta:

C vi) If impact is small they may be left where they are with increased security of water level control at the outlet.

C vii) If impact is large, due to portion above the lake level which cannot be flooded, consider removing this portion and transporting back to the tailings impoundment.

5.1.3 Cost/Benefit Estimates

The following cost/benefit estimates are very crude as the writer has had to make assumptions regarding areas and volumes involved and sources and availability of materials.

Some abatement option:

This comprises B i) plus C i) plus continued maintenance of current embankment and Djupsjoen lake outlet. This low cost (few tens of thousands of dollars) will do little to abate AMD and there will be little benefit, but the situation will be prevented from worsening.

Moderate abatement option:

B i)	say	$10,000
B ii)	5 ha @ $5/sq.m	$250,000
C i)	2 km @ $20/m	$40,000
C ii)	20 ha @ $6/sq.m	$1.200,000
C v)	10,000 cu.m @ $6/cu.m	$60,000
C vi)		say $100,000
	TOTAL	$1,660,000

It is anticipated that this abatement option will have a large abatement of possibly 60 to 90% resulting in a significant improvement in downstream quality but not in a return to close to background levels.

Extensive abatement option:

B i)		say $10,000
B iii)	5 ha @ $20/sq.m	$1,000,000
C i)	2 km @ $20/m	$40,000
C iv)	Embankment	say $2,000,000
	Dredging 200,000 cu. m @ $5	$1,000,000
C v)	10,000 cu.m @ $6/cu.m	$60,000
C vii)	40,000 cu.m @ $10/cu.m	$400,000
	TOTAL	$4,510,000

The abatement achieved by this alternative should be very high, with only low levels of contaminated seepage from the impoundments resulting mainly from the groundwater leaching of contaminants stored in the tailings and waste rock.

5.2 Kongens

5.2.1. AMD Impact

The Kongens mine, located as shown on Figure 5, has a history and impact that are similar to those for Storvarts. The mine was closed in 1972. AMD from the Kongens, Sextus and Leirgruvebakken mines flow into the Orva river, which is highly impacted, and then into the Glama. This discussion is confined to the Kongens mine.

The underground workings of the Kongens mine have flooded and was observed to be yielding a substantial discharge (approximately 20 l/s) at one location. The quality of this mine drainage is as follows:

Copper	16.6 mg/l
Zinc	30.9 mg/l
Iron	135 mg/l
Sulphate	648 mg/l
pH	2.8

Mine wastes have been widely distributed over the mine site, similar to Storvarts. AMD from some of the waste areas are draining towards Orvsjoen Lake. This lake also receives AMD from Sextus and nei Leirgruvebakken mines and discharges to the Orva river.

The outlet of lake Orvsjoen has the following quality (NIVA, 1980):

Copper	0.095 mg/l
Zinc	0.71 mg/l

Approximately 1 million tons of floatation tailings have been deposited in a tailings dam constructed of till as a side hill impoundment. The maximum embankment height is about 10 m and the area was estimated to be about 15 ha. AMD from the mine, from some waste areas and the mill site are being channelled into the tailings dam, from where it is discharged without treatment to the Orva river. The quality at the outlet is as follows (NIVA, 1980):

Copper	1-20 mg/l
Zinc	35-133 mg/l
Iron	14-261 mg/l
Sulphate	430-1860 mg/l
pH	2.8

Seepage is occurring through some of the embankments and foundation soils and acidic kill of vegetation is apparent on some of the down hill slopes.

The total loads to the Orva river from the Kongens mine has been estimated (NIVA, 1980) at:

Copper	1.8 tonnes/year
Zinc	16.1 "
Iron	47.5 "
Sulphate	237 "

With the contributions from the other mines loadings are more than twice this with the result that the Orva river is devoid of fish.

5.2.1 Alternative Abatement Options

The evolution of acid generation in the tailings impoundment may still not have reached its maximum release rate. The back end of the tailings impoundment, where the coarsest tailings have deposited, appears to be relatively permeable and the release of acidic products is expected to increase with time.

The alternative abatement options that may be considered for the Kongens mine are similar to those for Storvarts. The major differences are:

AMD from the mine:

Abatement measures are required for the acid mine drainage. No information was available on the layout of the mine and its workings during the site visit. Typically abatement measures that can be considered include installation of mine plugs to increase the percentage of the mine that is flooded; plugging of mine openings to reduce entry of water and air; grouting with a filler or basic material; collection and treatment.

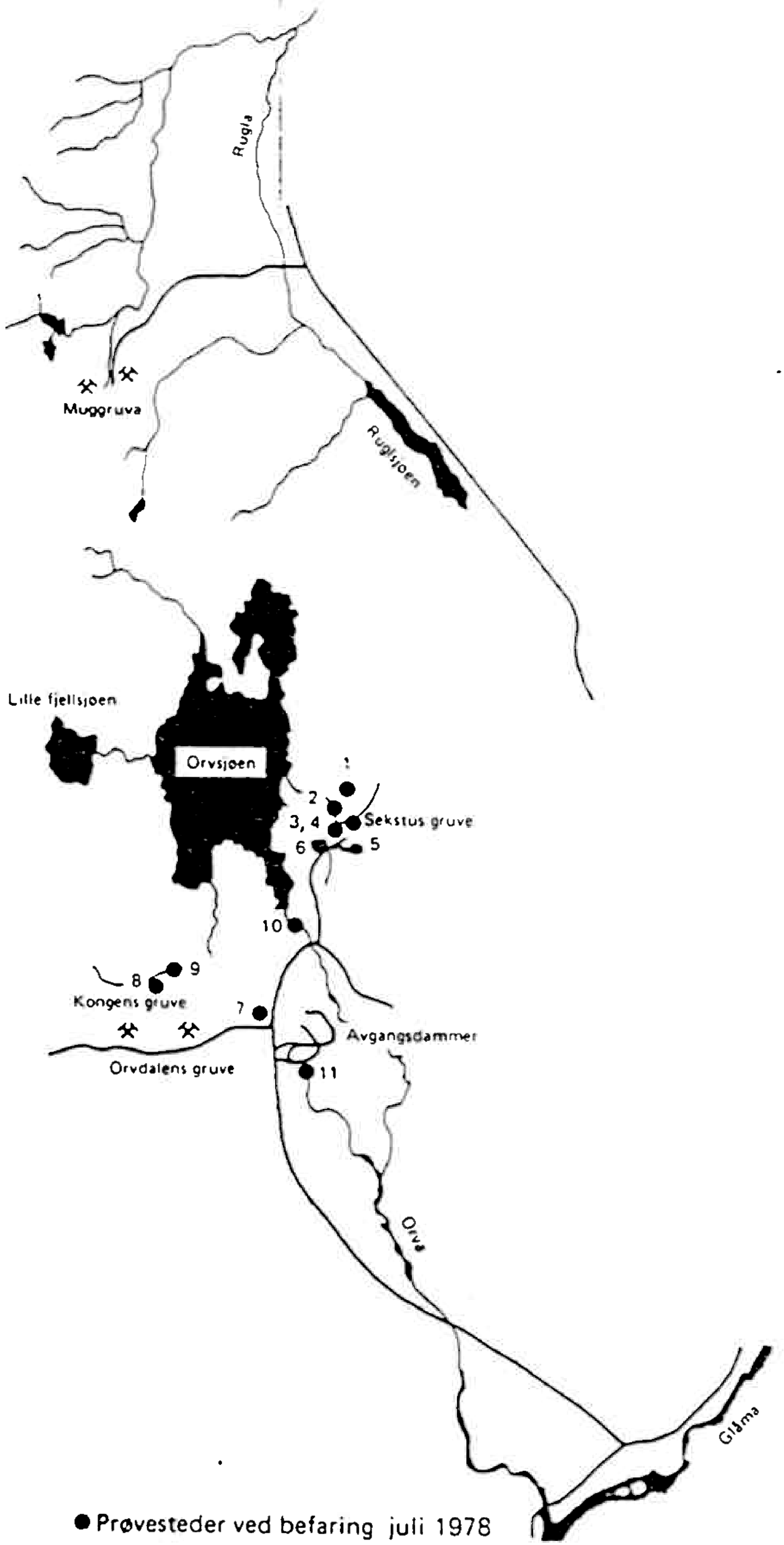


Fig. 5 Location of Kongens (after NIVA, 1980)

Tailings:

The tailings embankment is a side hill embankment of a quality unsuitable for the safe retention of water for a flooded water cover. Covers that may be considered for the tailings are therefore:

i) till,

ii) till plus waste to develop a saturated zone or bog above most of the tailings, similar to that described by Robertson, 1987,

iii) synthetic liner plus till cover.

With each of i) and ii) there will be a significant seepage though the tailings and transport of the acidic products currently stored in the tailings. This will limit the initial effectiveness of these abatement measures. In view of the high permeability of the back end of the tailings impoundment the initial effect may be to increase AMD. With iii) infiltration from the surface will be reduced to a small fraction of the current value and oxygen exclusion at the surface will also be effective. Groundwater leaching will continue with all three options. The tailings embankments will have to be stabilized to ensure long term stability.

Consideration may be given to developing a series of paddocks on the tailings surface, using till for construction, in which wetlands (bogs) are developed to both seal the underlying tailings (from oxygen) and provide some wetland treatment for the acid mine drainage. The author is unable to anticipate the effectiveness of such a system.

Consideration can also be given to the removal of all surface deposits of tailings and mine wastes to Orvsjoen Lake for below water disposal. This lake is already impacted by other wastes deposited in it and AMD to it.

5.2.3 Cost/Benefit Estimates

In the absence of any mine data, no estimates can be provided of the costs/benefits of acid mine drainage abatement.

Assuming that the volumes of wastes and tailings are similar for Storvarts and Kongens, the abatement costs for equivalent abatement measures will be essentially similar. Because of the greater consolidation of tailings in the Kongens tailings impoundment (hence smaller area), the cost of similar covers will be a little less. The effectiveness of abatement of the AMD from the impoundment will be more difficult to achieve because of the seepage through the tailings embankments and foundation soils.

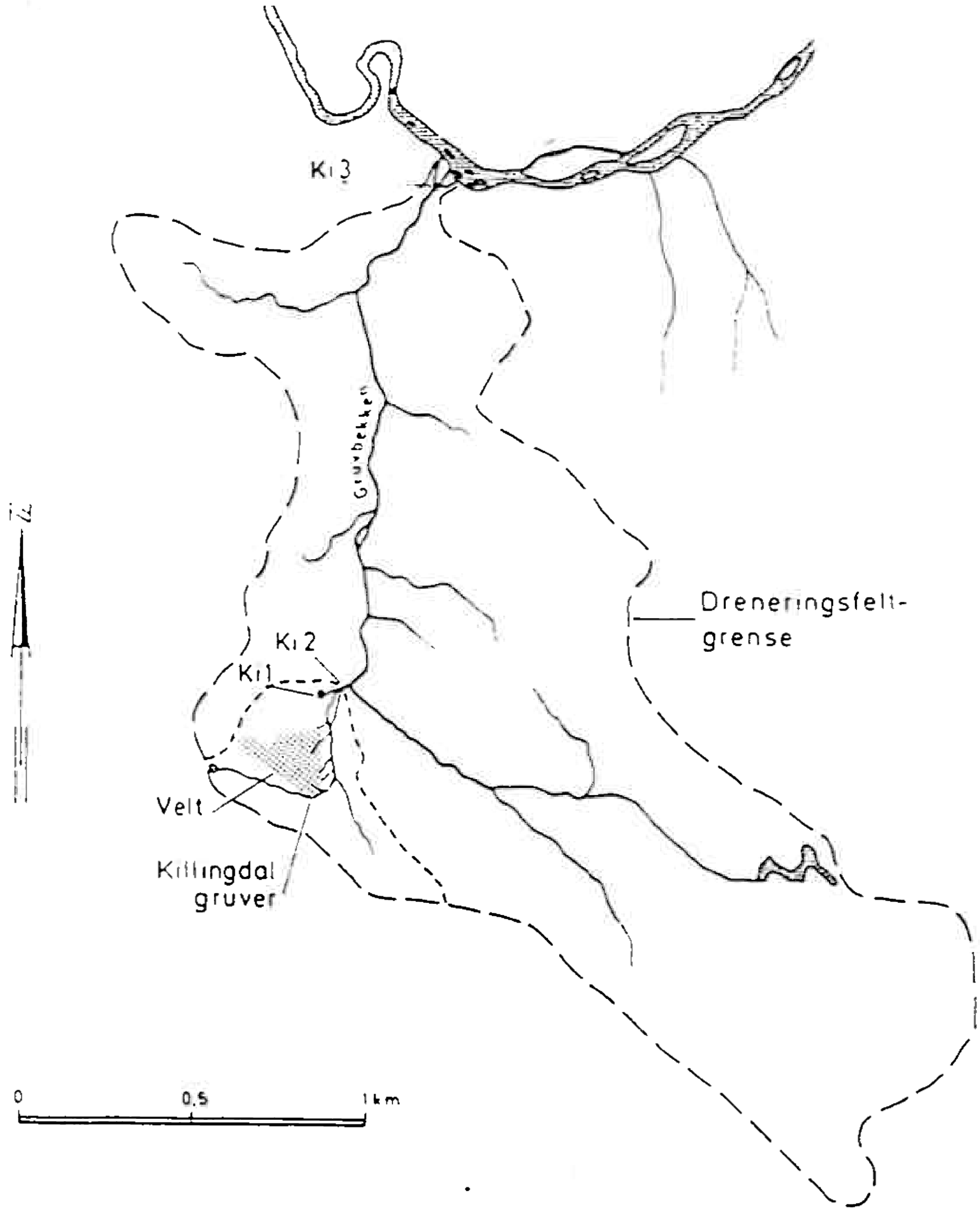
The cost of lake disposal will be high, estimated at about $6/cu.m for all wastes and tailings. During deposition of the contaminant loaded wastes in the lake it will be necessary to treat and control the discharge from this lake. Long term control is expected to be effective.

5.3 Killingdal

5.3.1. AMD Impact

The mine is located as shown on Figures 3 and 6. Two mining areas are involved; the Nordre mine started in 1674 and abandoned in 1968 at the top of the mountain, and the Sondre mine started in 1793 and closed in 1986. The two mines are connected and extend to a depth of about 1500 m, as illustrated in Figure 7. Ore mined comprised 1 to 2% copper, 3 to 8% zinc and 20 to 50% sulphur. Since 1953 the ore from the mines have been dressed at the companies flotation plant in Trondheim.

Fig. 6 Mine Location - Killingdal (after NIVA, 1979)



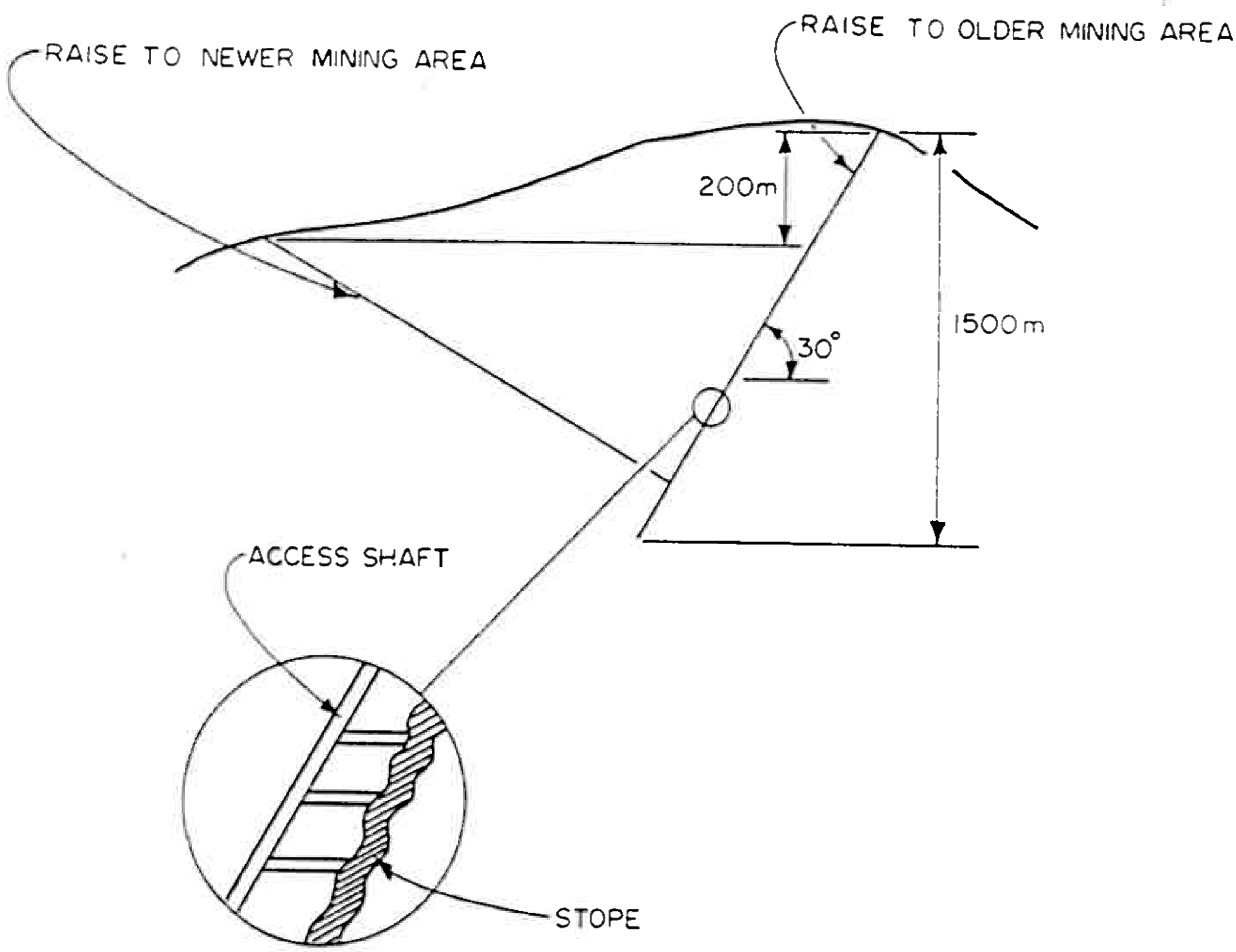


FIGURE 7. SCHEMATIC SECTION THROUGH KILLINGDAL MINE

Three primary sources of AMD exist.

i) AMD from the adit just below the entry to the upper mine, Ki 1 on Figure 6. Average flows from this adit (NIVA, 1979a) are reported to be 0.2 l/s with the following average quality:

pH	2.4 to 2.9
Copper	80 mg/l
Zinc	385 mg/l
Iron	1175 mg/l
Sulphate	4650 mg/l

This yields a total annual load of about:

Copper	0.2 tonnes/year
Zinc	1 "
Iron	3 "
Sulphate	12 "

The typical mine development consists of an incline shaft at about 30 degrees in the hanging-wall with cross-cuts to the stope. The stopes are mainly room and pillar open stopes, though waste has been placed in some of the older stopes.

ii) Mine drainage used to be pumped from the mine (NIVA, 1979a) at an average rate of 0.033 l/s (personal communication with mine personnel) from below the current raise intersection with old raise (see Figure 7). This pumped water had a concentration and annual loading as follows:

	Concentration	Load
Copper	625 mg/l	1.2 tonnes/year
Zinc	7550 mg/l	13 "
Iron	6150 mg/l	11 "
Sulphate	43000 mg/l	75 "

This loading represents about one third of the total AMD from the mine. Discharge has now been discontinued as the mine has ceased pumping. The mine has done calculations that indicate that the evaporation from the shaft caused by natural ventilation exceeds the inflow rate and have concluded that there will not be a long term accumulation of water in the mine. The author has not reviewed these calculations, but questions why pumping was required in the first instance, if evaporation exceeded inflow. It is noted that the inflow is very small and consequently the rate of mine filling would be slow. The period before any discharge will occur is therefore long, if ever.

iii) AMD from the waste dumps.

Approximately 150,000 cu m of waste rock has been placed in piles near the upper (old) mine entry. The waste varies considerably in nature, including coarse and fine wastes in discrete piles, burned or roasted wastes and newer very coarse waste. An area of approximately 4 ha is covered in wastes or is heavily contaminated by AMD and erosion products. Seepage from the dumps has approximately the following concentrations and annual loading:

	concentrations	load
pH	2.6 to 2.9	
Copper	40 mg/l	2.5 tonnes/year
Zinc	175 mg/l	12 "
Iron	484 mg/l	63 "
Sulphate	2300 mg/l	163 "

To the extent that the above concentrations and loadings are representative of the annual values, it is apparent that the majority of the AMD loading results from the mine wastes.

AMD from the mine drains into Gruvbekken which flows into the Gaula River 2 km below the dumps. The water quality in the creek at the confluence is as follows:

pH	3.0
Copper	1.83 mg/l
Zinc	8.5 mg/l

Water quality in the Gaula at different distances downstream from the confluence are as follows:

	500m	6 km	10 km	20 km
pH	6.5	6.8-7.2		
Copper	0.2	0.05-0.11	0.02-0.09	0.002-0.006
Zinc	0.1-4.	0.12-0.4	0.1-0.2	0.005-0.01

The concentrations are such that the Gaula does not support fish for a considerable length (> 10 km).

5.3.2 Alternative Abatement Options

Since the major AMD loading to the Gaula appears to be from seepage from the waste dumps, the abatement measures may focus on this. Three alternative abatement measures may be considered.

i) Backfill to the mine

The natural mine ventilation should be maintained, if this is required for control of the mine drainage. It may be feasible to do this by placing the waste in the old stopes leaving the raise free for ventilation. Since approximately 3 million tonnes of ore was extracted from the mine there is ample room for the waste. Placement in the old stopes will require rehabilitation of the old raise to allow the transport and tipping of the wastes into the stopes.

ii) Water cover in a dam on Gruvbekken

A dam could be constructed on Gruvbekken with a basin volume sufficient to contain the relocated wastes and maintain a water cover. During relocation of the wastes to the basin, lime would be mixed with the wastes to neutralize any stored acidity. Following waste placement, the surface of the waste would be covered with a thin, 0.3 m, layer of low permeability soil or till to prevent convective transport of dissolved oxygen to the waste. A shallow but secure water cover would be sufficient to considerably abate acid generation.

iii) Consolidate wastes under a synthetic membrane cover.

The wastes would be consolidated in a single pile contoured to promote drainage and this would be covered with a synthetic membrane liner installed on a suitable bedding material and covered with a 0.6 m protective till layer which would be revegetated. Drainage ditches would divert surface flows around the pile.

Abatement of the AMD from the upper mine adit depends on the sources and mine layout, which were not reviewed during the visit.

5.3.3 Cost/Benefit Estimates

i) Mine backfill

Cost estimate:

Raise rehabilitation		$400,000
Mine preparation		$200,000
Waste placement	150,000 cu.m @ $10	$1,500,000
Reclamation	4 ha @ $5,000	$20,000
	TOTAL	$2,120,000

Since this abatement measure effective eliminates the transportation mechanism, it has a high potential for achieving an adequate level of abatement.

ii) Water cover

Cost estimate:

Construction of embankment		
	assume 100,000 cu.m @ $6	$600,000
Placement of wastes	150,000 @ $4	$600,000
Lime and mix	7,500 tonnes @ $100	$750,000
Cover placement	20,000 cu.m @ $6	$120,000
Spillway		$100,000
Reclamation		$20,000
	TOTAL	$2,160,000

The water cover is considered to provide effective abatement of additional acid generation. However the release of contained acidity and soluble contaminants must be prevented by neutralization and treatment during the period of waste relocation.

iii) Synthetic membrane cover

Cost estimate

Consolidation of wastes	50,000 cu.m @ $4	$200,000
Contouring and ditching		$ 50,000
Placement of membrane & cover	40,000 sq.m @ $20	$800,000
Vegetation	4 ha @ $2,000	$ 8,000
	TOTAL	$1,058,000

Since the membrane prevents infiltration and AMD transport, it is considered that the abatement will be effective. Because of the location of the pile it is anticipated that there is little potential for groundwater leaching. Provision must be made for long term replacement.

5.4 Kjoli

5.4.1. AMD Impact

The location of Kjoli mine is indicated on Figures 3 and 8. Mining occurred in 1766-98 and 1857-68 when copper was extracted and 1896-1907 and 1910-1920 when pyrite was exported. The principal dumps were established in 1986-1920 and have a total volume of approximately 80,000 cu.m (NIVA, 1987).

AMD is occurring from the underground workings and from the waste dumps. NIVA conducted water quality sampling and testing of these drainages in 1977-78, (NIVA, 1979b), and the results are summarized as follows:

	Mine drainage			Mine plus waste dumps		
	Concentration		Load	Concentration		Load
Flow	2.5-7	l/s		3.6-60		l/s
pH	2.8-3.1			2.8-3.1		
Copper	2.6	mg/l	0.4 tonne/year	6.6	mg/l	2.7 tonne/year
Zinc	0.2	mg/l	0.05 "	0.2	mg/l	0.2 "
Iron	40	mg/l	5.3 "	57	mg/l	27 "
Sulphate	260	mg/l	36 "	340	mg/l	140 "

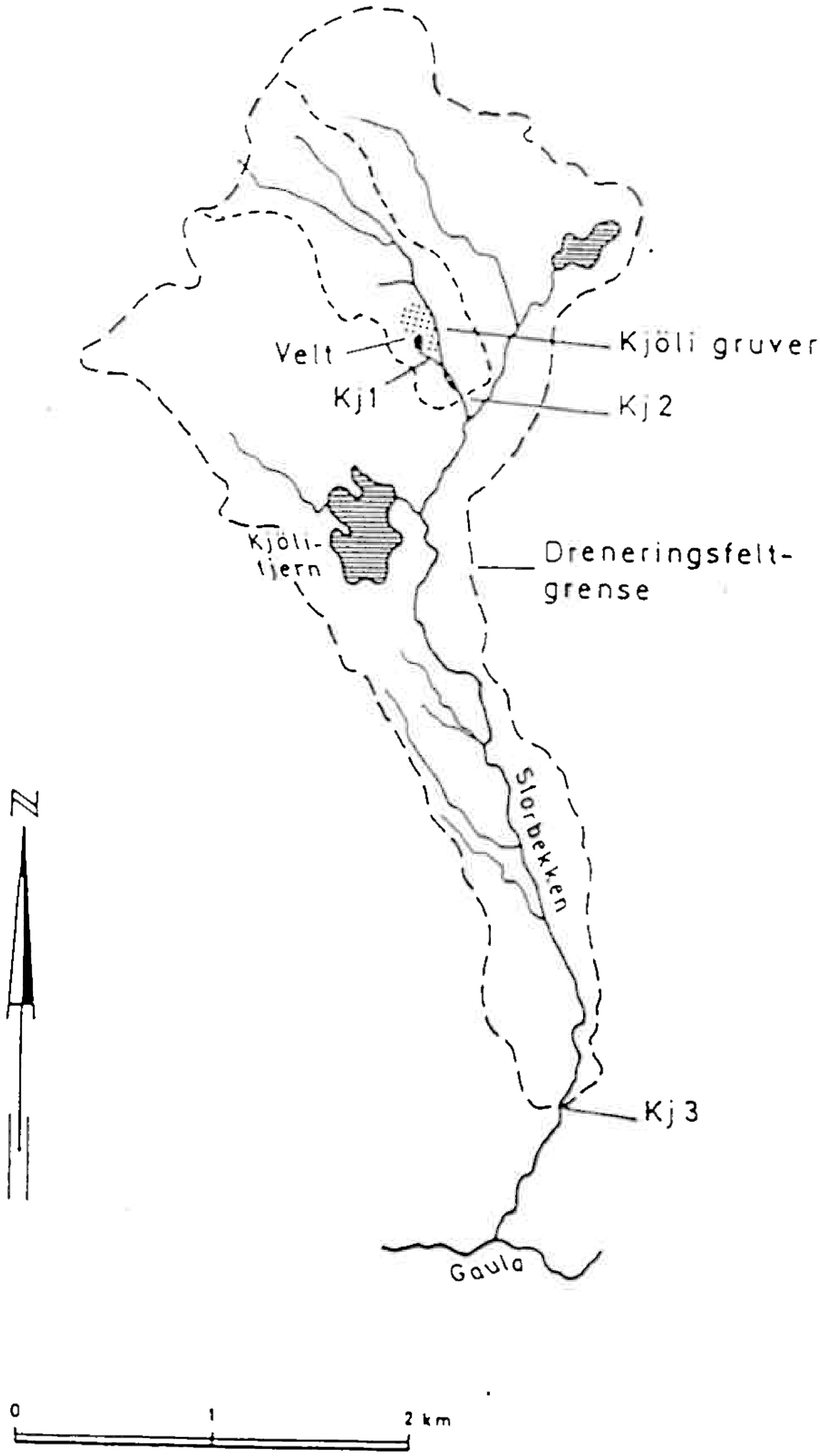
According to this data the AMD load is primarily from the waste dumps. During the field visit it was noted that the mine drainage appeared to be significantly greater than the maximum reported flow; an observation that must be checked and which illustrates the need for monitoring to cover the full annual cycle. Drainage from the mine and dumps flow into Storbekken and then into the Gaula river where it contributes to the impact caused by AMD from Killingdal mine as discussed in NIVA, 1979b.

In an attempt to improve AMD the Norwegian Department of Industry undertook abatement measures in 1981. The dumps were consolidated, contoured to a gentle, uniform slope and 100 tons of hydrated lime was sprayed onto the 2 ha surface of the dump. These abatement measures are reported in NIVA, 1986. The results were unexpected. AMD has increased each year since 1981. This is illustrated by the following average annual copper concentrations and loads:

	Concentration mg/l	Load tonnes/year
1977-78	7.26	4.2
81	11.0	4.9
82	13.0	5.7
83	24.8	10.9
84	22.2	13.1
85	45.5	23.4
86		
87		

It is the author's opinion that this increase in AMD is primarily due to the disturbance of the waste, resulting in the establishment of new infiltration and seepage paths, which enables accumulations of acidity and leach products to be transported out of the dumps. The surface application of lime had little effect on the total stored acidity. The disturbance of the piles would have exposed fresh surfaces and provided new pathways for air and water entry for additional acid generation, but this is considered a secondary effect. It is considered that the increase in the transportation mechanisms has a greater effect than the increase in the acid generation mechanisms and these two effects are together considerably greater than the control afforded by the surface application of the lime.

Fig. 8 Location Plan - Kjoli (after NIVA, 1979b)



5.4.2 Alternative Abatement Options

Mine drainage:

Current mine drainage is issuing from an adit at the 1047 m level. NIVA, 1986, indicates that by plugging this lower adit it may be possible to flood about 50% of the underground workings up to the 1064 entry level. The potential for additional plugging should be investigated.

Waste Dumps:

Two alternatives are considered appropriate for this project.

i) Synthetic membrane cover

This alternative would involve the diversion of all surface drainage around the waste dumps; covering it with a 10 cm layer of (till) bedding material; placing a 2 mm HDPE membrane cover; covering this with a 0.6 m till protective layer and establishing vegetation on the cover.

ii) Water cover in Kjolitjern lake

This alternative would require the dewatering of the lake, followed by the removal of all waste to the lake where it would be mixed with lime in sufficient proportions to neutralize the contained acidity before placement below the ultimate water level. Following placement the waste would be covered with a thin layer of fine grained till to prevent convective transport of dissolved oxygen into the waste, and the lake allowed to flood. The water quality at the outlet from the lake will be controlled by lime addition and settling prior to discharge, until the lake water quality improves sufficiently to allow direct discharge.

5.4.3 Cost/Benefit Estimates

Mine plugging:

The geometry of the mine outlets is not known and an allowance of $150,000 is made for the installation of an appropriate plug or plugs.

Waste dumps:

i) Synthetic membrane

Cost estimate

Diversion ditches		$40,000
Bedding, membrane & cover	20,000 sq.m @ $20	$400,000
Vegetation establishment		$5,000
	TOTAL	$445,000

Allowance should be made for the cost of long term replacement of the membrane. The effectiveness of the abatement will depend partly on wether any groundwater leaching is occurring. In the absence of any groundwater leaching, it should be very effective, since the transport mechanism is essentially eliminated.

ii) Water cover

Cost estimate

Lake drainage and water control	allow	$100,000
Waste removal to lake	80,000 cub.m @ $4	$320,000
Lime addition	4,000 tonnes @ $100	$400,000
Till cover placement	10,000 cub.m @ $6	$60,000
	TOTAL	$ 880,000

During waste placement in the lake the water quality in the lake will have to be controlled by treatment. Once the water quality has improved to the extent that it can be discharged without treatment, the abatement should be effective and long term. Little or no long term maintenance should be necessary.

5.5 Lokken Verk

5.5.1. AMD Impact

The location of the Lokken mine is shown on Figure 9 (Burridge, 1985). A section through the mine is shown in Figure 10, and Figure 11 shows the location of the mine waste deposits and tailings (NIVA, 1983). Mining of this deposit started in 1654 and about 20 million tonnes of ore grading 2.5% copper and 1.5% zinc has been extracted. Present mining is based on the Astrup shaft.

The mine is located on the steep slope of the west embankment of Raubekken stream, in a highly populated area. AMD draining to Raubekken, as measured at station 7 (see Figure 11) carries an annual contaminant load of 42 tonnes copper, 54 tonnes zinc, 292 tonnes iron and 4745 tonnes sulphates. Loadings at stations 2,5 and 6 account (NIVA, 1983) for only about 30% of this loading indicating that the majority of the load reaches the stream via groundwater. Raubekken is currently entirely diverted to the Svorkmo Power Station from which it flows into the River Orkla at Svorkmo, at which point dilution is so great as to limit the impact of the AMD. Because of the high cost of abatement at this site, the most economic option may be to maintain the current diversion of Raubekken. Concern exists if such diversion cannot be continuously maintained.

Previously some 500,000 cu.m/year of mine water drained from the Wallenberg workings into the Astrup mine. Any mine water to be disposed of was discharged via gravity flow through a wood stave pipe to the sea. This drainage was cut off by installing three plugs in the drifts draining into the Astrup mine. The old mine workings are expected to flood to the outlet level near Fearnley shaft in about two years, potentially adding to AMD.

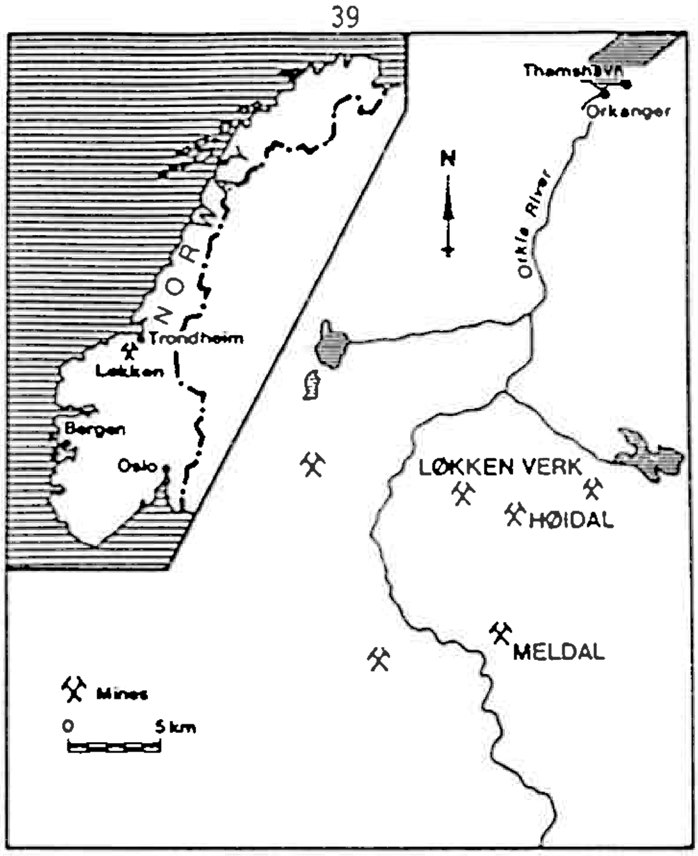
The various areas of AMD concern are described and discussed in the next section.

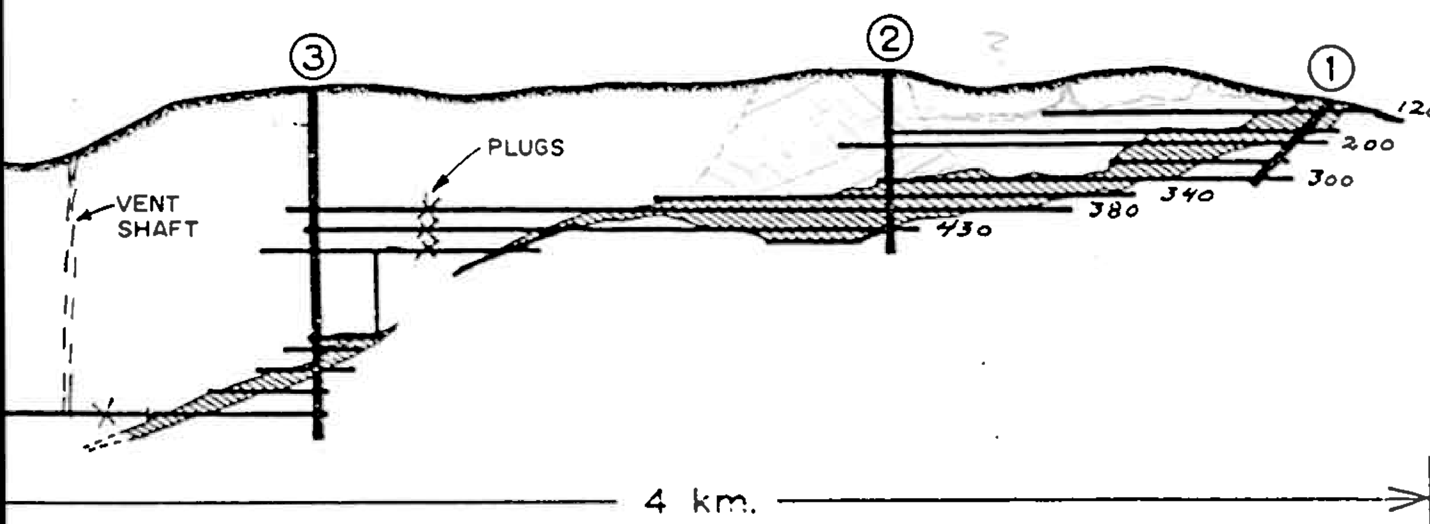
5.5.2 Alternative Abatement Options

i) AMD from the old mine workings.

Mine collapse has occurred over the old workings and it is not feasible to seal inflows to the mine. Discharge will ultimately occur at the quality of the surface waters in the mine. Sampling of the surface water in the Wallenberg shaft indicates an improvement of the water quality as it rises (communication with mine personnel). This may be due to the establishment of a contaminant concentration gradient in the mine. Should such a concentration gradient remain stable then it is possible that the quality of the water draining from the mine will be considerably better than the average in the mine. There will be a flow gradient from the recharge areas towards the outlet and this may cause upward displacement of the poorer quality water. The potential for this may be checked by installing submerged sampling

Fig. 9: Sketch map of Løkken district with (inset) a map of Norway showing Løkken's locations.





① FEARNLEY SJAKT
② WALLENBERG SJAKT
③ ASTRUP SJAKT
X TETT STØP
TALLENE ANGIR GRUBENS HOVEDNIVÅER

VANNSTAND 01 01 84 : 423 m.
" 85 : 379 "
" 86 : 354 "
" 87 : 324 "

Fig. 10 Section Through Lakken Mine

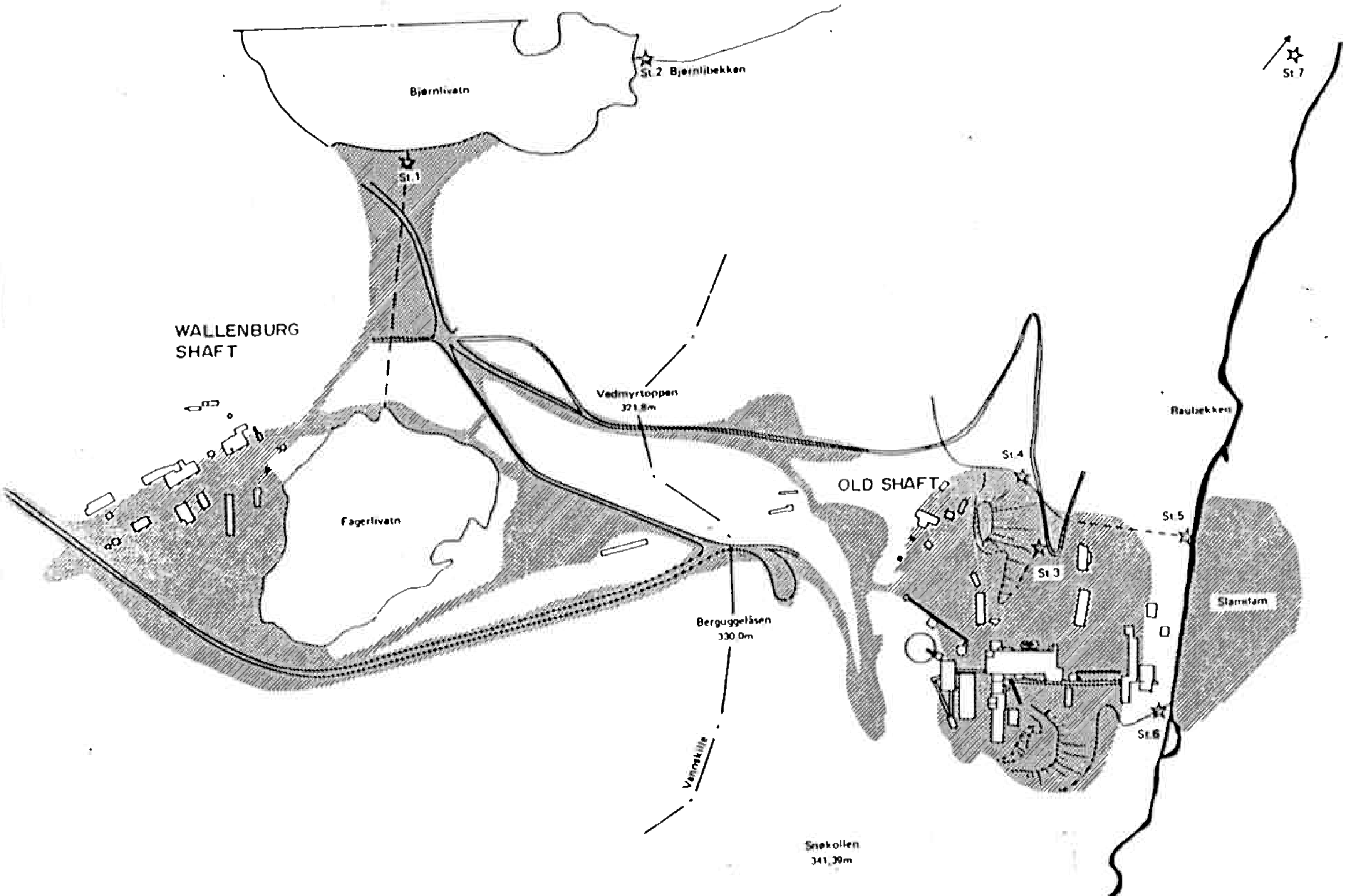


Fig. 11 Location of Mine Wastes and Tailings at Lokken Verk (New Tailings Dam Not Shown)

points in the Fearnley shaft between the 200 and 120 levels and conducting mine surface water pumping tests prior to mine water reaching its discharge level.

The time to discharge can be delayed for a few years by allowing mine water to drain to the Astrup mine, but this will require sealing of the vent shaft in the Orkla valley.

Experience of AMD from other flooded mines suggests that contaminated drainage can be anticipated, though the quality may improve due to the flooding of most of the exposed surfaces in the mine. Provision may be made for the long term collection and treatment of this drainage or re-implementation of the sea disposal system. The volume and quality may be improved by sealing off inflows and increasing the level of flooding by installing additional plugs. The potential for this is not known to the author.

Collection and sea disposal or treatment involves conventional technology and is effective but requires long term care and maintenance.

ii) New tailings area

New tailings are placed in tailings impoundments 2 km west of the mill. The embankments for these impoundments are constructed of apparently non-acid generating coarse float tailings with an upstream facing of filter fabric and cement plaster. The cement plaster was observed to be fractured and cracked, resulting in considerable seepage through the embankment. Tailings are placed under water and against the upstream facing to help seal the embankment. It is understood that these tailings will be abandoned in a flooded condition to prevent acid generation. It would be appropriate to evaluate the long term stability of the embankment facing and the potential for piping under a long term water cover. A water cover is considered to offer good, long term control of acid generation in these tailings.

iii) Old tailings

Approximately 3 million tonnes of ore washings, old flotation tailings and metal hydroxide precipitates have been placed in various contiguous impoundments constructed on the east bank of Raubekken. Recently unsuccessful attempts were made to process these old tailings. At the time of the visit tailings were being excavated and moved to terrace the tailings area to form a parking lot and playing fields. An inspection of some of these old tailings (3 to 10 years old) indicated that oxidation had penetrated only a few tens of centimeters into the mass of most of the tailings, though penetration to 2 meters was observed locally in dessication fractures. In areas of high water table the penetration was even less. Along embankments, such as along Raubekken, extensive oxidation has occurred. Strong AMD from surface drainage and seepage is apparent. Unless this is abated it may be expected to get worse as the dessication and oxidation front penetrates further.

Abatement measures that may be considered are:

a) Dredging and pumping (or hauling) to the new tailings impoundments for disposal with a water cover. This is an extremely high cost option but provides effective cleanup, in the long term, for Raubekken.

b) Stabilization in-situ with a synthetic membrane cover. This would involve the construction of stable embankments along Raubekken; contouring and covering the tailings with a synthetic membrane with suitable cover and reclamation. Surface waters will be diverted around the tailings area in diversion ditches. While considerable AMD abatement will result there will still be significant groundwater leaching of the tailings and tailings contaminants because of the tailings location relative to groundwater discharges. Thus there will continue to be considerably reduced but significant contaminant yields to Raubekken.

c) As for b) except that the upper surface of the tailings will be covered in sandy soils to allow the accumulation of surface water,'bog', or subsurface 'ponding' of water on the tailings to prevent air and oxygen entry. Because of the increased hydraulic gradient and infiltration through the tailings, the contaminant yield from this option will be significantly greater than for b).

d) Canalization of Raubekken, from above the AMD impacted zone, to isolate it from the AMD; and the collection of all AMD reaching the valley bottom which would then be treated or discharged to the sea. A possible canalization and collection drain section, for installation in the valley bottom, is illustrated in Figure 12. Installation of the canal and drains in the narrow valley bottom will cause some inconvenience. It may not be possible to maintain isolation during large flood flow periods. The measure should be effective in that it provides not only for AMD from the tailings and known surface sources but also AMD from all other sources discharging via the subsurface to Ruabekken. Long term maintenance of the system will be required.

iv) Waste dumps adjacent to mill

Approximately 3 million tons of mine and mill wastes have been deposited on the steep slopes adjacent to the mill. These dumps are standing at their natural angle of repose. Attempts to cover and revegetate them has been successful for their relatively small upper surfaces but unsuccessful on the large sloping surfaces. The hill side slopes are generally too steep for the dumps to be flattened to enable either soil or synthetic membrane covers to be placed. AMD abatement options that may be considered include:

a) Removal and placement in the new tailings impoundment with water cover. While this is considered an effective long term abatement measure it is also extremely costly.

b) Consolidate isolated acid generating waste piles; install ditches for surface diversion of uncontaminated water; cover upper surfaces and any surfaces that can be adequately sloped (3 horizontal to 1 vertical) with synthetic membrane to reduce infiltration, flatten remaining slopes as much as possible and, where possible, place a low permeability soil cover, using cable tools on the slopes. While some abatement will occur it will be marginal. This alternative considers those measures that can be relatively easily achieved, but benefits are minimal.

c) Canalize Raubekken and collect AMD as per iii) d) above. Comments on abatement are also similar.

v) Waste surrounding Fagerlivatn and Bjornlivatn

Large quantities of waste and hand picked ore surround Fagerlivatn and lie along the drainage to Bjornlivatn. The coarse float tailings appears to be non-acid generating and is placed above the acid generating hand picked ore. Some of these dumps are currently being covered with a shallow layer of peat and organic soils to establish a vegetation cover. It is considered that such revegetation will do little to prevent AMD. Fagerlivatn received mine water in earlier years and may contain accumulated sludges. A seep survey and dump acid generation survey should be conducted on all wastes to identify which are and which are not acid generating. Abatement measures that may be considered include:

a) Synthetic membrane covers as previously discussed for other sites. While this will prevent infiltration and oxygen entry, it will not prevent groundwater leaching. The resulting abatement is not readily anticipated but it should result in a significant improvement.

b) Water cover, achieved by raising the level of the outlet to Fagerlivatn and moving wastes below the flood level. Submerged waste piles should be neutralized to stabilize stored acidity and covered with a layer of soil to prevent convective transport of oxygen into the waste and rapid

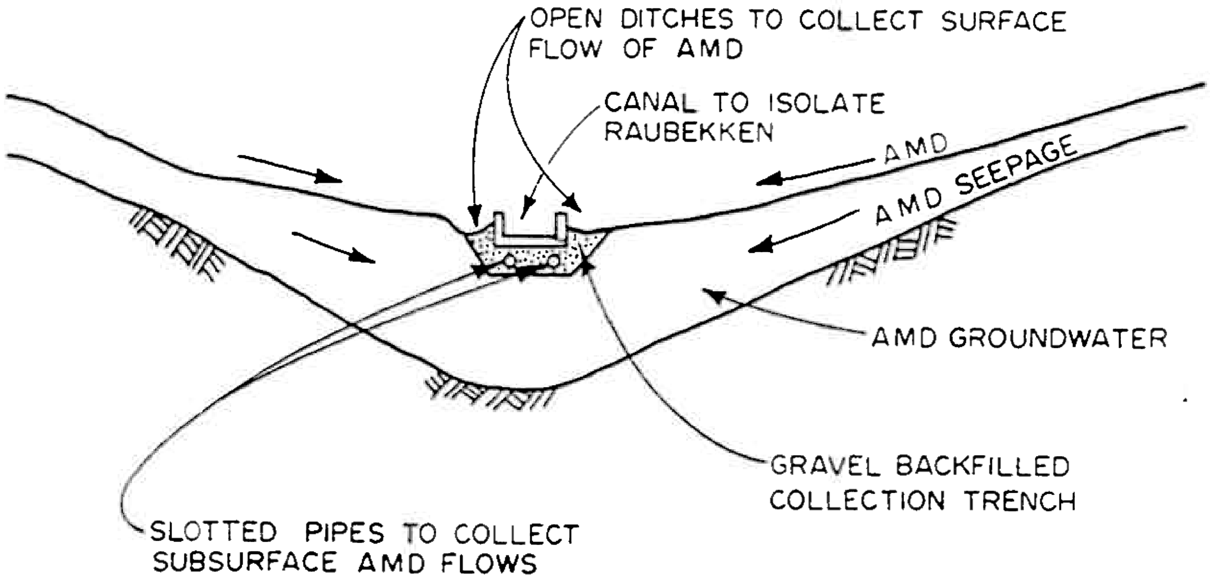


FIGURE 12. CROSS SECTION THROUGH RAUBEKKEN ISOLATION CANAL AND AMD COLLECTION SYSTEM

leaching of stored contaminant products. To allow construction it may be necessary to temporarily reduce the water level in the lake by pumping. Because of the large contaminant load in the submerged wastes it will take some time for the water quality to improve. The long term control should be effective.

c) Collection of drainage from these lakes and treatment or discharge to the sea.

5.5.3 Cost/Benefit Estimates

The areas impacted are large and complex and a cost estimate for alternative abatement options is beyond the scope of such a conceptual assessment. An assessment of the potential effectiveness of each of the abatement measures has been included in the previous section.

5.6 Skorovas Gruber

5.6.1. AMD Impact

The location of Skorovas mine is shown on Figure 13. Mine production started in 1952 with ore averaging 37% sulphur, 1.1% copper and 2.2% zinc at a production rate of 500 tonnes per day. AMD from the mine site impacts two different drainages as indicated on Figure 13. The mine has recently shut down.

Mine water is drained at an average rate of about 2 l/s from an adit (originally the exploration adit) to Stallvikelva creek which drains to Tunnsjoen Lake, 10 km from the mine. Tunnsjoen lake, at 90 sq. km, is the 5th largest lake in Norway. Stallvikelva creek is heavily polluted and the contaminant discharge to the lake in 1985 was approximately as follows (NIVA, 1987a):

Copper	15 tonnes/year
Zinc	30 "
Iron	50 "
Sulphate	800 "

Impact on Tunnsjoen lake is confined mainly to Stellvekelva Bay at stations A10 and A29 where copper levels of about 0.015 mg/l and higher are measured (NIVA, 1987a). The effect on fishing is difficult to determine as the regulation of the lake for hydroelectric purposes has a large impact on the lake fisheries.

A waste dump of about 600,000 tonnes has been dumped on the steep sloping topography adjacent to the mine. Drainage water from this waste dump, outcrop, mill, and old mill tailings drain into Dausjoen lake (30 ha - 4 million cu.m). In the 1970's the heavy media tailings were placed in the lake; later dozed into the lake and now form a terrace 1 m below the surface of the lake. A diversion ditch has been constructed (blasted into rock) to divert clean water around the waste dumps. A collection ditch collects a substantial percentage of the low pH AMD from the waste dump and directs it to a liming station at the discharge to Dausjoen lake. Since 1963 floatation tailings have been placed, at a pH of about 8 to 9, into the lake.

Natural acid drainage to the lake, prior to mining, resulted in a dead lake (personal communication with mine personnel). At the exit to Dausjoen lake the liming station adds hydrated lime (approximately 230 tonnes in each of the last two years) to the mine waste and mill drainage and this is discharged to the creek flowing to Store Skorovatn lake (60 ha). The creek acts as a mixing channel and sludges are settled in Store Skorovatn lake. The metals loading in this flow in 1985 was the following:

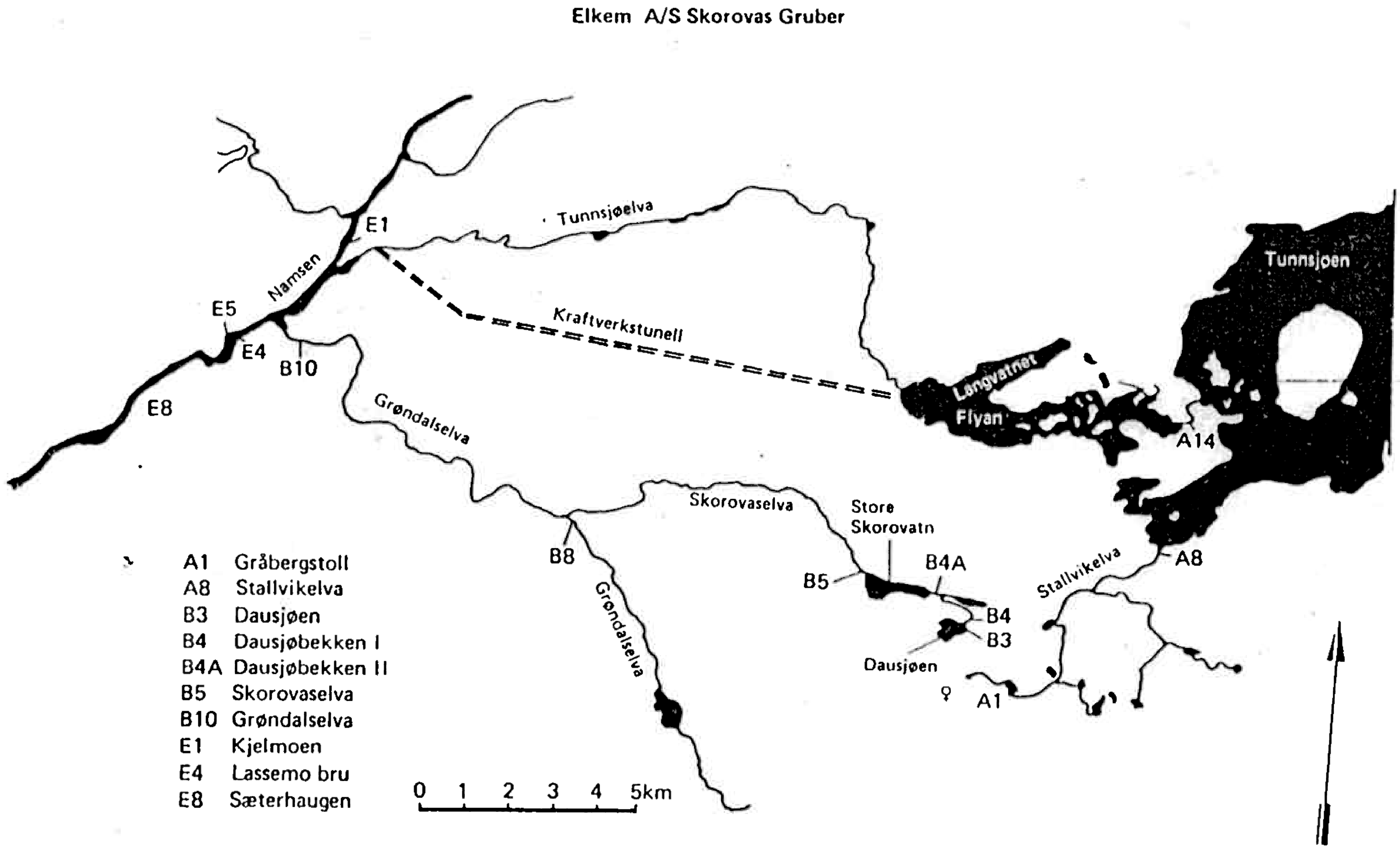


Fig. 13 Location of Skorovas Mine and Sampling Stations

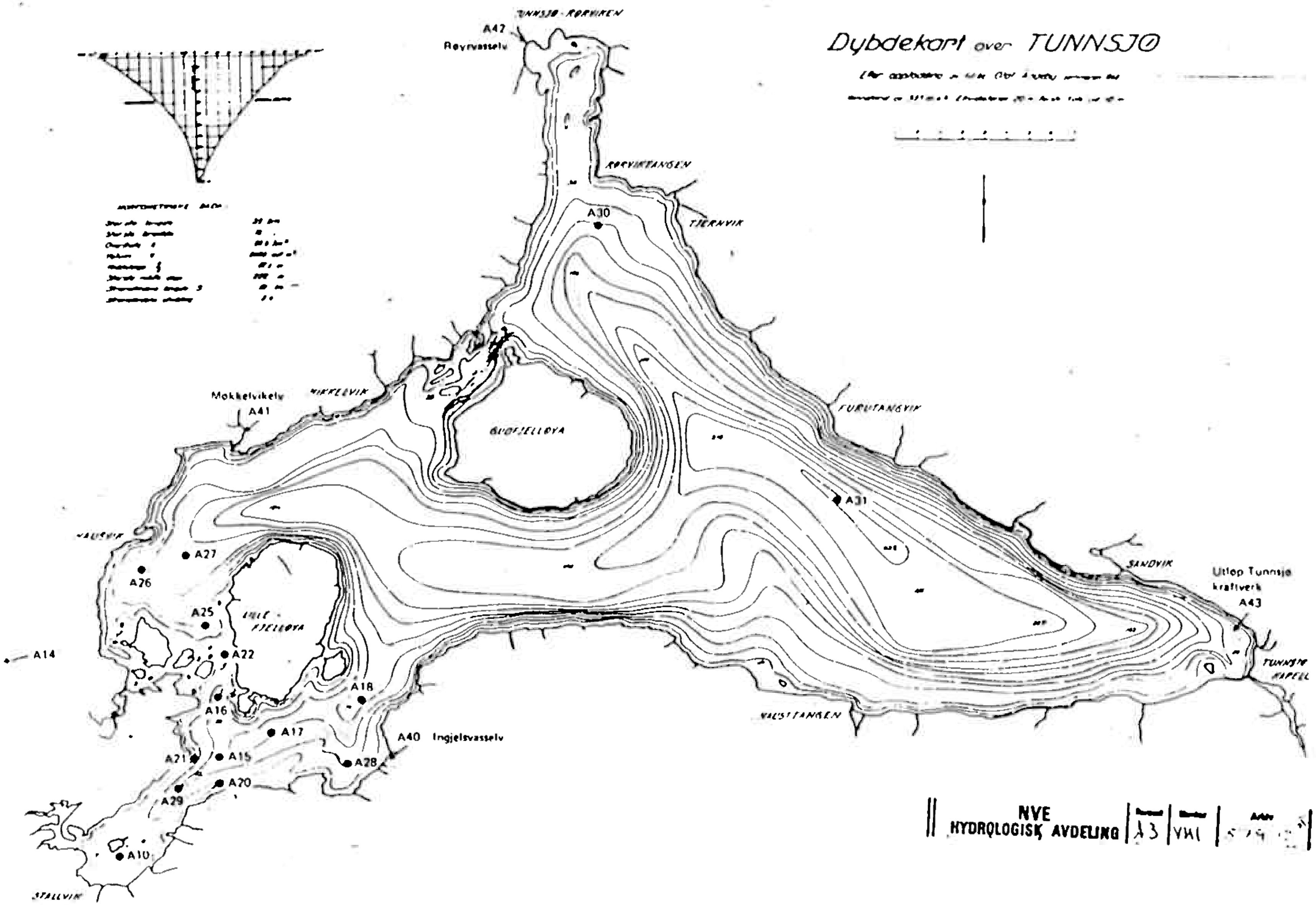


Fig. 14 Sampling Stations in Tunnsjoem Lake

Copper	11.2 tonnes/year
Zinc	35.7 "
Iron	23.0 "
Sulphate	774 "

Reductions may be occurring as a result of mine shut down but insufficient data is to hand to quantify this. Discharge from Store Skorovatn flows to the Namsen river. The NIVA , 1987a, report indicates that water quality in the Namsen was being effected at times particularly prior to 1976 when average concentrations of about 20 and 60 micro grams/l of copper and zinc were experienced at station E4 shown on Figure 13. This report also serves to demonstrate that while copper is settled in Store Skorovatn, zinc tends to be flushed through, possible as a result of dissolution. The high volumes of sludges lining the flow path to the Namsen provide a source for metal loadings should the pH of this drainage ever be allowed to drop.

5.6.2 Alternative Abatement Options

i) Mine drainage

The layout of the mine workings are not known to the author. It may be possible to achieve some abatement by plugging the exploration adit and the main mine entry. The location of discharge would have to be established and the drainage direction and impact evaluated. Fish mortality tests would help to determine the consequences of lake water degradation and hence the need/benefit of abatement on this drainage.

If treatment of AMD is to continue in the Skorovatn drainage, consideration may be given to the diversion of the mine drainage to this drainage. This could be achieved by constructing a ditch along contour from the one drainage to the other.

ii) Waste dumps

a) Synthetic membrane cover

The potential of placing a synthetic Membrane cover on the wastes should be considered. To achieve a stable protective cover on the synthetic membrane, the waste must be sloped at no greater than 3 horizontal to 1 vertical. Because of the steep terrain the required flattening may be impractical - it must be investigated. Alternative protective covers would be very costly. Such a cover would effectively prevent infiltration. Groundwater leaching at the site is probably low, if any. Surface water diversion could be improved by installing more effective lined diversion ditches. Thus the AMD transport could be reduced to a small fraction of the current value, effectively abating AMD. Long term maintenance and replacement would be required.

b) Water cover

The following sequence of events is envisaged. Drain Dausjoen Lake, treating the discharge water. Spread lime in the lake basin. Place waste 'in the dry' to levels which will be covered by water, mixing in lime to neutralize contained acidity. Cover the top and sides of the waste, including the old heavy media tailings, with a thin 30 cm layer of till to prevent convective leaching and oxygen transport to the waste. Allow lake to recover. Some natural acid drainage as well as the AMD from residual mine disturbance will remain and the quality of Dausjoen lake will be poor. If the pH in the discharge from the lake drops appreciably then there may be dissolution of the metal sludges deposited in the creek and Store Skorovatn lake. This potential should be evaluated. Total AMD generation will be reduced considerably but the improvement in the contaminant load to the Namsen requires considerably more evaluation. It may be necessary to continue with chemical treatment of drainage from Dansjoen Lake in the long term.

5.6.3 Cost/Benefit Estimates

Abatement benefits have been reviewed in the previous section and it is not possible to make realistic estimates of abatement cost within the scope of this review.

5.7 Sulitjelma

5.7.1. AMD Impact

A number of mines have been developed over the last one hundred years in the area surrounding Lake Langvatn at the locations shown on Figure 15. Two mines are located to the south of the lake.

Jakobsbakken is an underground, room and pillar mine located at a high elevation above the lake and has been abandoned for about 20 years. The mine dips into the mountain and entry was via an adit a short distance below the outcrop. Subsidence cracking occurs above the old workings. The mine is flooded and AMD is issuing from the adit, estimated at 2 l/s during the visit. It appears that seepage is also occurring from a crack at about 20 meters lower elevation with a flow estimated at about 5 l/s. There is a waste pile of about 50,000 cu.m just below the adit. Adit and crack drainage flows through the dumps and there may be additional discharges not visible below the waste. The AMD from the mine drains to a small lake and then via Jakobsbakken to Granheibekken and Langvatn lake.. The flow in Granheibekken typically has the following quality (NIVA, 1987b):

pH	2.8 to 5
Copper	1.5 mg/l
Zinc	4.8 mg/l
Iron	57 mg/l
Sulphate	426 mg/l

Some of the waste has been used in recent years for road fill and for the construction of a pad at a camp site. NIVA, 1987b, reports that this has resulted in local AMD.

Sagmo Gruve is a newer, 'dry' mine with an associated waste dump. The quality of drainage from this mine has not yet been determined.

A large number of mines are located north of Langvatn Lake. These include Mons Petter gruva, Gilken/Charlotta gruva, Hankabakken gruva and Ny Sulitjelma gruva. These are all interconnected with a strike length of about 3 km and a dip length of about 1.5 km, extending from about 500 m above lake level to 400 m below lake level. There are many entries and openings to the mine complex and a number AMD locations mainly into Giken creek. The typical quality of Giken creek is as follows (NIVA, 1987c):

pH	3.3 to 6
Copper	3.3 mg/l
Zinc	4.9 mg/l
Iron	8.6 mg/l
Sulphate	150 mg/l

It was estimated during the visit to be flowing at a rate of 100 to 150 l/s. Numerous waste piles are located at various locations along the side slopes of the lake. AMD from these have not been determined.

Both the old smelter and the new concentrator are located on the north bank of Lanvatn Lake. Wastes and tailings from these have been discharged into the lake forming large deltas (approx. 10 ha).

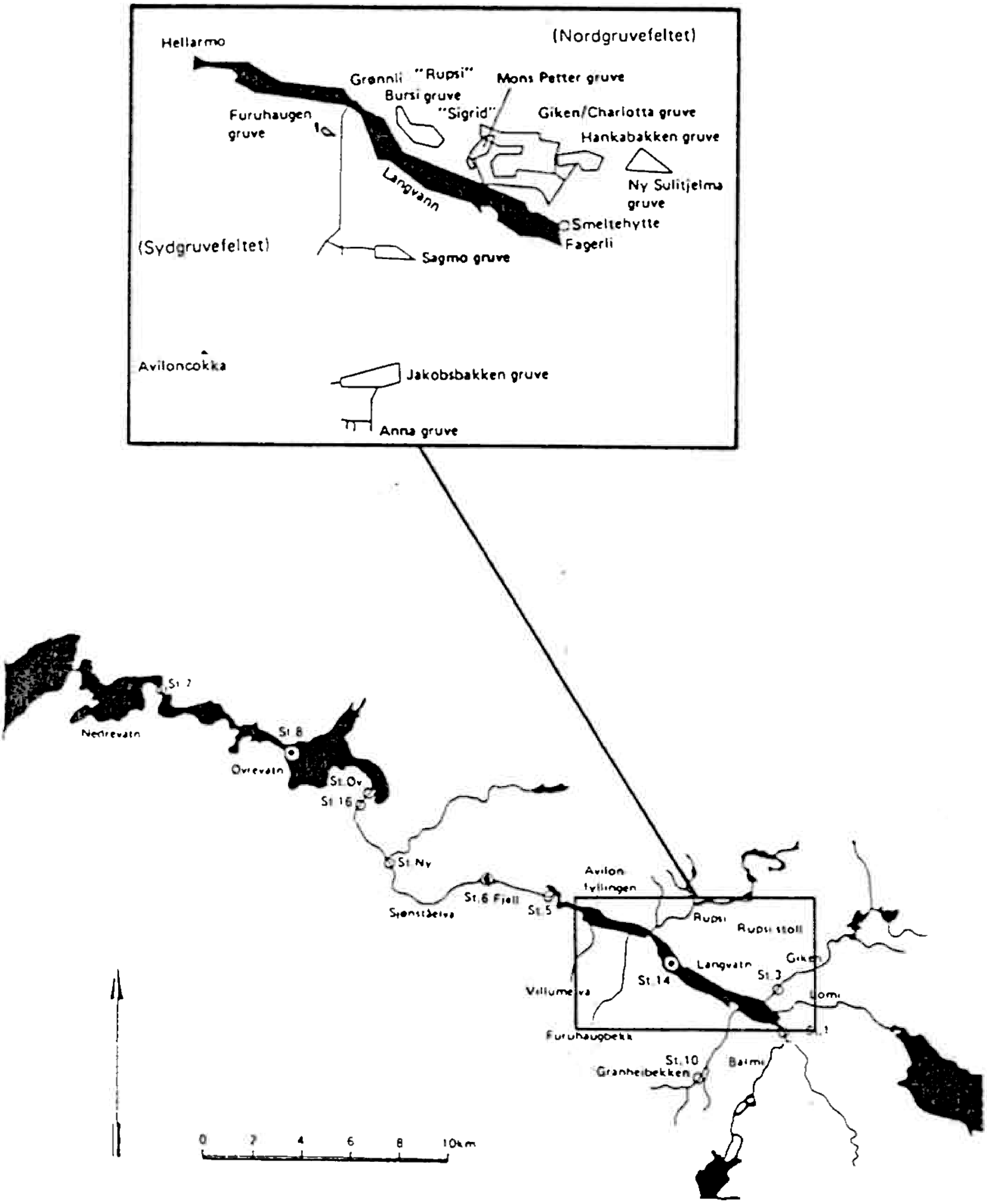


Fig. 15 Location of Mines at Sulitjelma

Current tailings discharge is from a open ended pipe, 10 m below the surface, well out in the lake. With such a discharge a high level of turbidity and suspended solids may be anticipated. Suspension may be maintained by the substantial hydro power surge inflows that occur into the lake. To reduce these suspensions, consideration may be given to de-airing the tailings and effecting a deeper, near lake bottom discharge. The combined impact of all AMD to Langvatn lake water is demonstrated by the concentration profiles for copper as shown in Figure 16 (after NIVA, 1987c). There are no fish in the lake. Langvatn discharges to Lake Orevatn which is also substantially impacted. Copper concentration profiles for Orevatn are shown in Figure 17. A significant and possibly increasing impact is apparent.

5.7.2 Alternative Abatement Options

i) Jakobsbakken

If the AMD from the scattered wastes are significant, they may be excavated and consolidated with the main wastes. Because of the leaching of the base of the main waste pile by AMD from the mine there is little benefit in covering the pile in-situ. Consideration should be given to removing it to a new land location and covering it with a synthetic membrane liner, or placing it under water cover in the small lake near to the mine. These measures should effectively abate AMD from the wastes.

AMD from the mine appears to be the major source of AMD. It may be partially abated by plugging the current outlets, though only limited additional mine flooding is anticipated, if any. The potential for diverting inflows into the mine through surface cracks, using diversion ditches or crack sealing, should be investigated. Abatement of the remaining AMD may be achieved by making the maximum use of natural biological treatment in natural and enhanced wetlands along the flow path to Langvatn lake.

ii) Mine wastes and tailings from mines north of Langratn

A detailed survey of the location of all wastes and acid seepage is required to identify the sources. Following this definition alternative remedial measures can be proposed. The following are a list of possible actions that may be considered for deposits that were identified during the site visit.

a) Slag and tailings delta: The materials located above the water table in these deltas could be dozed or dredged into the lake so that a water cover can be developed. Mixing with lime during removal may be appropriate to neutralize stored acidity.

b) Acid generating wastes located near to the lake could be removed to the lake in a similar manner.

c) Acid generating wastes on land dumps at higher elevations can be covered with till or synthetic membrane covers.

iii) AMD from mine workings.

Abatement of this AMD is a difficult problem. The secure technical solution is collection and treatment, for which the on going costs would be high. Other solutions would involve:

a) sealing the mines to water and air entry. The potential for this appears low.

b) piping the water to the sea. This would probably involve a pressure pipeline along the road route. Advantage could be taken of the natural head available at the AMD discharge points to avoid the need for pumping.

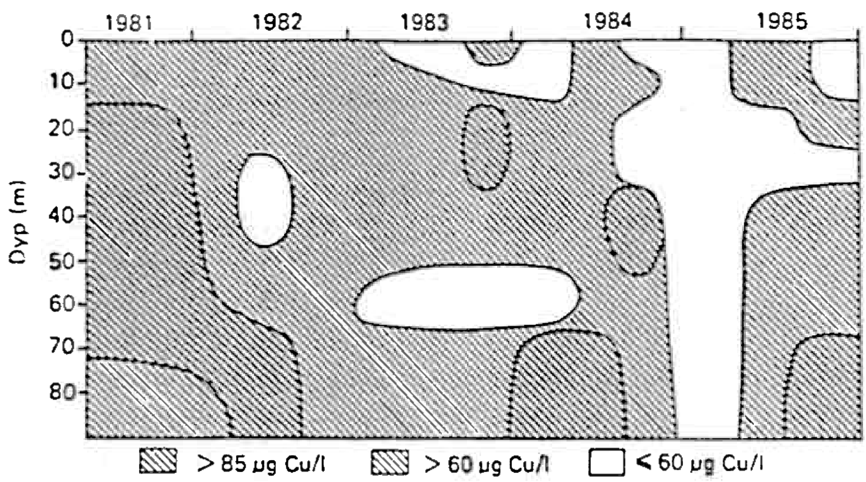


Fig. 16 Copper Concentration Profiles in Langratn Lake

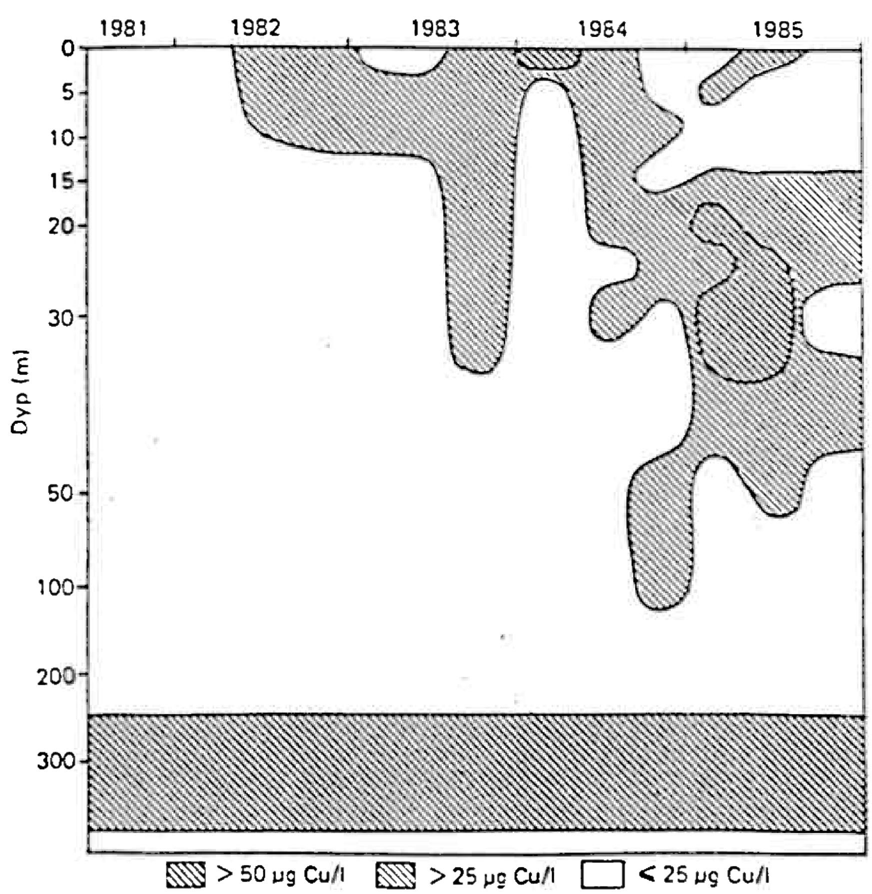


Fig. 17 Copper Concentration Profiles in Ovrevatn Lake

5.7.3 Cost/Benefit Estimates

A realistic assessment of cost/benefit for the alternative abatement measures at this site is not practical at this stage.

6.0 CONCLUSIONS

The impact of AMD from operating and abandoned mine sites in Norway is substantial. Lakes such as Langvatn and Orevatn near Sulitjelma, Tunnsjoen and Store Skorovatn at Skorovas, Orvsjoen at Sekstus Gruve and Djupsjoen near Roros are all impacted to the extent that their aquatic quality and fisheries are affected. Rivers Glama and Guala near Roros are similarly impacted and the Namsen and Orkla rivers could be further impacted if adequate abatement measures are not implemented or maintained.

By implementing abatement measures it is possible to reduce the impacts and achieve beneficial gains in environmental quality. The responsibility of implementing abatement measures at old abandoned mines accrues to the State. The extent to which public funds should be expended on abatement is dependant on the perceived benefits, quantifiable and indeterminate. The quantifiable impacts (such as loss of fisheries productivity, effects on sports fishing and tourism or recreational value), can be translated into monetary terms, but this has not been done for any of the mine sites in Norway. By comparing the costs of alternative abatement options with the fiscal benefits it is possible to arrive at one measure of the appropriate level of abatement.

At operating mines there is the issue of the consequences of additional impacts resulting from proposed close-out measures. The damage resulting from such additional impacts can be translated into fiscal terms and this used to assess the appropriate level of abatement.

The question of who should bear the costs of abatement is a matter of opinion. The author's perception is that the minerals recovered in the generation of the wastes were used for the benefit of the citizens of the State, providing employment and taxes. The liability and benefits, as they were understood, were accepted by all beneficiaries at the time of project initiation. It is appropriate that they all share in the incremental costs of AMD abatement required as a result of recognition of an increased impact and the desire to abate that impact.

Selection and development of the most appropriate abatement options at the various Norwegian mines will require a program of site, sources and impact characterization; alternatives design and evaluation; implementation and monitoring. It is recommended that those sites where the abatement options will potentially have the best cost/benefit be considered first. Kjoli appears to be the site most worthy of early investigation and abatement.

At the same time as working towards abatement at the abandoned mine sites, attention must be focused on operating mines and their proposed close-out measures. It is recommended that each operating mine be required to prepare an abandonment plan which includes the site, waste and environment characterization necessary to allow the impacts and costs of the proposed close-out measures to be evaluated. It may be necessary to provide technical and/or financial assistance to the mine to be able to do so. The early development of the close-out plan allows some of the abatement and close-out measures to be performed as part of the ongoing operations of the mine, often with considerable savings. An early development of the close-out plan also allows provision to be made for the fiscal requirements of the plan.

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APPENDIX A: Papers of Direct Relevance

(i) "Alternative Acid Mine Drainage Abatement Measures" by A. MacG. Robertson

(ii) "Overview of AMD Prediction Methodologies" by K.D. Ferguson and P. Erickson

ALTERNATIVE ACID MINE DRAINAGE ABATEMENT MEASURES

Dr. A. MacG. Robertson, President, Steffen Robertson and Kirsten (B.C.) Inc., Vancouver, B.C., Canada

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ALTERNATIVE ACID MINE DRAINAGE ABATEMENT MEASURES

Dr. A. MacG. Robertson, President, Steffen Robertson and Kirsten (B.C.) Inc., Vancouver, B.C., Canada

INTRODUCTION

Acid mine drainage (AMD) is recognized as potentially the single largest cause of detrimental environmental impact resulting from the mining of ores. There are currently over 2 billion tons of potentially acid generating tailings in Canada (John, 1987). If the cost of abatement measures were $1.00 per ton, an approximate figure which appears to apply in many cases, this represents an industry liability of at least 2 billion dollars. This figure does not provide for any consequential costs for environmental damage nor does it allow for abatement of AMD from mine wastes other than tailings.

AMD from abandoned and operating mines in the USA has resulted in a large number of these properties coming under mandatory Federal clean-up and abatement as a consequence of the US Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA) and Superfund Amendments and Re-Authorization Act of 1986 (SARA). Under these acts Federal Authorities are empowered and required to investigate sources of environmental impact, implement remedial actions, compensate those who have suffered significant damage and recover the costs of these actions from those responsible for causing the environmental damage. Natural resource damage, resulting from AMD, and related remedial costs suits under this legislation currently totals many billions of dollars.

The US experience serves to demonstrate the level of public concern (as reflected in US legislation) and the responsibility and liability that the US public has attributed to the US mining industry. Canadian industry should anticipate similar responsibility and liability.

Both mining industry concerns and public concerns have developed as a consequence of increasing environmental awareness and relatively recent realization of the long-term detrimental impact of AMD. The industry finds itself with a liability which it neither anticipated nor fully understands. Industry has not had sufficient time, or financial resources, to develop a complete understanding of the processes of AMD and has, as yet, not been able to develop a secure, cost effective control technology. It is necessary that the mining industry, together with the public that benefits from it, and the regulatory authorities responsible for its promotion and control, work together to find and implement control technology that will provide adequate abatement of existing AMD and permit the continued recovery of natural resources from mineral deposits with potential for AMD.

This paper presents a brief review of available and alternative AMD abatement technology. It considers primarily the technology relevant to mine waste and tailings, though some of the techniques may also be applicable to exposed pit walls and drainage from underground workings.

Technology for AMD abatement is most extensively developed for the coal fields of the eastern United States. This technology is valuable, particularly as regards AMD collection and treatment. There are a number of fundamental differences between typical AMD from hard rock waste dumps and local overburden spoil piles. For example, coal overburden in the eastern US usually contains a high percentage of slaking mudstones or shales. Large changes in permeability of these materials, with time, results in substantial natural reductions in infiltration and oxygen penetration. Hard rock waste dumps remain highly permeable to both water and air for a long time and similar reductions in AMD do not occur as rapidly. There are also differences in the nature of the soluble metals. Transfer of this technology must be done with due caution.

A distinction must be drawn between existing deposits and new deposits. The technical options available for AMD abatement are considerably greater at new deposits than at existing deposits. At existing deposits, abatement measures are limited by the site specific and deposit conditions. Further, the operator of an existing deposit is limited by the economic constraints which have developed as a result of planning that may not have included provision for AMD abatement. The necessity now to include provision for abatement may demonstrate that the cost of such abatement cannot be supported by the remaining resource value. This is amply demonstrated by the lack of funds available to abate AMD at abandoned mine properties. Under these circumstances, the always important factor of 'cost of abatement' becomes a crucial factor, which may eliminate some of the otherwise more appropriate alternatives.

KINETICS OF ACID GENERATION

An understanding of oxidation and acid generation is essential in the development of abatement measures.

The process of acid generation in mine wastes has been described by numerous workers including Knapp, 1987; Paine, 1987; Errington and Ferguson, 1987. It is a time dependent process controlled primarily by:

- the presence and nature of reactive sulphides.
- availability of water.
- availability of oxygen (convection and diffusion)
- bacterial action
- temperature
- pH or presence of base alkaline reactants

If a base source is present (and available) in the waste in sufficient quantities, the acid products are immediately neutralized and additional acid generation is inhibited. This effectively abates AMD. Most mine wastes contain at least some base potential. Unfortunately this base is sometimes not all available for acid neutralization, or it becomes available at a rate which is insufficient to neutralize the acid at the rate at which it is generated. Figure 1 illustrates, diagrammatically, the availability with time of base and acid in two hypothetical materials. Both materials have a net acid generating potential. Material type A does not produce sufficient acid for all the base to be consumed, and AMD does not occur in the time period of interest. pH controls may prevent material type A from ever producing AMD. Material type B produces sufficient acid that, after a period represented by OF, it becomes acidic, and thereafter acid generation rates increase as a result of bacterial action and other pH dependent reactions and AMD becomes severe. Thus AMD is dependent not only on the acid/base potential of the material, but also on the total quantities of acid or base that are available with time.

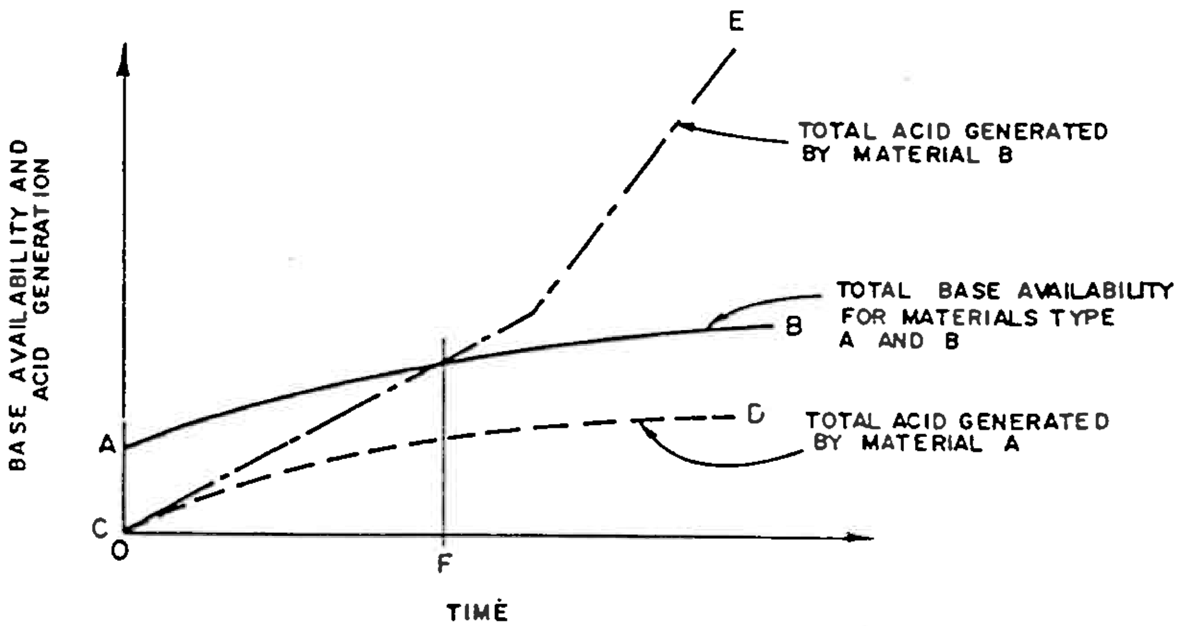

FIGURE 1 : RELATIONSHIP BETWEEN ACID GENERATION AND BASE NEUTRALIZATION POTENTIAL WITH TIME FOR TWO DIFFERENT WASTE TYPES

Acid generation prediction techniques are described by Ferguson and Erickson, 1987. Both static and kinetic tests are used in an attempt to determine the long-term behaviour of the wastes. Concern exists that the conditions imposed during some testing, to accelerate the time dependent reactions, result in unrealistically conservative results.

For coarse waste materials spiked (acidulated) tests may be more appropriate. Waste is not mixed or blended to the same degree as tailings. A single small quantity of highly acid generating waste may be mined and placed, without intermixing, in the waste pile. This small quantity of rock waste may become acid generating, long before the remainder of the waste in the pile, and act as a seed or natural spike for the remainder of the pile. Thus the initiation of AMD in waste rock dumps may be triggered by such natural seed quantities and not by the more typical average conditions.

The extrapolation of laboratory or field acid generation test results to estimates of AMD from a field deposit requires that the field deposit be modelled. Such modelling is extremely complex. The most comprehensive modelling program developed to date (RATAP or Reactive Acid Tailings Assessment) was developed for acid generating tailings and has been calibrated against the Elliot Lake uranium tailings (SENES and Beak), 1986. This model requires calibration for application to other tailings and must be further developed for application to waste rock. This model must be linked to other such modelling programs to evaluate the full pathway for any AMD before it impacts surface waters. The current state of the art is such that the predictive accuracy of long-term modelling of new tailings or mine waste deposits is extremely uncertain. Thus it is also very difficult to evaluate the absolute effects of alternative abatement measures in the long-term. Modelling does permit an effective evaluation of the comparative effects of different abatement measures.

ALTERNATIVE ABATEMENT APPROACHES

Alternative approaches to AMD abatement are reviewed in this section, while the effectiveness of specific abatement measures are evaluated in the section entitled 'Alternative Abatement Measures'.

AMD abatement may be divided into three broad approaches: control of acid generation; control of acid migration; and collection and treatment of AMD.

CONTROL OF ACID GENERATION

Acid generation control implies the prevention of acid formation. This requires one or more of the following:

i) Removal of acid generating minerals. Methods of pyrite removal have been evaluated by Hester and Associates, 1984 and it has been concluded that the economics cannot generally support this alternative.

ii) Rendering acid generating minerals inactive by developing chemical coatings. Hester and Associates, 1984, demonstrate that while these approaches hold promise they do not as yet represent applicable technology.

iii) Exclusion of water. This would require dry placement and the exclusion of any infiltration. Such conditions could only be achieved with a synthetic membrane cover. Clay or other low permeability soil covers have been evaluated by Steffen Robertson and Kirsten (1986a) in the context of covers for uranium tailings. It is concluded that long-term degradational effects would result in sufficient water penetrating the cover to enable acid generation to continue.

iv) Exclusion of oxygen. Entry of oxygen into the reactive waste is controlled by:

a) the convection of air into the waste. Convection plays an important role in coarse waste piles. Daily and seasonal variations in barometric pressure result in the pile 'breathing', almost like a lung. Temperature differentials between the interior of the pile and the ambient air, particularly in instances where the heat of oxidation has elevated temperatures, result in thermal convection.

b) the diffusion of oxygen through the cover or into the water. The diffusivity of typical dry tailings is about 2.0×10^{-2} cm/sec (Halbert et al, 1983). At this value, unacceptably high rates of oxidation and acid generation are being experienced in the reactive tailings at Elliot Lake and many other tailings deposits. Diffusion rates into coarser waste must be considerably greater. Tailings therefore are not considered a suitable candidate material for the elimination of acid generation, by either placing as a cover or by injection into the waste. Studies of diffusion through uranium tailings (Silker and Kalkwarf, 1983) have demonstrated that similar diffusivity values are obtained for typical soils, including clayey soils, if they are dry or only partially saturated. Thus dry soil or clay cover layers are also probably inadequate as oxygen excluders, although they may play another, more effective, role as inhibitors of the water transport medium, as discussed in the next section. The diffusivity of oxygen through water is about 2.0 x 10^{-6} cm/sec (Klohn Leonoff, 1981), i.e., about four orders of magnitude less than that for typical dry tailings.

Water covers have been demonstrated to be effective for tailings, reducing acid generation to very low levels. Figure 2 shows the correlation between the diffusion

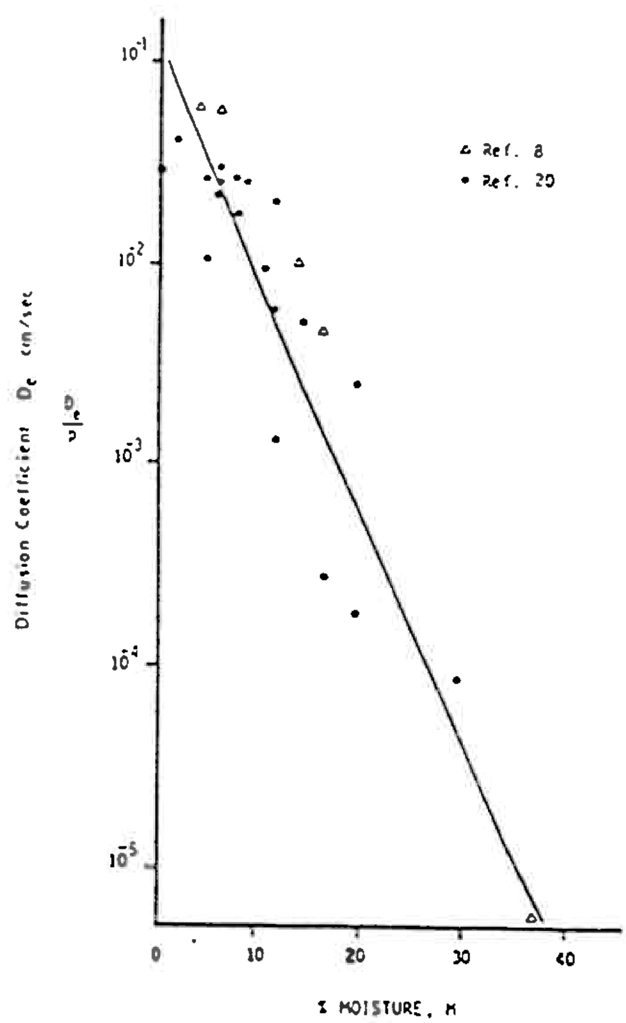

FIGURE 2: MOISTURE DEPENDENCE OF THE DIFFUSION COEFFICIENT (ROGERS AND NIELSON, 1981)

coefficient for a gas and the moisture content of tailings as determined by Rogers and Nielson, 1981. This figure demonstrates that a high degree of saturation is required to reduce diffusion to values approaching that of water cover. If a saturated layer can be developed above the tailings this would provide an effective oxygen barrier. Maintenance of saturation is important, requiring either the establishment of permanent bog conditions, or the design of a complex layered cover to trap precipitation and inhibit evaporation as has been proposed for the Faro Mine tailings in the Yukon (Steffen Robertson and Kirsten, 1986b).

v) Control of bacterial action. Once the pH in a waste pile drops below about 4.5 bacterial action increases the rate of acid generation by five fold or more. This bacterial action cannot be controlled by spraying or mixing the waste with bactericides (Sobek, 1987). Only biological acid generation is controlled in this manner, and the period of effectiveness of the bactericide must be taken into account.

vi) Temperature control. The rate of acid generation depends on temperature as illustrated in Figure 3 (Knapp, 1987). The

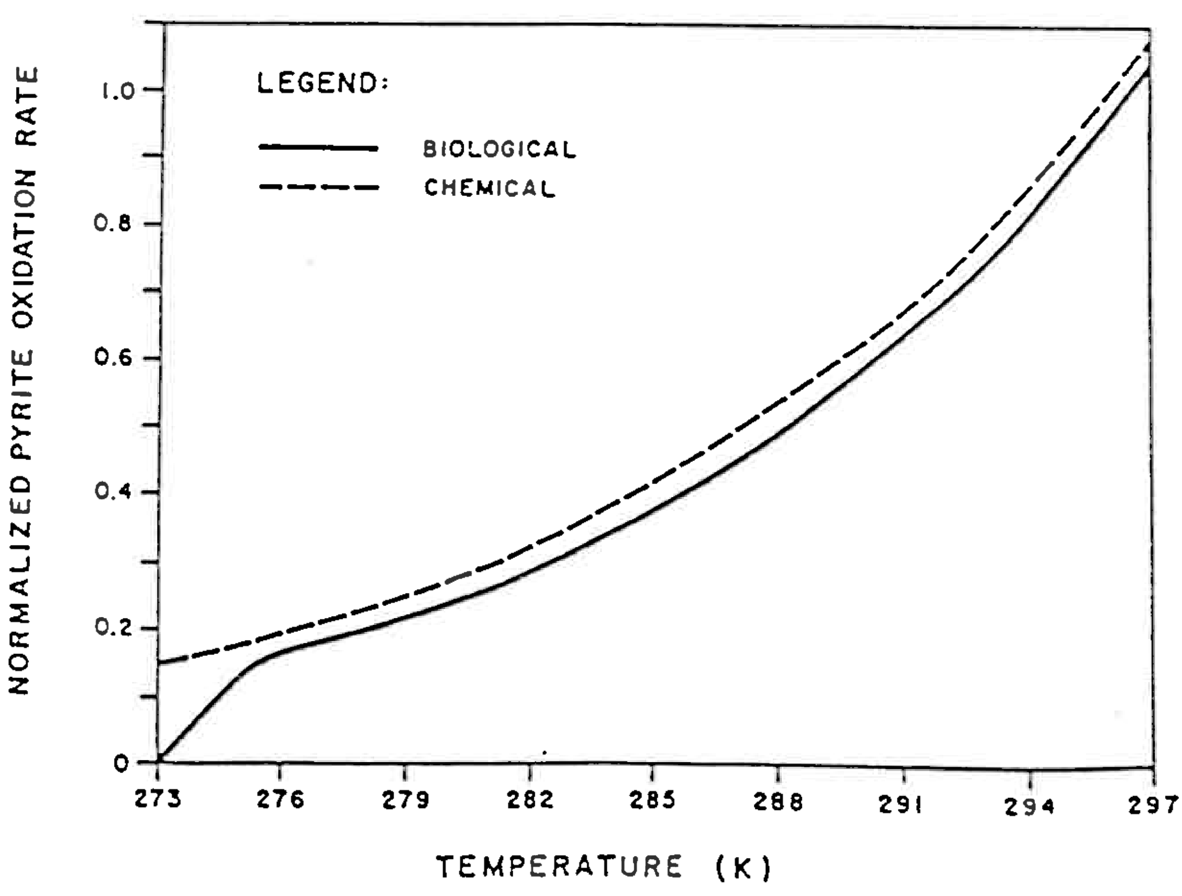

FIGURE 3: EFFECT OF TEMPERATURE ON BIOLOGICAL AND CHEMICAL OXIDATION RATES (AFTER KNAPP, 1987)

effect of temperature is considered to be large in the northern parts of Canada. Diffusion in gases varies as a power of the absolute temperature, somewhat greater than 3/2. Bacteria also are less active at low temperatures and become dormant during the frozen season. The period of elevated summer temperatures may be too short to allow the bacteria to become fully active and propagate. This is demonstrated by low bacterial activity at the Faro Mine in the Yukon, (Steffen Robertson and Kirsten, 1986b). Placement of a cover has the effect of reducing the maximum temperatures of the upper surface of the reactive tailings, with a commensurate decrease in the rate of acid generation. Frozen conditions effectively reduce acid generation. In areas of permafrost, the potential for achieving permanently frozen conditions provides a means of AMD control. In marginal or discontinuous permafrost areas, consideration may be given to the development of natural frost inducing conditions, or the use of artificial frost inducing devices such as Cryopiles or thermal syphons. These procedures are as yet experimental.

vii) pH control. The effects of pH on biological and chemical oxidation rates are illustrated in Figure 4 (Knapp, 1987). Control of pH can be used to reduce both the chemical and biological oxidation rates. Control is achieved by adding limestone or lime (or other alkaline materials) to the waste or spreading it on the surface of the pile. Again, acid generation is not eliminated, and the effective period of the pH control must be considered. The addition of an alkaline material, however, has the advantage that acids are neutralized as they are produced, reducing the potential for low pH water to dissolve and transport metal contaminants from the piles. The neutralized acid products remain in the piles. Many of these products are more readily soluble in acid drainage, and regeneration of AMD would result in an increased environmental impact. Redevelopment of AMD must therefore be prevented.

CONTROL OF ACID MIGRATION

Acid migrates as a flow of low pH water from the waste pile into the environment. Because of differences in permeability, coarse waste dumps and tailings deposits do not behave the same, and abatement measures have different effects.

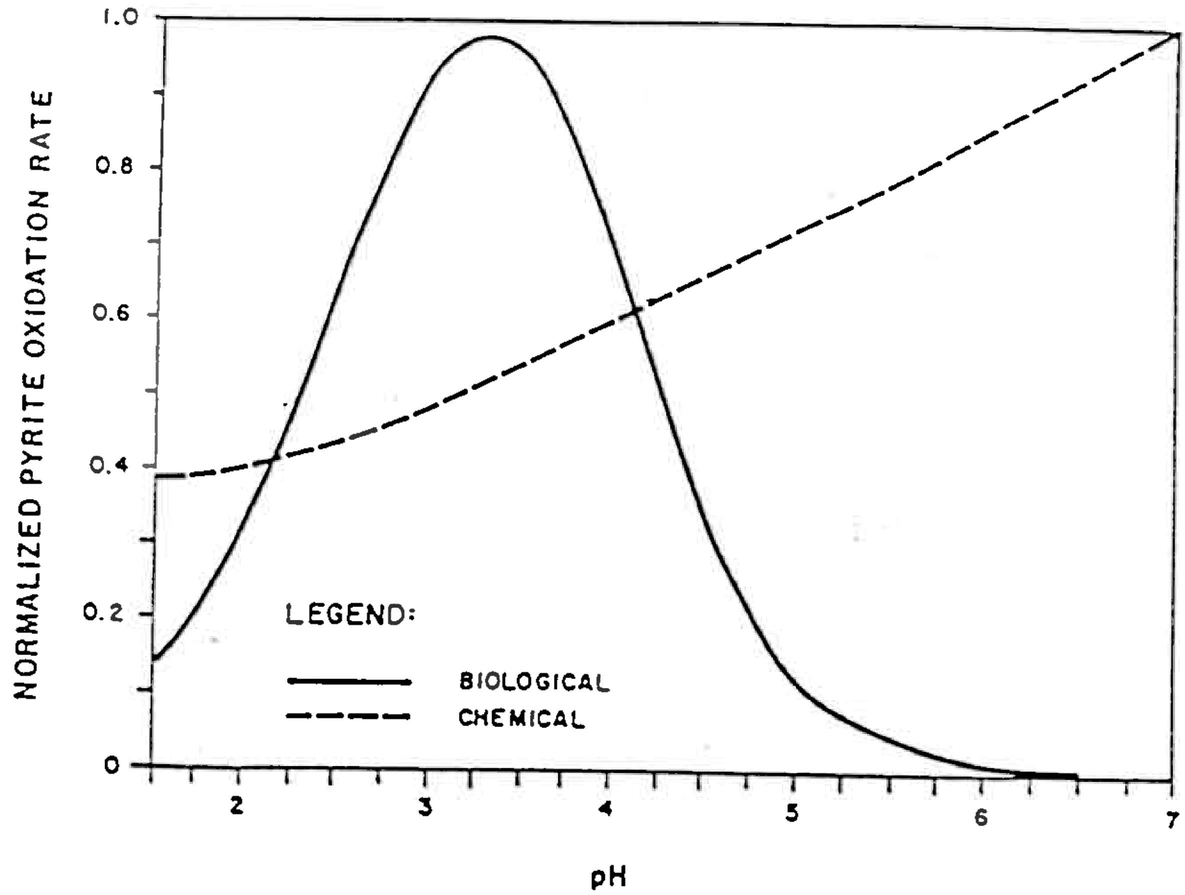

FIGURE 4 : EFFECT OF pH ON BIOLOGICAL AND CHEMICAL OXIDATION RATES (AFTER KNAPP, 1987)

Coarse Waste Dumps

Coarse waste dumps have a very high permeability to both air and water. Oxygen and water are available almost throughout the dump, allowing acid generation and reaction products to be distributed anywhere within the mass. Precipitation and surface flows onto the dump infiltrate rapidly. Runoff is often low. Flow through the dump tends to follow preferred channels along which the acid products are regularly flushed out. There are zones in the dump that are seldom flushed by running water but receive moisture as a result of water vapour migration and condensation. These condensates may become highly acidic and trickle down to join the less acidic flows. The combined flow exits the toe of the dump, or enters the foundation soils to form a subsurface contaminant plume. Flow through the dump is rapid and rainfall often produce increased flow from the dump toe within minutes or hours of the start of the rain. The dump acts as a store of acid products, which are partially flushed out from time to time. Concentrations of contaminants in the toe seepage may reflect the flushing history. During periods of intense flushing, the concentration of contaminants in the AMD may reduce. Conversely, after a period of low infiltration and flushing the first significant rainfall may result in both high concentrations and large AMD loadings. Figure 5 illustrates diagrammatically the flow rate, concentration and loading variations that have been experienced at a typical small sized hard rock waste pile in British Columbia at a location where high rates of infiltration are experienced in June and October (Steffen Robertson and Kirsten, 1987).

Sulphide oxidation commences as soon as the first coarse waste is placed in the dump. Thus AMD from the toe of the dump can sometimes be detected within a few months of waste placement, depending only on the time required for part of the dump to become acid generating (refer Figure 1).

Tailings Impoundments

In contrast to dumps, infiltration and flow of air and water through tailings impoundments is restricted by the relatively low permeability of the tailings. This flow restriction results in the development of zones of oxidation and contaminant migration within the tailings, as illustrated in Figure 6a.

Water infiltrating through the tailings surface enters a zone of partial saturation, Z1. After passing through this zone it joins the water below the water table in the tailings. Infiltration water accumulates in this layer, Z2, and displaces the original process interstitial water downwards into underlying soils (in this case alluvium) where it mixes with the groundwater. Oxygen entering the tailings is consumed within the zone of oxidation, Z3, which limits the depth in which acid generation occurs to this zone. Water infiltrating through the tailings surface becomes acidic in this zone, Z3, and then flows downwards into a zone, Z4, in which the acidic waters react with the alkalies in the tailings and are neutralized. Depending on the rate of reactions and the elapsed time, the base of Z4 could be above or below the base of Z2. As the sulphides are consumed in the oxidation zone the depth to which oxygen can penetrate increases, and Z3 increases. As the alkalies are consumed in the tailings, the depth of zone Z4 also increases and the acid contaminated seepage and other dissolved products progress deeper into the tailings. After a period of time, zone Z4 penetrates the foundation soils and the groundwater becomes contaminated.

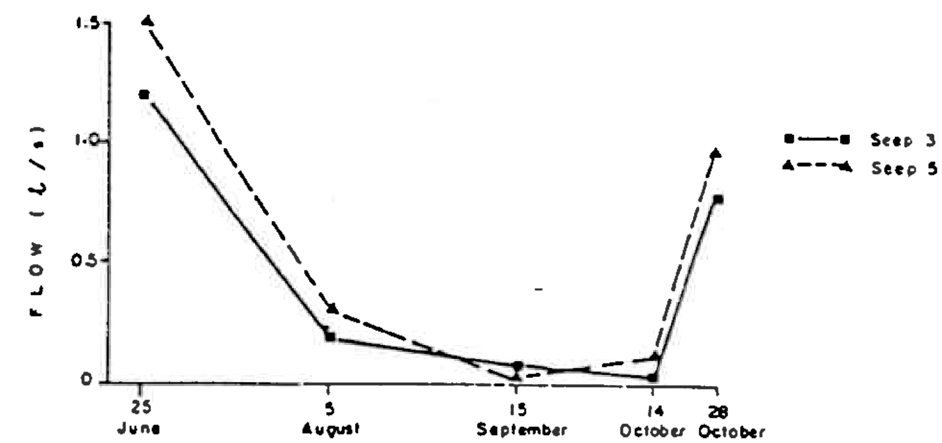

FIGURE 5(a) TEMPORAL VARIATION IN FLOWS AT SEEPS 3 & 5

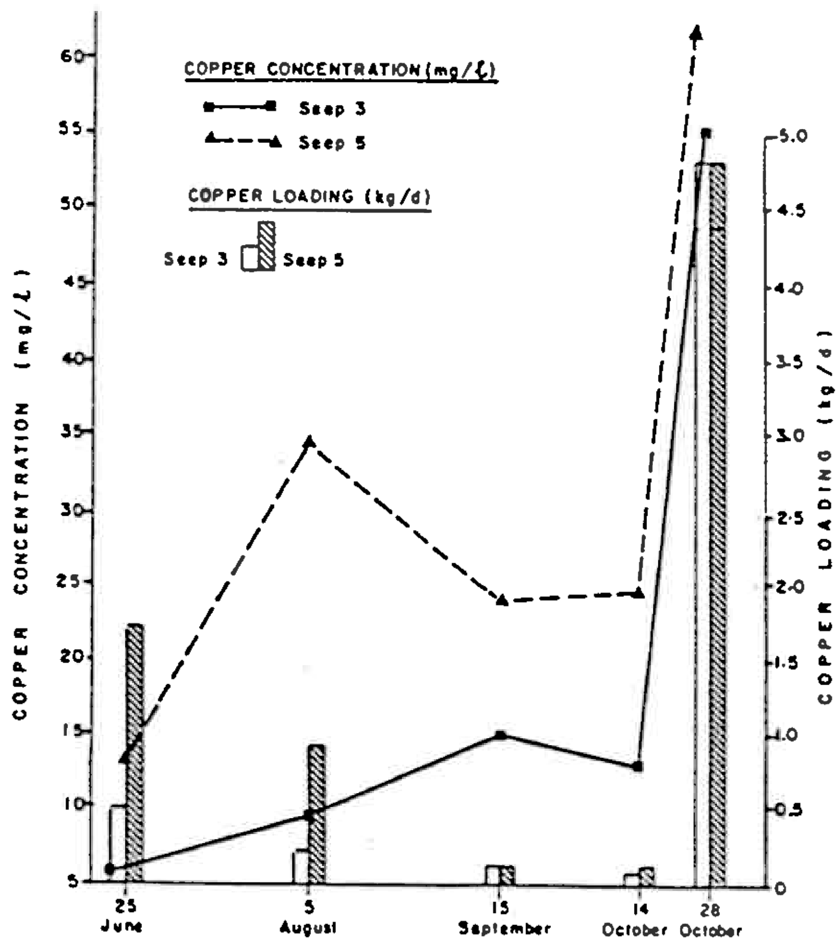

FIGURE 5(b) TEMPORAL VARIATION IN COPPER LEVELS AT SEEPS 3 & 5

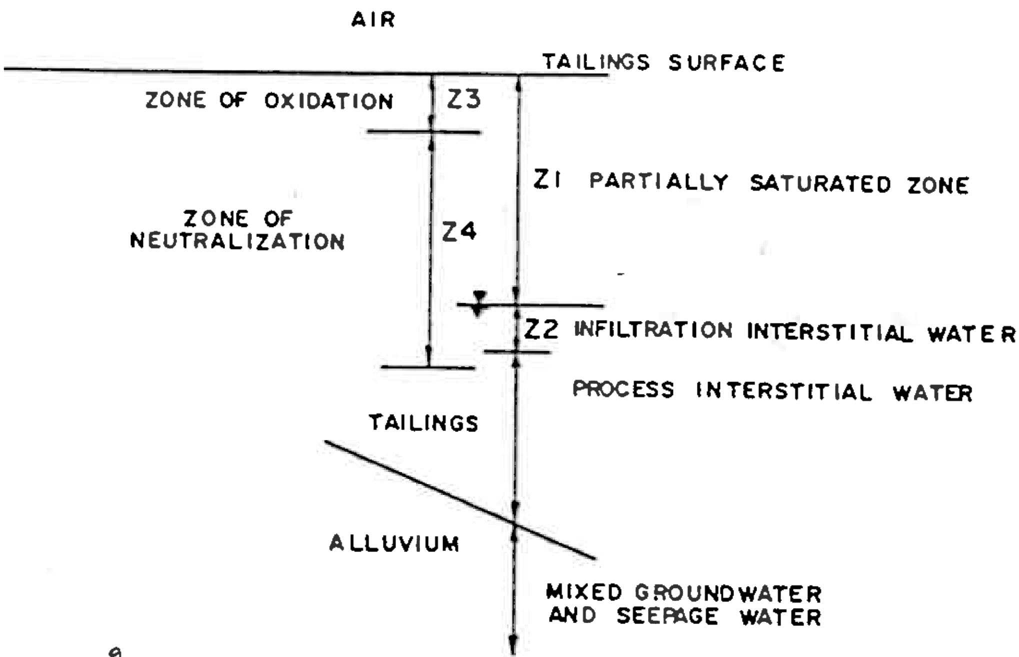

FIGURE ~~6~~ 2(a) WATER QUALITY MODEL FOR TAILINGS COLUMN

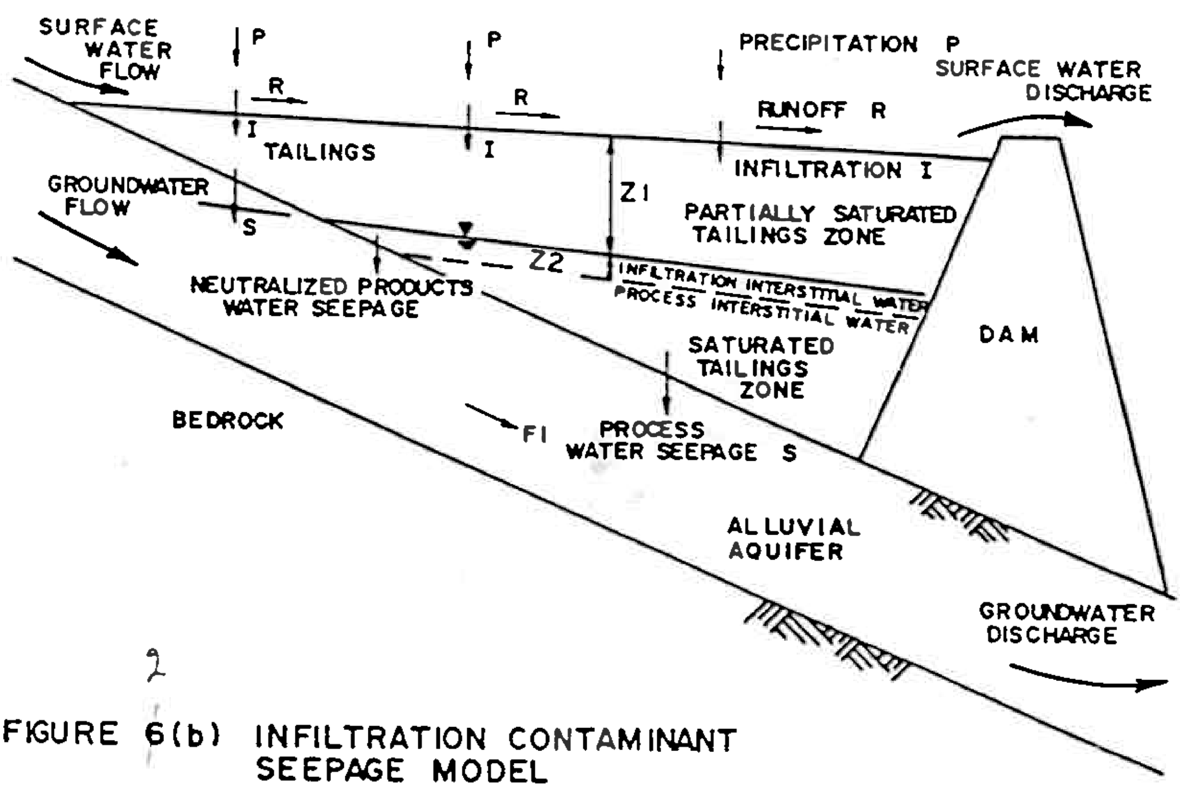

FIGURE ~~6~~ 2(b) INFILTRATION CONTAMINANT SEEPAGE MODEL tailings is consumed within the zone of oxidation, Z3, which limits the depth in which acid generation occurs to this zone. Water infiltrating through the tailings surface becomes acidic in this zone, Z3, and then flows downwards into a zone, Z4, in which the acidic waters react with the alkalies in the tailings and are neutralized. Depending on the rate of reactions and the elapsed time, the base of Z4 could be above or below the base of Z2. As the sulphides are consumed in the oxidation zone the depth to which oxygen can penetrate increases, and Z3 increases. As the alkalies are consumed in the tailings, the depth of zone Z4 also increases and the acid contaminated seepage and other dissolved products progress deeper into the tailings. After a period of time, zone Z4 penetrates the foundation soils and the groundwater becomes contaminated.

Groundwater contamination is expected to occur first near the edges of the tailings impoundment where the depth of tailings is least and Z4 reaches the foundation soils first (refer Figure 6b). This contaminant load increases as Z4 increases and a larger portion of the impoundment contributes to groundwater contamination. The initial acidic flows reaching the foundation soils are neutralized by the base capacity of the foundation soils, which can be very large.

Acid generation in the tailings generally only commences after the last of the tailings have been discharged to the surface of the impoundment. Prior to this, acid generation is prevented by the neutralization which occurs each time a new layer of tailings is deposited (effectively alkaline control). Once acid generation starts it may be years (possibly tens of years) before the acid product front passes through the tailings and foundation soils to reach a surface discharge location. Surface flows across the tailings, on the other hand, may produce AMD shortly after tailings discharge stops, depending again on the time period for the tailings on the surface of the impoundment to become acid generating.

Initially it may appear that there are two approaches to the control of acid migration: (i) control of seepage <u>entry to the waste</u> using diversion ditches, and low permeability covers and (ii) control of seepage <u>away from the waste</u> using liners or cutoff walls. Interception systems consisting of wells or trenches are not migration barriers but collection systems and are reviewed in the next section. In practice approach (ii) cannot function in the long-term since infiltrating water which enters the waste and becomes acidic, and which can not escape via seepage, must accumulate until the available storage is filled and then discharge over the lip of the liner or cutoff wall. Thus the only effective long-term abatement is infiltration control and the prevention of groundwater entry to the waste.

Groundwater leaching can be very difficult to prevent if the site has not been selected to avoid it. Cutoff walls may be considered. Surface infiltration can be reduced considerably by diversion of surface flows, surface contouring to promote runoff, and placing low permeability covers. Almost complete elimination of infiltration can be achieved by a synthetic membrane cover, though provision would have to be made for its long-term replacement. Low infiltration rates (sufficient to achieve large AMD reductions) can be achieved by placing clay or other low permeability covers. Such covers may be subject to degradation with time, due to erosion, frost action and roots.

ACID DRAINAGE COLLECTION AND TREATMENT

The collection and chemical treatment of AMD has been the most widely applied abatement measure to date. A large number of papers are available on the topic (Vachon, et al, 1987). Collection of surface flows is usually fairly readily achieved. The collection of subsurface flows requires the installation of collection trenches, or wells, or cutoff walls to force the groundwater flow to the surface, where it can be collected.

Chemical treatment involves the addition of alkaline materials (usually lime or quick lime), and the settling of the resulting sludges in a settling pond. Major concerns relating to this abatement measure are the need for long-term treatment, the quantities of sludges produced and the requirement for the long-term stabilization of the sludges.

AMD can generally be expected to continue for many decades if not centuries. Treatment must be maintained therefore during these long periods. If there is a breakdown of the treatment plant, the AMD is essentially unabated for the period of the breakdown. During this period substantial damage may result to aquatic resources in receiving waters. Thus it is essential that such a treatment plant be built with adequate reliability and redundancy to minimize the risk of breakdown. This contrasts with most other abatement measures, which fail progressively allowing corrective measures to be implemented before the impact becomes excessive.

Each tonne of pyrite in the waste has the capacity to produce approximately 3 tonnes of acid, which, when neutralized, produces about 7 tonnes of dry sludge. These sludges are extraordinarily difficult to densify in a settling pond. Steps taken to improve the density (thickening, underdrainage, evaporative drying, etc.) typically result in settled sludge densities of 10 to 15 percent solids by weight. Thus one tonne of sulphides may produce 70 tonnes of wet sludge.

1000 - 500 - 1500

Thus a waste pile containing 2% sulphides has the potential of producing a volume of sludge that exceeds the volume of the original waste.

Often these sludges are readily leachable by moderately low pH waters and are a potential long-term source of contaminants. They in turn must be carefully stored and protected. In the long-term, the containment of the sludge may be a considerably more difficult task than the alternative methods of AMD abatement.

Passive methods of treatment have and are being investigated. These include:

i) Passing AMD through limestone trenches or channels. The large volumes of sludge produced result in rapid coating of the alkaline materials and filling of the voids between them. Thus the trenches have a short effective life and cannot be considered in the long-term.

ii) The effectiveness of wetlands for the biological treatment of AMD has been well demonstrated (Kalin and Van Everdingen, 1987). A number of difficulties and uncertainties, however, still limit the general application of this abatement measure.

ALTERNATIVE ABATEMENT MEASURES

This section provides a brief review of the effectiveness of some abatement measures. Many of the measures are not by themselves sufficient to control AMD from any given site. Used in combination with other measures they may yield the most cost effective means to achieve the desired short and long-term abatement.

DIVERSION OF SURFACE WATER

Diversion of surface waters is almost always an inexpensive effective abatement measure. Diversion also reduces the potential for erosion of the other measures. Unfortunately, diversion works are themselves highly vulnerable to long-term disruptive forces, such as sedimentation, debris and ice blockage, erosion, etc., and maintenance is usually necessary at fairly frequent intervals.

Diversion of groundwater, by installing cutoff walls, etc., may also be considered where this can prevent leaching of the wastes.

CONDITIONING OF TAILINGS

Some improvements in AMD can be achieved by placing tailings in a condition that is more favourable for AMD abatement. Placement of tailings or wastes entirely under water (subaqueous deposition, i.e., providing a water cover) effectively prevents oxidation and therefore AMD.

Placing tailings in a systematic managed manner, to achieve a uniform deposit with maximum density and minimum segregation results in the minimum permeability to both air and water. Layered tailings placement, with minimized pool areas and maximized discharge densities is the most suitable placement method. This technique is often referred to as 'sub-aerial' (Knight and Haile, 1983) or 'semi-dry' placement. Reduction in acid generation due to oxygen exclusion is negligible. Reduction in infiltration (due to reduced surface permeability) is significant but still comparatively small. Where underdrainage is maintained this may increase the rate of both oxygen entry and AMD. Thus the direct beneficial effect of layered tailings, on AMD abatement is small, and, in some instances, may be detrimental. Of greater importance is the improved consolidation characteristics and surface trafficability which permits easier cover placement.

The relatively poor control of AMD provided by layered or sub-aerial deposition is graphically demonstrated by the experience with South African gold tailings, where layered tailings deposition is practiced extensively. Oxidation and acid generation has penetrated many meters, in some cases tens of meters, into these tailings.

COVERS

Covers offer one of the best means for long-term abatement of acid generation.

(i) Vegetation

It is desirable to establish vegetation on the waste pile in order to return the surface to a state similar to the surrounding environment. Vegetation also serves the very useful purpose of reducing erosion and therefore the frequency of maintenance work (Feldhuizen, Sewek and Blowes, 1987). There is also some evidence that vegetation may produce a marginal reduction in acid generation in the long-term. Root penetrations will result in permeability increases in any clay or low permeability soil layers installed as cover seals. Vegetation has a significant effect on the rate of infiltration through the cover; this may be either an increase or decrease depending on the precipitation pattern and cover properties.

(ii) Soil, till or clay

Soil covers act:

- to shed surface flows
- as a poor inhibitor of oxygen entry
- as a moderate inhibitor of infiltration
- as a temperature insulator
- to provide a medium for vegetative growth
- as a sacrificial erosion layer

Infiltration of air and water through the cover depends on the permeability of the material used as well as the cracks and holes that may develop over time. Erosion, dessication cracking, frost action, settlement cracking, piping into coarse waste, root penetration and burrowing by biota are all severe long-term disruptive forces, which tend to increase the rate of infiltration and oxygen entry. A recent study of the long-term integrity of such covers (Steffen Robertson and Kirsten, 1986a) has demonstrated that they are vulnerable to these disruptive forces. Their primary effect is the reduction of infiltration and under marginal AMD circumstances this may be sufficient to achieve the desired amount of abatement. Modelling programs are available for the determination of infiltration rates through simple and composite covers. The HELP program (Hydrologic Evaluation of Landfill Performance, Schroeder et al, 1984) is an example. Unfortunately these programs are as yet crude compared with actual field conditions, and they are useful only for comparison of the effectiveness of alternative covers and not for determination of absolute infiltration rates. In a recent study, infiltration estimates were made for a large variety of simple and composite covers (Steffen Robertson and Kirsten, 1987). These indicate that in a high precipitation environment, such as the British Columbia coast, infiltration through a clay cap would still be a significant percentage of the annual precipitation.

(iii) Synthetic membrane liners

Because of their vulnerability to puncture, membrane liners must be installed with adequate bedding preparation and surface protective covers. They are of low permeability and offer the potential of acting both as oxygen and infiltration barriers. Thick (2 mm) high density polyethelene (HDPE) membranes are less susceptible to the disruptive forces effecting soil liners, except for the likelihood of tearing under differential settlement and long-term weathering. To allow for long-term degradation it will probably be necessary to provide for liner replacement in 50 to 100 years.

(iv) Water

Water cover is currently the most secure oxygen inhibiting technique. It is therefore the most secure AMD abatement measure. There are, however, a number of limitations and disadvantages associated with a water cover. These include:

- It may not be possible to achieve water covers on some of the existing deposits.

- Water covers generally imply water retention dams. Such structures have the potential for catastrophic failure and require more secure construction and maintenance.

- Water covers provide driving force resulting in increased flushing of the wastes. If there are other soluble deliterious products in the wastes this may result in increased contaminant loadings to the environment.

- For existing piles, removal to an underwater disposal site may not be feasible because of the loading of oxidation products that have accumulated since the waste was placed.

- Reliable water sources must be available to ensure a continuous cover of sufficient depth to avoid erosion due to wave action or water flow.

- Placement of the wastes in natural water bodies may have other environmental impacts.

Despite these limitations, disposal of acid generating wastes in lakes appears to offer a large number of advantages. The lakes are usually stable basins with an assured water supply, and adequate capacity and depth to insure substantial depths of water coverage. Costs are usually low.

(v) Saturated soil or bog

The effectiveness of a saturated soil layer, for the exclusion of oxygen has been demonstrated.

Bog conditions can be achieved by a combination of a shallow soil cover with a shallow water cover provided for by a water retaining structure. Under these circumstances the tailings will be effectively under water. The soil helps to prevent total loss of coverage when the water depth reduces during dry periods, and it also prevents convective currents and wave action. Vegetative accumulation is also believed to have a marginal but beneficial effect on AMD abatement.

It is also possible to develop a shallow saturated layer of soil as a perched water table on the tailings as illustrated in Figure 7. In this cover a layer of tailings slimes are used to seal the upper surface of the tailings. A layer of till serves as a reservoir to hold water above the slimes. This is covered by a layer of coarse waste rock (non-acid generating) which serves to induce infiltration and retard evaporation. The need to construct a large water retaining embankment is eliminated. The long-term stability and effectiveness of such a cover remains to be evaluated.

BASE ADDITION

(i) Mixing with waste

If low cost sources of base materials, such as limestone, are available the option of mixing them with the waste to render it non-acid generating should be considered.

(ii) Surface application

The potential for acid generation control by surface applications of alkaline materials is less attractive than mixing them with the waste. Limestone has a low solubility in near neutral water, and the resulting alkaline charge is therefore small and insufficient to control AMD. Surface inflows tend to be concentrated at isolated locations such as depressions, cracks , permeable zones, etc. At these locations the available alkaline materials are quickly exhausted. The method has been found to be ineffective.

ALKALINE TRENCHES

(i) Upstream of waste

The alkaline charge can be increased by concentrating surface water flows and directing them through trenches filled with large quantities of limestone and other alkaline materials before deliberately introducing the flows into the waste piles

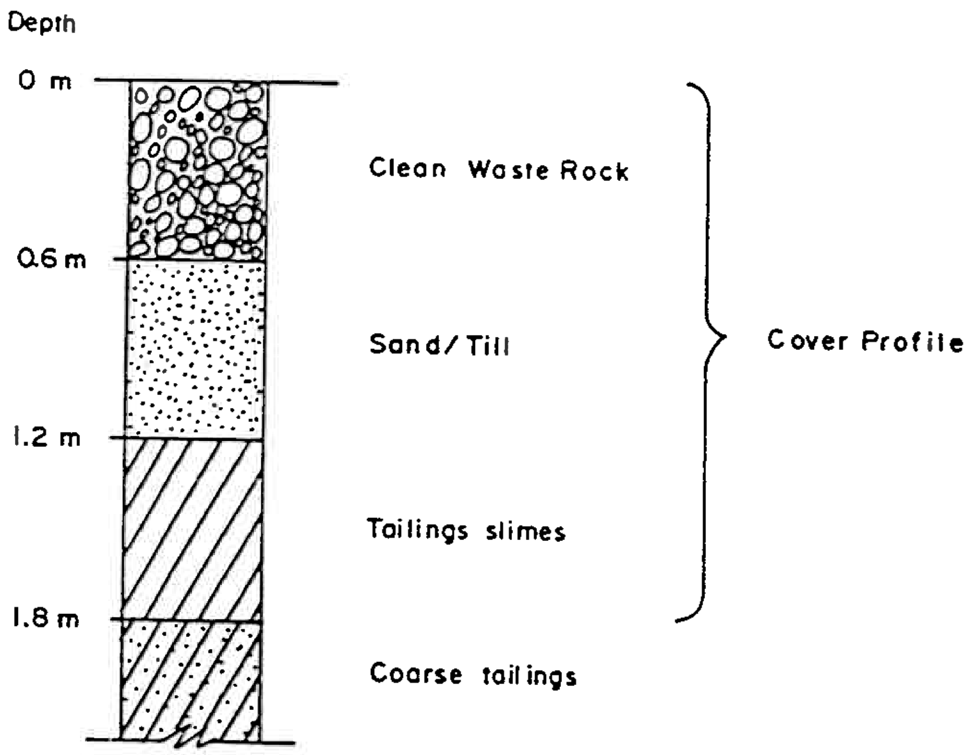

FIGURE 7 COVER PROFILE TO DEVELOP SATURATED ZONE

(Caruccio and Geidel, 1987). While this method is an improvement on surface applications, and may have some application to marginal acid producing coarse rock waste, its long-term effectiveness and application to tailings is questionable, because of its rapid consumation and blinding.

(ii) Downstream of waste

Alkaline materials placed in trenches constructed downstream of the waste pile are quickly coated, blinded and clogged and have been found to be ineffective in the medium or long-term.

BACTERICIDE

(i) Mixing with waste

Bactericides serve only to control acid generation that occurs as a result of bacterial activity. The reduction in AMD is unlikely to reduce contaminant concentrations to levels required for abatement. The current state of the art is such that bactericides mixed with wastes during placement may be effective for up to 5 years (Sobek, 1987). Because of their partial and short term effectiveness they are applicable in only limited circumstances where other forms of control become the primary abatement measures.

(ii) Surface applications

Repeated surface applications of sprays are considered ineffective due to selective infiltration and the very short period for which the sprayed bactericides are effective.

COLLECTION AND TREATMENT

(i) Chemical treatment

AMD collection and chemical treatment is currently the most widely practiced AMD abatement measure. It represents mature and known technology, and it is functioning to successfully abate AMD impacts on the environment. The degree of abatement depends largely on the percentage of the surface and groundwater AMD that can be intercepted and collected. This measure is most successful where groundwater conditions are such that seepage releases are naturally concentrated and guided to a surface discharge.

Accumulation of sludge, its storage and long-term stable containment, is the primary concern associated with this measure. Treatment costs often increase with time as the maximum acid generating potential of the waste pile develops and as the natural buffering capacity along the seepage path is exhausted. Treatment costs are high. Failure of the treatment plant for relatively short periods can result in large contaminant releases, requiring secure operating facilities.

Collection and treatment may be considered as an interim measure until a more secure cost effective set of abatement measures can be implemented. It may also be required as a final polishing step to improve the discharge water qualities achieved by other less costly, more reliable but less effective abatement measures.

(ii) Peat and woodwaste treatment

Alternative treatment methods using peat have been demonstrated by Sridhar et al, 1974, Kadlec and Rathbun, 1983, and Dissanoyake and Weerassoriya, 1981. While large percentage improvements can be obtained, such treatment requires large quantities of peat or woodwaste. Concerns regarding methods of channeling AMD flows to achieve efficient material consumption, the fate of the metals when the organic materials decompose, as well as the environmental impact resulting from the decomposition of the organic products render these treatment methods as experimental, at best.

WETLAND TREATMENT

Wetland treatment offers much promise as a long-term passive treatment system at those locations where wetlands can be established and maintained. Biological activity reduces in the winter and water flows through the wetland are canalized by ice formation. This considerably reduces the effectiveness of the biological activity during the winter period. Unless adequate treatment can be maintained all year round it will be necessary to store winter flows for treatment in the summer. This generally requires a large storage and treatment area. The rate of accumulation of sludges, metals and organic matter is large. This will require the periodic maintenance of the wetland. The long-term fate of the metals accumulated in the organic deposits so formed have not been determined. These deposits may themselves become the source of contaminant releases in the long-term as organic matter decomposes. Where feasible, wetlands should be considered as a final polishing step in the treatment of residual AMD following the implementation of other abatement measures. Considerable research and field trials are required to demonstrate the long-term effectiveness of this abatement measure and the long-term fate of the metals accumulated in the organic wetland deposits.

EVALUATION METHODOLOGY

The lack of reliable, accurate technology for the prediction of acid generation, the effectiveness of abatement measures and the long-term changes, makes the evaluation and selection of the most advantageous measures difficult. Much of the assessment methodology is necessarily based on judgement rather than deterministic analysis. In a recent study for the abatement of AMD from a waste dump (Steffen Robertson and Kirsten, 1987) an assessment method was used which involved consideration of each of the following factors:

- Practicality of implementing the measure
- Durability (life expectancy) of the measure
- Security/risk of failure
- Environmental impacts of constructing the measure
- Construction complexity and duration
- Inspection and maintenance requirements
- Effectiveness in reducing AMD

For each alternative abatement measure a point score was assigned for each of the above factors using a five point scale. The scale values are illustrated by an example from this evaluation in Table 1. The point score for each factor was weighted according to the importance of the factor and the total points counted. This point count was used to rank the various alternatives. The resulting ranking is shown on Table 2. A cost estimate was prepared for each measure which included the capital and operating costs for 50 years. By dividing the cost by the ranking points a measure of the cost benefit is achieved. The cost/benefit values determined for this project are also shown on Table 2. They demonstrate that some of the abatement measures, such as diversion ditches, have a comparatively large benefit at low cost, even though they are individually not capable of reducing AMD to low levels. Such measures are of particular benefit where funds available for abatement are very limited, and they should be considered as part of any AMD abatement plan.

The ranking procedure was used for the selection of the most advantageous alternatives, which were then investigated and evaluated in greater detail.

LONG-TERM MONITORING AND MAINTENANCE

It should be the objective of all AMD abatement plans to reduce monitoring and maintenance to <u>a minimum commensurate with the available technology and long-term costs</u>. The current state of the art of long-term AMD abatement is that a condition of no monitoring and no maintenance may not be achievable, even at very high cost, for some of the existing waste deposits. The perpetual forces of erosion, weathering, root action, frost action and burrowing activities of biota (insects, animals and man), as well as a host of lesser effects, will result in disruption of abatement measures with time (refer Steffen Robertson and Kirsten, 1986a). To avoid the detrimental effect of these disruptions it is considerably more cost effective to provide for a measure of maintenance, rather than to attempt to construct structures that will operate without maintenance, in perpetuity.

TABLE I ABATEMENT OPTION: SYNTHETIC LINER OPTION

ACID ABATEMENT ALTERNATIVES RANKING SYSTEM

	Consideration	Description	Scale					Weighting Factor	Point Total	Cost/Effect Score
			Low		Medium		High			
1.	Practicality	Technical feasibility	1	2	3	4	5	x4	20	
		years	1	10	50	100	>100			
2.	Durability	Life expectancy	1	2	3	4	5	x2	10	
			High		Medium		Low			
3.	Security/Risk	Risk of failure	1	2	3	4	5	x4	12	
4.	Environmental Impacts	Adverse impacts due to construction or operation								
			High		Medium		Low			
		(a) Construction	1	2	3	4	5	x1	4	
		(b) Operation	1	2	3	4	5	x1	5	
			Maximum				Minimum			
5.	Construction	a) Complexity, magnitude	1	2	3	4	5	x2	6	
		months	>6	4-6	2-4	1-2	<1			
		b) Duration of construction	1	2	3	4	5	x1	3	
			Wkly	Month	Quat.	Ann.	>Ann.			
6.	Inspection and Maintenance	a) Frequency of inspection	1	2	3	4	5	x1	4	
			High	Mod.	Low	V.Low	None			
		b) Level of maintenance	1	2	3	4	5	x5	20	
			<75	75-90	90-95	95-100 less secure	95-100 more secure			
7.	Effectiveness	Reduction of contamination	0	2	3	4	5	x10	40	100
		TOTAL RANKING POINTS							124	

Cost/Benefit = $\frac{\text{Preliminary Total Estimate}}{\text{Total Ranking Points}} = \frac{\$820,000}{124} = \$6612/\text{Point}$

TABLE 2

OPTION RANKING ACCORDING TO TOTAL OF RANKING POINTS

Rank	Points	Options $/Point	Cost/Benefit
HIGH PROBABILITY OF SUCCESS (>95% REDUCTION)			
1	124	Synthetic Liner	6,612
2	97	Chemical Treatment	11,948
MODERATE PROBABILITY OF SUCCESS (75-95% REDUCTION)			
1	103	Compacted Till Cover	2,720
2	87	Removal of Pyritic Waste to Lakes	13,655
3	82	Alkaline Trenches	10,609
4	79	Waste Removal and Mixing	17,632
LOW PROBABILITY OF SUCCESS (<75% REDUCTION)			
1	75	Diversion Ditches	367
2	73	In-Situ Seep Neutralization	1,643
3	70	Surface Application of Limestone	12,471
4	62	Limestone Barriers	6,532
5	57	Topsoil and Revegetation	4,052
6	56	Wetland	8,503
7	40	Peat	6,562
8	36	Woodwaste	12,471
9	34	Bactericides	21,617

Further it is noted that the present value of maintenance to be undertaken in 50 to 100 years time is small. Thus it is justifiable, by conventional economic principles, to provide for such very, long-term monitoring and maintenance with relatively low levels of present funding. Acceptance of this principle allows for the provision of maintenance in the design of AMD abatement measures. Under these circumstances some of the alternative abatement measures, which might otherwise be rejected as being of too short a duration, become more advantageous. An example is the use of synthetic liners to cover waste. A thick (2 mm) HDPE membrane liner, appropriately installed as part of a composite cover, may be expected to last for 50 to 100 years or more. The cost of establishing a fund for the replacement of the membrane at intervals of 50 to 100 years is relatively small.

ABANDONMENT PLAN DEVELOPMENT PROGRAMS

A current trend in regulatory control is to require that abandonment plans include adequate AMD abatement measures, in order to preclude AMD impact on the environment in the long-term. It may also be required that the abatement measures should be secure and maintenance free (in perpetuity). Given the present state of knowledge and analytical tools, both requirements may not be achievable.

However, with provision for maintenance and contingent treatment, the preclusion of AMD impacts are possible for the very long periods of interest. Many of the currently proposed abatement measures may turn out to provide adequate long-term abatement, but it may not be possible to demonstrate this conclusively with current technology.

The development and testing of AMD abatement measures is a rapidly evolving technology. It will be spurred by the research proposed in programs such as the Canadian Reactive Acid Tailings Studies (RATS) program (Ferguson, 1987). To allow appropriate new development and technology to be included in future abandonment plans, it is recommended that regulatory authorities should not insist on rigidly specified abandonment plans written into operating permits, but that they adopt a more flexible approach of requiring the implementation of appropriate abandonment plan development programs with defined objectives. Such programs should provide for field trials to demonstrate the effectiveness of AMD abatement measures, prior to their being incorporated in abandonment plans. This approach to regulation is now likely to ensure that the funds provided for abandonment measures are spent in the most cost effective manner.

The recommended approach is not unusual when compared to other fields of geotechnical engineering. In the field of earth dam engineering, for example, it has long been the practice to build to a flexible design that is monitored by instrumentation. The details of the design may be changed during construction should conditions in the field turn out different from what was anticipated during early design stages. This is known as the Observational Method (Peck, 1969). When applied to AMD it is likely to be a more pragmatic and successful approach, rather than trying to proscribe for contaminant control.

Specifying performance criteria (or end-product specifications) in terms of certain numbers to be met by law may be unrealistic. It may be simply unworkable because our understanding of the problem in the long-term is as yet too imprecise. It is better to require the mining industry to work to specific design criteria and regulate at this point, rather than be faced with the frustration of a *fait accompli* that does not meet up to performance specifications set out in statutes. In practical terms this alternative approach means working to design criteria aimed at limiting or eliminating acid generation (Smith and Van Zyl, 1983). These criteria are exclusion or limitation of oxygen availability, water ingress/infiltration and bacterial activity. Methods of achieving these criteria are described in their paper together with case histories.

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WILL IT GENERATE AMD?

AN OVERVIEW OF METHODS TO PREDICT ACID MINE DRAINAGE

By K. D. Ferguson[1] and P. M. Erickson[2]

INTRODUCTION

Many believe that mining effluents always contain metals. While technically correct most mine effluents contain either high or low levels though the concentration and loading of metals varies from mine to mine. Acid mine drainage (AMD) is usually the cause of high metal concentrations in contaminated effluents. Mines that have AMD have difficulty in meeting environmental standards and usually expend large financial resources on pollution control. It is imperative that mining companies and regulatory agencies attempt to predict AMD prior to mining so that mines can be designed that are cost effective and compatible with the environment. If AMD is generated, then companies should identify the source so that mitigation measures can be accomplished for minimal cost. In this paper, we discuss the prediction of AMD prior to mining and the identification of acid sources after drainage has become contaminated.

Formation of AMD

Prediction of acid mine drainage must be based on a thorough knowledge of the process of formation and the factors that control it. Unfortunately, the process is complex and not completely understood

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despite several decades of intense study. The knowledge base is considerable, however, and growing as illustrated below.

Kleinmann et al., 1981, identified three stages in the generation of acid mine drainage. In the first stage, chemical and, or, biological oxidation of pyrite and other sulfide minerals slowly produces acid (Figure 1). This acid may initially be neutralized by carbonate minerals in the rock. After the carbonates are consumed, the process enters the second stage. As the pH drops in the microenvironment around the sulfide minerals, populations of acid loving (acidophilic) bacteria such as Thiobacillus ferrooxidans begin to multiply. These bacteria cause a further decrease in pH, causing the process to enter the third stage. When the pH around the sulphides decreases to below 3, ferric iron remains in solution. Ferrous iron is converted by the bacteria to ferric iron which in turn oxidizes the sulphide minerals. The rate of acid formation is rapid in the last stage and is limited by the concentration of ferric iron.

The four equations in Figure 1 generally describe the acid production process with pyrite (FeS_2) as occurs at coal mines, but, describe only a few of the many sulphide oxidation reactions at metal mines. The mix of sulphide minerals at metal mines accounts for the diversity in AMD quality.

Ferguson and Erickson (1986) classified the factors that control AMD formation as primary, secondary, and tertiary. The primary factors are those directly involved in the acid production process. Secondary factors control the consumption or alteration of the products from the acid generation reactions, while tertiary factors are the physical aspects of the waste materials or minesite that influence acid production, migration, and consumption.

The primary factors include pyrite and other sulfide minerals, oxygen, water, ferric iron, and iron-oxidizing bacteria (Figure 1). The physical relationship of these factors in an acid generating mine spoil is shown in Figure 2. Here, water and oxygen are provided by the external environment to the acid generating sites within the spoil. The massive transfer of reactants supports rapid acid production.

Regardless of the presence of pyrite and other sulfide minerals, AMD may not be a problem at a minesite due to secondary factors. The most important secondary factor is the neutralization of acid by alkalinity released from carbonate minerals in the mine waste, such as calcite ($CaCO_3$) and dolomite [$CaMg(CO_3)_2$] as described previously. The neutralization by calcite of acidity produced from pyrite oxidation may be represented as follows (Williams et al., 1982):

$$FeS_2(s) + 2CaCO_3(s) + 15/4O_2(g) + 3/2H_2O \rightarrow Fe(OH)_3(s) + 2SO_4^{2-} + 2Ca^{2+} + 2CO_2(g)$$

Other secondary factors include alteration of oxidation products by further reaction, such as ion exchange on clay surfaces, gypsum precipitation, and acid-induced dissolution of other minerals.

The physical characteristics of a mining waste, the spatial relationship between wastes, and the hydrologic regime are some of the tertiary factors that affect the AMD process. The important physical characteristics include particle size, physical weathering tendency, and mine waste permeability. The rate of acid production is a function of the particle surface area since that property reflects the amount of sulphide exposed for reaction.

Water serves several roles in the acid production system, acting as a reactant, as a reaction medium, and as a product transport solvent.

The overriding principle in the generation of acid mine water is that if the alkalinity released by minerals or already present in the contacting water exceeds the acidity produced then the system remains neutral or basic and the process does not enter Stage II. In this case, the bacteria responsible for catalyzing the acid generation reactions are inhibited and the solubilities of ferrous and ferric iron are reduced - both factors inhibit acid production.

The prediction of acid mine drainage involves the consideration of the few important factors that might control the acid production and consumption reactions at a minesite and the synthesis into a qualitative statement forecasting conditions during and after mining. Models and testing techniques have been developed to aid investigations in making this forecast.

Pre-Mine Prediction of AMD

Methods to predict acid mine drainage can be divided into five groups (Ferguson and Erickson, 1986).

1. geographical mining comparisons
2. paleoenvironmental and geological models
3. geochemical static tests
4. geochemical kinetic tests
5. mathematical models

Geographical mining comparisons involve examining and sampling drainage from abandoned or operating mines near the project under study. Simple extrapolations are made for similar mining operations in related geological environments.

Paleoenvironmental models examine the regional depositional environment of the sulphide minerals. Pyritic sulphur formed in marine or brackish water may have a greater tendency to generate AMD than that formed in fresh water environments (Caruccio et al., 1977, Williams et al., 1982). Geological models of a sedimentary sequence or igneous/ metamorphic orebody are valuable in estimating the location and size of potentially acid generating zones.

Geochemical static tests simply compare the amount of potentially acid generating sulphides to acid consuming carbonates in a rock sample (Figure 3). The acid/base accounting and APP: Sulphur Ratio tests are used frequently for coal mines in the United States while the B.C. Research Initial Test is used frequently for metal mines in Canada. The acid potential determinations are similar for the three tests, but, the neutralization potentials are different. The acid/base account neutralization potential involves a stronger digestion of carbonate minerals and may produce a slightly higher value than the other two procedures (Ferguson, 1982, Caruccio et al., 1977).

The acid/base account includes a format for presenting AMD prediction data useful for surface coal mines disturbing many lithologic groups (Figure 4). The net neutralization potentials of the individual groups may be used to develop a waste management plan for attempting to prevent AMD formation.

These tests are called static as they do not consider the relative rates of acid production and consumption. The rate of release of acidity from sulphide oxidation, and alkalinity from carbonate dissolution may be quite different and could affect the generation of acid mine drainage. Geidel (1979) suggests that the release of alkalinity is limited to a maximum value by the partial pressure of carbon dioxide while the buildup of acidity is not constrained. Various investigators have criticized the use of static tests and suggested they are only applicable to obvious cases where the quantity of acid producing or acid consuming material is in significant excess (Caruccio, et al., 1980; Williams et al., 1982; SENES, 1984). However, Ferguson and Erickson (1986) in their review of AMD prediction techniques, found that static tests were often accurate in predicting drainage quality and were particularly valuable as screening tests to determine if more sophisticated procedures should be used.

Static tests have been modified to account for possible differences in pyrite reactivity at coal mines. Caruccio et al., (1977) used reflected light microscopy to determine the morphology of pyrite which correlated with drainage quality from leaching tests. The fine grained "framboidal" pyrite was found to be the most reactive. Hammack (1986) used an evolved gas technique to provide a rapid determination of pyrite availability for oxidation. Samples were placed in a furnace, heated in an oxidizing atmosphere, and the evolved sulphur dioxide monitored at various temperatures. The pyrite oxidized from 150° C to 555° C, but only that which ignited below 220° C significantly affected drainage quality in parallel laboratory leaching tests.

Kinetic test methods attempt to model the acid producing and consuming processes in the laboratory or field. Various protocols have been developed as shown in Table 1. Kinetic tests might be expected to more accurately model the acid production and consumption processes though interpretation can be difficult. Kinetic tests are not commonly used because of their higher cost and longer duration, compared to static tests. B.C. Research has developed a type of kinetic test which assesses the ability of bacteria to oxidize sulphide minerals in a rock sample (Test 1 in Table 1). This test is of particular value in determining whether a mineral constituent of the waste is toxic to the microorganisms and inhibits their growth. Other kinetic tests follow the change in leachate quality over time and use the results in a physical or mathematical model of the AMD process. Mathematical models have recently been developed to quantitatively predict mine drainage quality (Caruccio, 1984; Jaynes et al., 1984; Ricca and Shultz, 1979; Halbert et al., 1983: Nicholson, 1984; Cathles, 1979; Davis and Ritchie, 1983). These models are rather complex and have not been extensively verified or used for prediction to date.

A major difficulty with prediction techniques is the lack of verification of test results with field experience. Two recent studies, one of metal mines and one of coal mines, examined the accuracy of some AMD prediction techniques.

Static (acid/base accounting) and kinetic (humidity cell) tests were conducted on 22 rock samples from seven metal mines in British Columbia and Yukon (Ferguson, 1985). Samples of tailings and waste rock were obtained from both active and abandoned minesites. The total sulfur contents ranged from 0.13 to 49.2 percent, and neutralization potentials ranged from 0 to 258 tonnes $CaCO_3$ equivalent per 1,000 tonnes of material. The very high percent sulfur contents of some samples yielded a definite prediction for acid production according to static methods. The static and kinetic tests correctly predicted the formation of AMD in all but six cases - three were incorrect and three were inconclusive. A few results from the kinetic tests (humidity cells) were difficult to interpret due to indefinite pH trends. The results supported the use of static tests for predicting AMD, but suggested there was only a poor relationship between the magnitude of the net base value (net neutralization potential) and the amount of acidity produced during the initial stage of the AMD process. The static tests may be useful for qualitative AMD predictions but not for estimates of actual mine water quality.

Correlations between the static test results (percent sulphur, neutralization potential, and net neutralization potential) and the kinetic test results (sulphate production rate, cumulative acidity production, initial leachate pH and final leachate pH) provided some insight into the geochemical processes involved in acid generation. As expected, the sulphate production rate increased with percent sulphur and was a good indicator of the sulphide oxidation rate. The initial pH of the leachate decreased with increased percent sulphur probably reflecting an increase in soluble metal salts in the samples at the start of the experiment. The cumulative acidity production and final leachate pH, however, correlated with the neutralization potential rather than the net neutralization potential as might be expected from static test prediction theory. In these tests, the quantity of neutralizing minerals in the samples appeared to play the critical role in initially controlling the acid generation process. In the longer term, acid production from sulphide minerals would be expected to "outlast" the alkalinity from carbonate minerals in potentially acid generating rocks and the AMD process would enter Stage II. These findings agree with results from other short-term (up to 6 months) kinetic tests where little acid was produced from samples exhibiting a strong acid potential according to static tests. Trends in sulphate production, acidity, and pH from kinetic tests must be examined together with the static test results to forecast the long term leachate quality at a mine.

An analogous study evaluated the predictive capability of static (acid-base account) and kinetic (a modified humidity cell procedure) tests applied to coal overburden (Erickson, 1986). Sites were deliberately selected to focus on hard-to-predict cases. In the preliminary analysis, 25 sites were analyzed by the static test; 14 of these were also subjected to the kinetic test.

A simple type of interpretation was applied to acid-base account data. The net neutralization potential (NNP) was calculated for each lithologic unit and the values for individual units were combined to generate a thickness-weighted average NNP to represent the whole overburden column. Thickness-weighted NNP values ranged from -20 to +250 tonnes $CaCO_3$ per 1,000 tonnes of overburden. These values were then compared to flow-weighted net alkalinity in drainage from reclaimed mine sections (Table 2). There is no linear correlation between observed drainage alkalinity and thickness-weighted NNP (Figure 5). NNP values greater than 33 tonnes/1,000 tonnes were always associated with alkaline drainage while NNP values less than 22 tonnes/1,000 tonnes were predominantly associated with acid drainage. The validity of a boundary between 22 and 33 tonnes/1,000 tonnes to distinguish sites that will not produce acidic drainage is uncertain because only 4 of 25 sites had higher NNP values. Further analyses are in progress to test other methods of evaluating overburden analysis data for water quality predictions.

Kinetic tests were conducted on overburden from 14 of these sites. The cumulative alkaline loads produced by each lithologic unit during the simulated weathering test were averaged for the whole overburden column on a thickness-weighted basis and compared to actual drainage quality. As with the acid-base account results, no linear correlation was found. Attempts to use the thickness-weighted alkaline load from the kinetic test to predict acid or alkaline drainage when the NNP value from the acid-base account was less than 22 tonnes/1,000 tonnes were unsuccessful.

These findings are not surprising in light of the high geochemical variability in coal overburden strata. Tests on individual lithologic units fail to account for interactions between potential acid- and alkaline-producing materials.

Uncertainty In AMD Production

The production of acid mine drainage is complex and yet, important factors can be modelled for prediction. However, uncertainty is always present and varies according to the nature of the waste material and its deposition and reclamation history. The range in uncertainty is illustrated in the four simple cases of waste rock disposal in Figure 6. Case one represents the most certain prediction where the waste is either completely composed (1a) or devoid (1b) of sulphide minerals; cases rarely seen at real mines. Uncertainty increases in case two where the sulphide content has increased or decreased from the two extremes in case one. The potential for AMD can be predicted with reasonable certainty by static tests in either high (2a) or low sulphide content (2c) samples. Predictions are more difficult when the sulphide and carbonate content are in critical balance (carbonates 3.125 times sulphides by weight) as shown in 2b. If every particle contains a balance of sulphide and carbonate minerals then theoretically, the two mineral types would be balanced. However, the maximum amount of pyrite that can be oxidized is unlimited, while carbonate dissolution is governed by equilibria. Further, coatings on both sulphide and carbonate mineral surfaces will form and affect the chemical reactions and ultimate drainage quality. Kinetic tests would be valuable in assessing reaction rates.

Homogeneous wastes are rarely found at real mines. More frequently, the acid generation potential will range reflecting spatial differences in geological environments. The numerical balance of the acid generation potential of the entire rock column would be assessed as part of the AMD prediction in acid/base accounting. This balance is important, but, the position of the waste types within the dump may ultimately determine drainage quality (Case 3). Potentially acid generating waste deposited on the exterior of a dump may generate AMD that may be only partially neutralized by carbonate minerals in the interior (3a). Coatings would form on carbonate minerals diminishing neutralization capability. In some cases, the sulphide minerals may be randomly distributed in the dump (3b). Acid generation may occur in only isolated zones and seep quality will vary depending upon its source. Potentially acid generating waste should be placed in the interior of the dump where only alkaline groundwater contacts the high sulphide waste (3c). This is AMD prevention by segregation. However, with large quantities, or highly reactive sulphides, the AMD may form despite the alkaline cover due to geochemical limitations on the maximum amount of alkalinity that can be delivered to the sulphide minerals, as discussed previously. A combination of static and kinetic tests, the latter using a simulated stacking sequence of wastes, might be valuable in assessing the relative rates of reaction.

Heterogeneous geochemistry, randomly distributed particles, and an overall sulphide/carbonate balance represents one of the most difficult predictions (Case 4). Very high uncertainty would be associated with a prediction in this case and must be supported by comprehensive kinetic tests on samples that simulate quantity and placement of various lithologic units. Case four describes the sites selected for the coal mine drainage prediction study described earlier. The inaccurate predictions identified in that study may demonstrate the role of spoil placement.

Determining the Source of AMD

The overall cost of AMD control would decrease if the most active areas within acid generating wastes could be identified and mitigation measures targeted for maximum benefit. Locating and sampling the seeps from mining wastes is usually the first step that companies and regulatory agencies take to identify acid generating zones. The pH, acidity, sulphate, metal, and microbial content, and flow of the seeps indicate the geochemical and hydrological processes within the waste. For example, sampling at an abandoned waste rock dump in British Columbia indicated that flow and quality of seeps varied, but their location remained the same; neutralization agents could be applied at these specific sites to reduce contaminant loading (Figure 7). The majority of copper released from the dump originated from two of the four drainage zones in the dump (zones BB and CC in Figure 7). Mitigation measures should be targeted to those areas.

At these sites, seasonal variation in AMD quantity and quality requires monitoring over a wide range of hydrologic conditions to quantify loadings to the receiving environment. Contaminated seeps are usually most readily identified in the spring and fall coincident with maximum melt or precipitation events following prolonged dry periods.

Yellow or red staining of rock and leachate may also indicate the source of AMD. AMD containing only ferrous iron is clear, but shows a red or yellow tint downstream as the ferrous iron is oxidized to the ferric form. AMD from metal mines may be blue-green reflecting high copper values. Dead vegetation near a seep may be an early indicator of contamination.

Characteristics of the waste such as total and sulphate sulphur, neutralization potential, and paste pH indicate the progress of AMD formation reactions. Samples of metal mine tailings can be obtained at various depths and analyzed by these geochemical tests to determine the depth of oxidation and other information useful in designing alkaline or impermeable covers for AMD control.

The partial pressure of oxygen available in the pore spaces throughout the waste indicates both oxygenated zones that can support acid production and zones where oxidation is occurring. Several oxygen profiles (O_2 partial pressure versus depth) are typical for mine wastes (Harris and Ritchie, 1985; Erickson, 1985). When oxygen is supplied by diffusion (induced by concentration differential) from the atmosphere, the profile shows decreasing oxygen with increasing depth. Active oxidation zones are reflected as sharp partial pressure gradients. When advection (induced by thermal or pressure differential) supplies oxygen to the mine waste, high partial pressures are observed at intermediate depths. Gradients are less easily correlated with active oxidation in these cases. However, the availability of oxygen in advection zones indicates that, if present, pyritic material will likely undergo oxidation.

Variations in the surface or interior temperatures of mining wastes can indicate acid generation since the reactions are exothermic. Temperatures up to 45° C were recorded by thermister strings buried in a very active acid generating waste rock dump in British Columbia (Patterson, 1986). A significant drop in temperature following reclamation of a waste rock dump was recorded by Harries and Ritchie (1986) using thermister probes in wells. Thermography (infrared imaging) was used to identify acid generating zones at three waste rock dumps and open pits in British Columbia. A related, but simpler approach, involved locating bare spots on a waste rock dump following the first snowfall of winter. In all cases, heat generation correlated with observed differences in water quality of seeps.

Remote detection of acid sources is possible with the geophysical method of electomagnetic induction measurements. This means of surveying detects terrain conductivity anomolies. In several surveys conducted at regraded coal mines, anomolies were observed and later confirmed to reflect pyrite oxidation, changes in saturation state, and disturbed area boundaries (Watzlaf and Ladwig, 1987). The method was found to be most useful as a reconnaissance tool preceding application of detailed monitoring methods.

Summary

Though the acid mine drainage process is complex and not completely understood, prediction techniques are available that are successful for many mining projects. Considerable skill and experience is required to conduct some tests and interpret results for accurate forecasts of mine drainage quality. Uncertainity is always present, and should be estimated and acknowledged in the prediction statement.

The location of acid generating zones within a mining waste can be determined by both simple and sophisticated techniques. In general, it is easier to locate the source of AMD than predict prior to mining if drainage will be acidic. Success in both of these areas of prediction are required if mining is to be conducted in an environmentally sound manner.

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TABLE 1. - KINETIC ACID MINE DRAINAGE PREDICTION TECHNIQUES

(Adapted from Ferguson, 1985 and Ferguson and Erickson, 1986)

TEST	DESCRIPTION	USE	REFERENCES
B.C. Research Confirmation Test	- Sample placed in 250 ml Erlenmeyer flask with 70 ml nutrient media, culture of Thiobacillus ferrooxidans at pH 2.2-2.5. - Flask placed on gyratory shaker at 35° C in CO_2 enriched atmopshere. - pH monitored and additional sample added. - If pH rises substantially then sample non-acid producer. - If pH remains low then sample potential acid producer.	Frequently used for metal mines in Canada	Bruynesteyn and Hackl (1984)
Shake Flasks	- Sample placed in 1L Erlenmeyer flask with 600 ml of water or nutrient solution. - Series of samples tested at various starting pH, inoculation, and temperature. - Samples incubated for up to 3 months. - Leachates analyzed weekly and bi-weekly for range of parameters.	Infrequently used for metal mines in Canada	Davidge (1984) Halbert, et al., (1983)
Soxhlet Reactor	- Standard or modified Soxhlet reactor used. - Water placed in reservoir, vaporized and passed into condenser. Condensed liquid drips into thimble with sample and thence back into the reservoir. - Leachate analyzed after 64-192 hrs. for a range of parameters.	Infrequently used in coal fields of Appalachia, U.S.A.	Sullivan and Sobek (1982) Renton (1983)
Humidity Cell	- Sample placed in plexiglass container connected to humidified air. - Humidified air passed over sample for 3 days. - Dry air passed over sample for 3 days and 200 ml of water added on seventh day. - Leachate removed and analyzed for range of parameters. - Procedure repeated for 8-10 weeks.	Relatively common in coal fields of Appalachia, U.S.A.	Caruccio et al., (1977) Caruccio et al., (1980)

TABLE 1. - KINETIC ACID MINE DRAINAGE PREDICTION TECHNIQUES (Continued)

(Adapted from Ferguson, 1985 and Ferguson and Erickson, 1986)

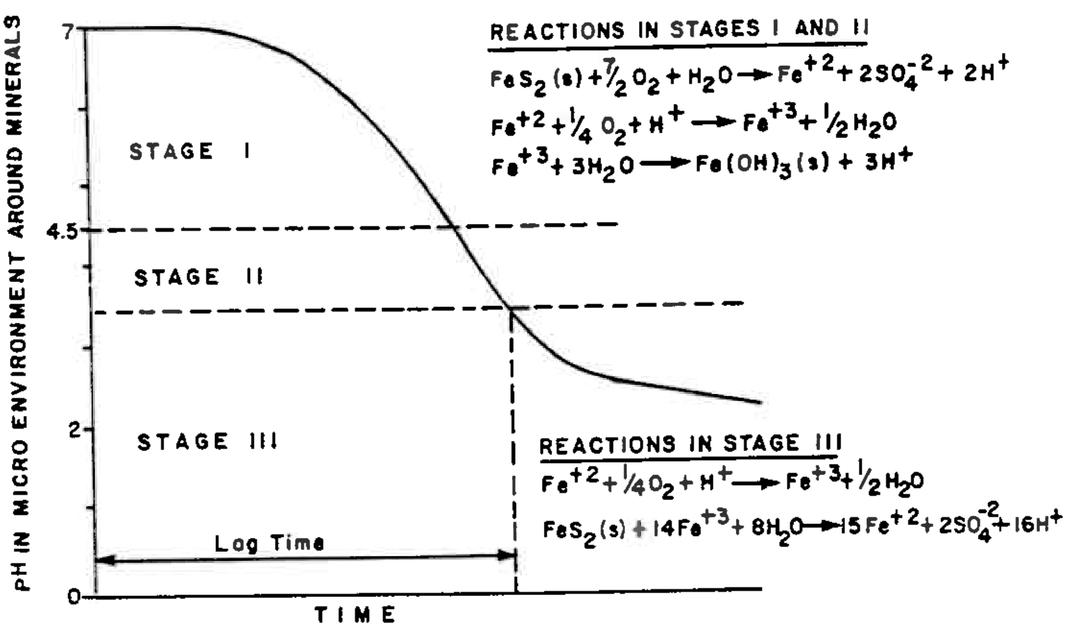
TEST	DESCRIPTION	USE	REFERENCES
5. Columns/ Lysimeters	- Sample placed in column and periodically leached by distilled water. - Samples of leachate analyzed for range of parameters. - Usually leached for 8-10 weeks minimum. - Several variations of set-up and leaching procedures in literature.	Relatively common in Canada and U.S.A.	Sturey et al., (1982) Apel (1983) Ritcey and Silver (1981)
6. Test Plots/ Pits/Piles	- Run of mine or modified sample placed on impervious surface. - Precipitation provides leachate which is collected in sump. - Samples of collected leachate analyzed for range of parameters. - Test usually run for at least 1 year.	Very infrequently used in Canada and U.S.A.	Eger et al., (1981) Murray and Okuhara (1980)

TABLE 2. - COMPARISON OF OVERBURDEN ANALYSIS AND POST-MINING DRAINAGE QUALITY FOR 25 U.S. COAL MINES[1]

SITE	ACID-BASE ACCOUNT Thickness-weighted net neutralization potential, tonnes/ 1,000 tonnes as $CaCO_3$.	MODIFIED HUMIDITY CELL Thickness-weighted cumulative net alkaline load, g/300g as $CaCO_3$.	POST-MINING DRAINAGE QUALITY Flow-weighted net alkalinity, mg/L as $CaCO_3$.
1	-1.6	--[2]	-120.
2	-11.	--	7.6
3	7.5	--	65.
4	4.8	18.	-72.
5	250.	18.	170.
6	34.	33.	110.
7	-20.	--	-340.
8	-2.9	-18.	-110.
9	-3.5	--	-85.
10	-2.9	--	-100.
11	-6.8	-170.	100.
12	3.0	2.2	-290.
13	7.4	23.	110.
14	3.2	7.7	-860.
15	22.	--	-13.
16	11.	--	-240.
17	150.	--	100.
18	-4.6	-210.	-500.
19	2.7	-24.	-190.
21	3.2	27.	-8.7
22	90.	-280.	430.
25	20.	21.	310.
27	9.8	--	52.
29	3.6	-42.	-610.
30	-0.49	--	38.

(1) All values have been rounded to 2 significant figures.
(2) Dashes indicate that the kinetic test was not performed.

STAGES IN THE FORMATION OF ACID MINE DRAINAGE



OVERALL PROCESS (Stumm and Morgan, 1981)

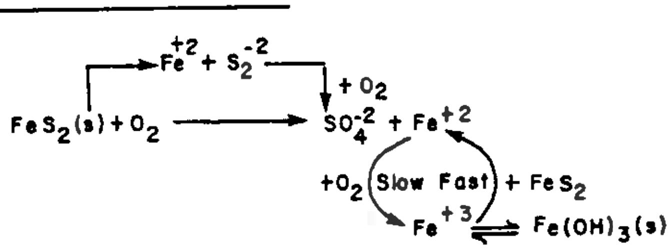


FIGURE 1 FORMATION OF ACID MINE DRAINAGE FROM PYRITE OXIDATION

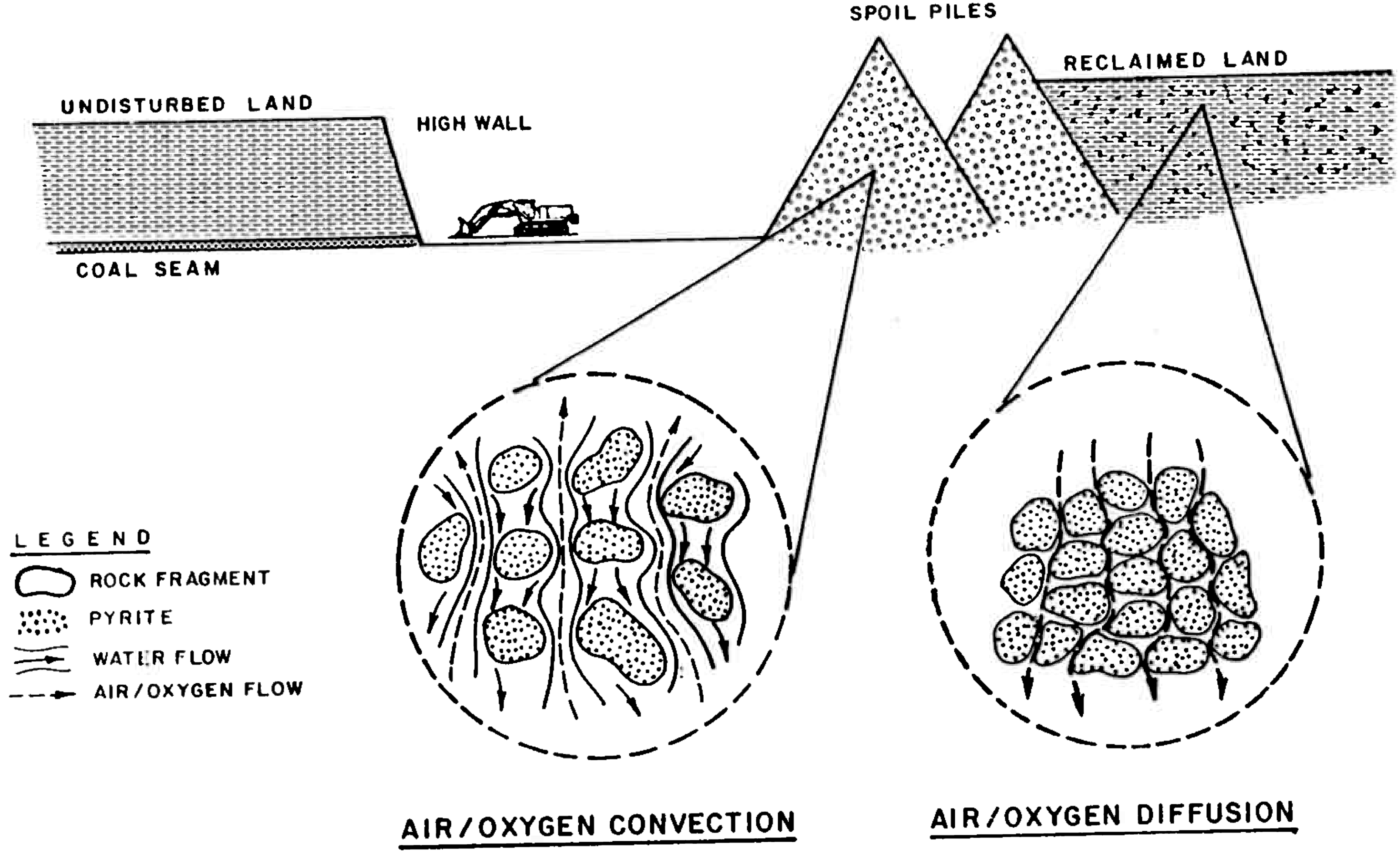


FIGURE 2 PRODUCTION OF ACID MINE DRAINAGE IN POROUS WASTE DUMPS (AFTER CATHLES 1982)

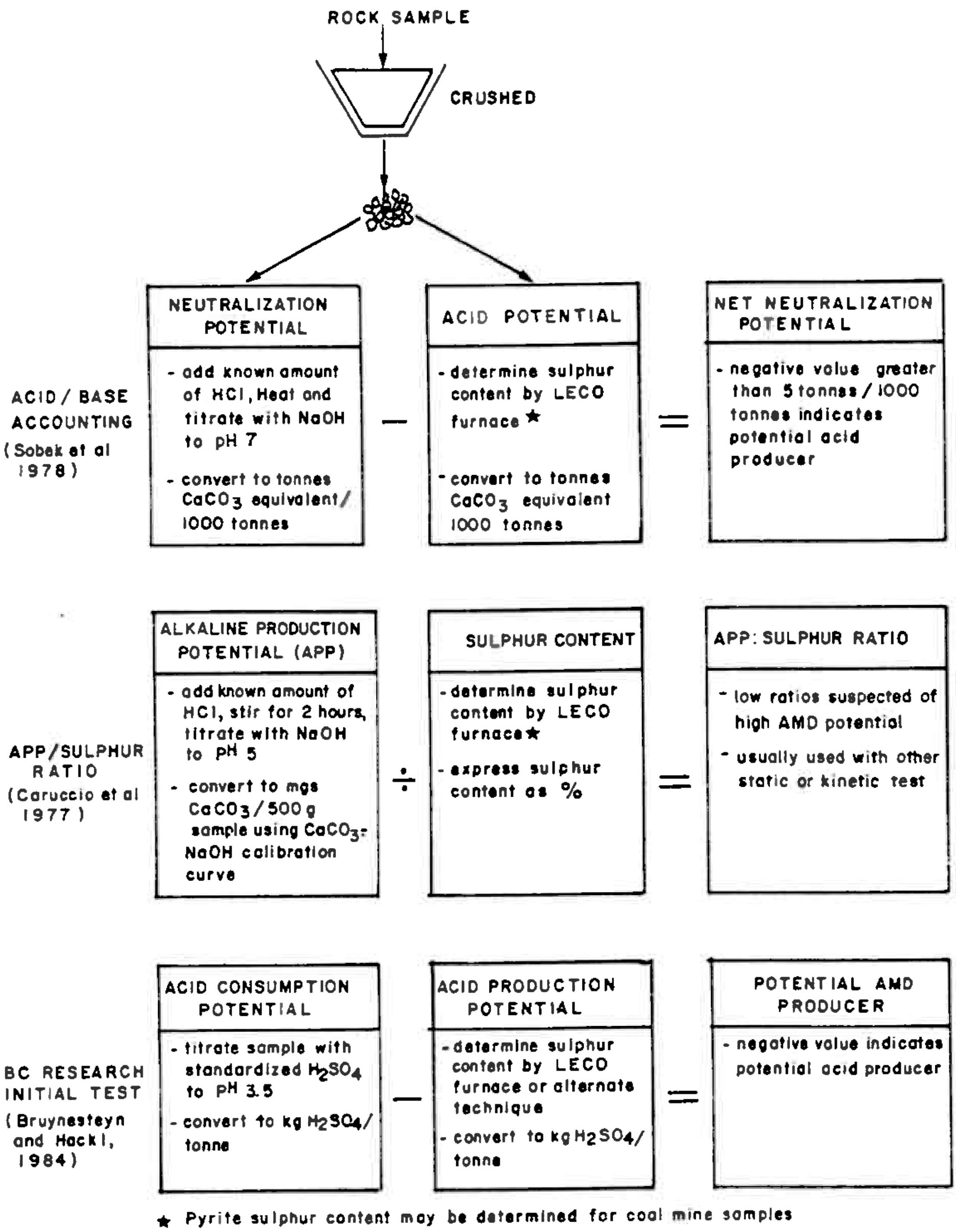


FIGURE 3 STATIC ACID MINE DRAINAGE PREDICTION TECHNIQUES

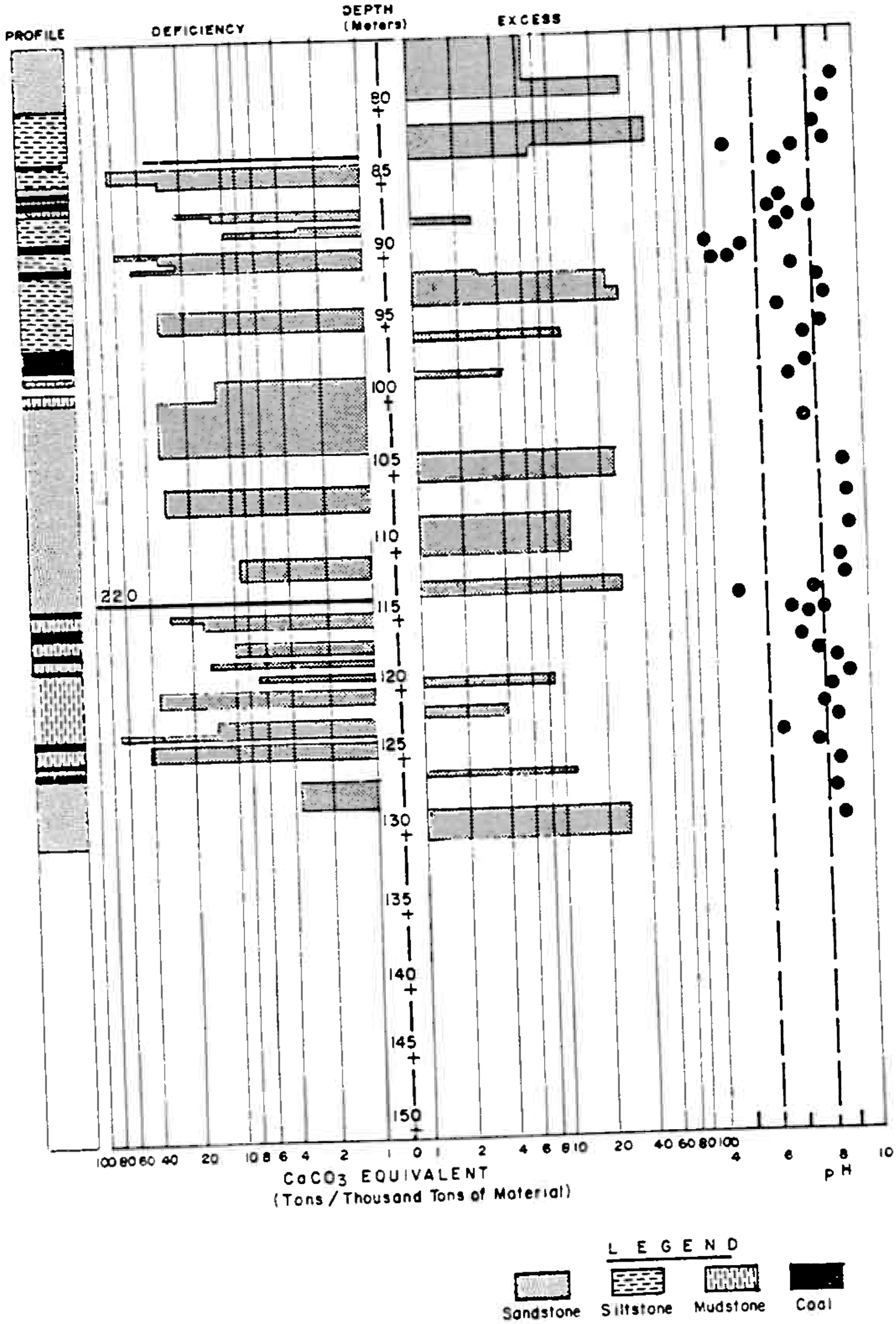


FIGURE 4 ACID/BASE ACCOUNT FOR B.C. COAL MINE (FROM STURM ENVIRONMENTAL SERVICES, 1985)

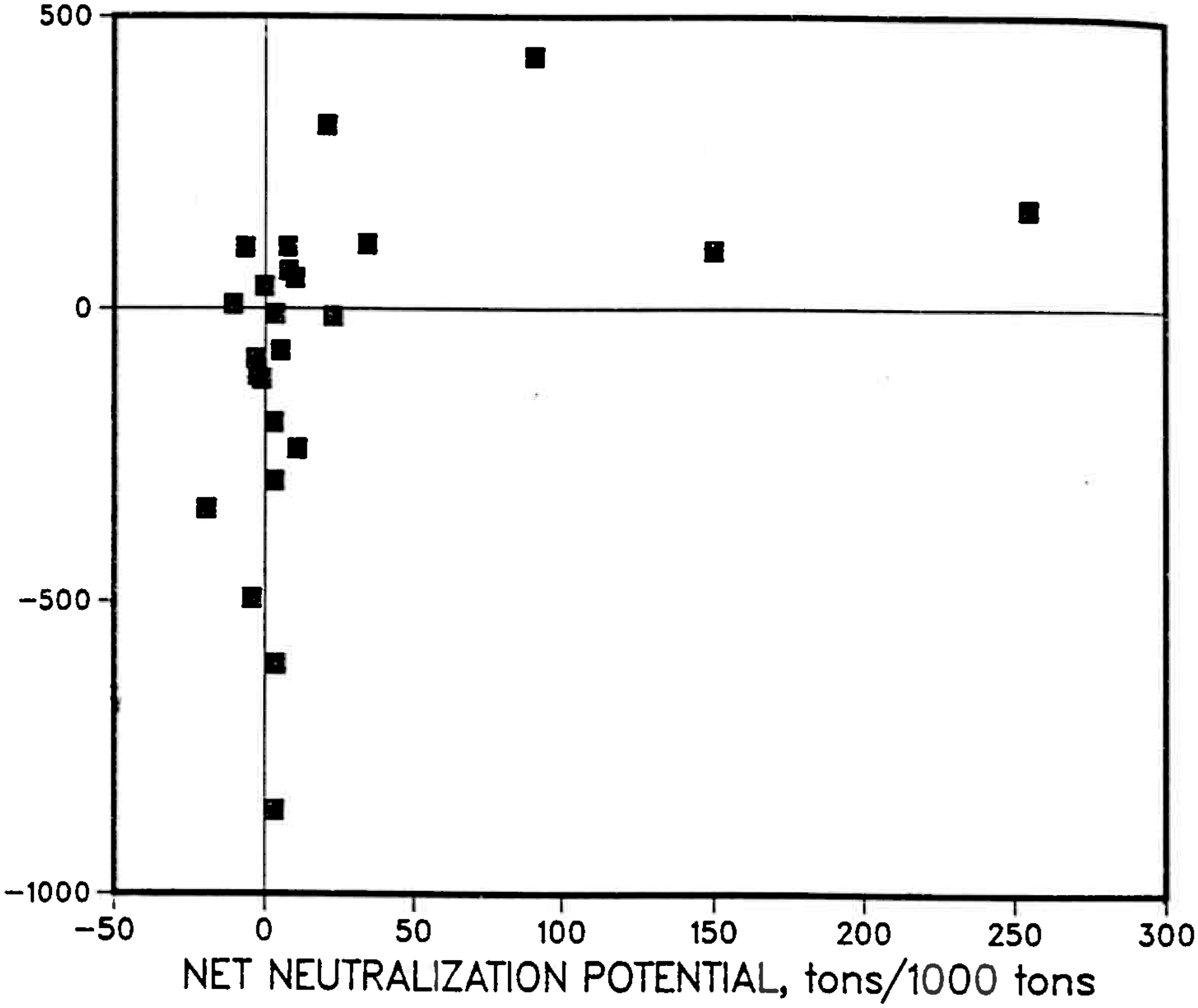


FIGURE 5 RELATIONSHIP BETWEEN POST-MINING WATER QUALITY (NET ALKALINITY) AND ACID-BASE ACCOUNT PREDICTOR (THICKNESS-WEIGHTED NET NEUTRALIZATION POTENTIAL)

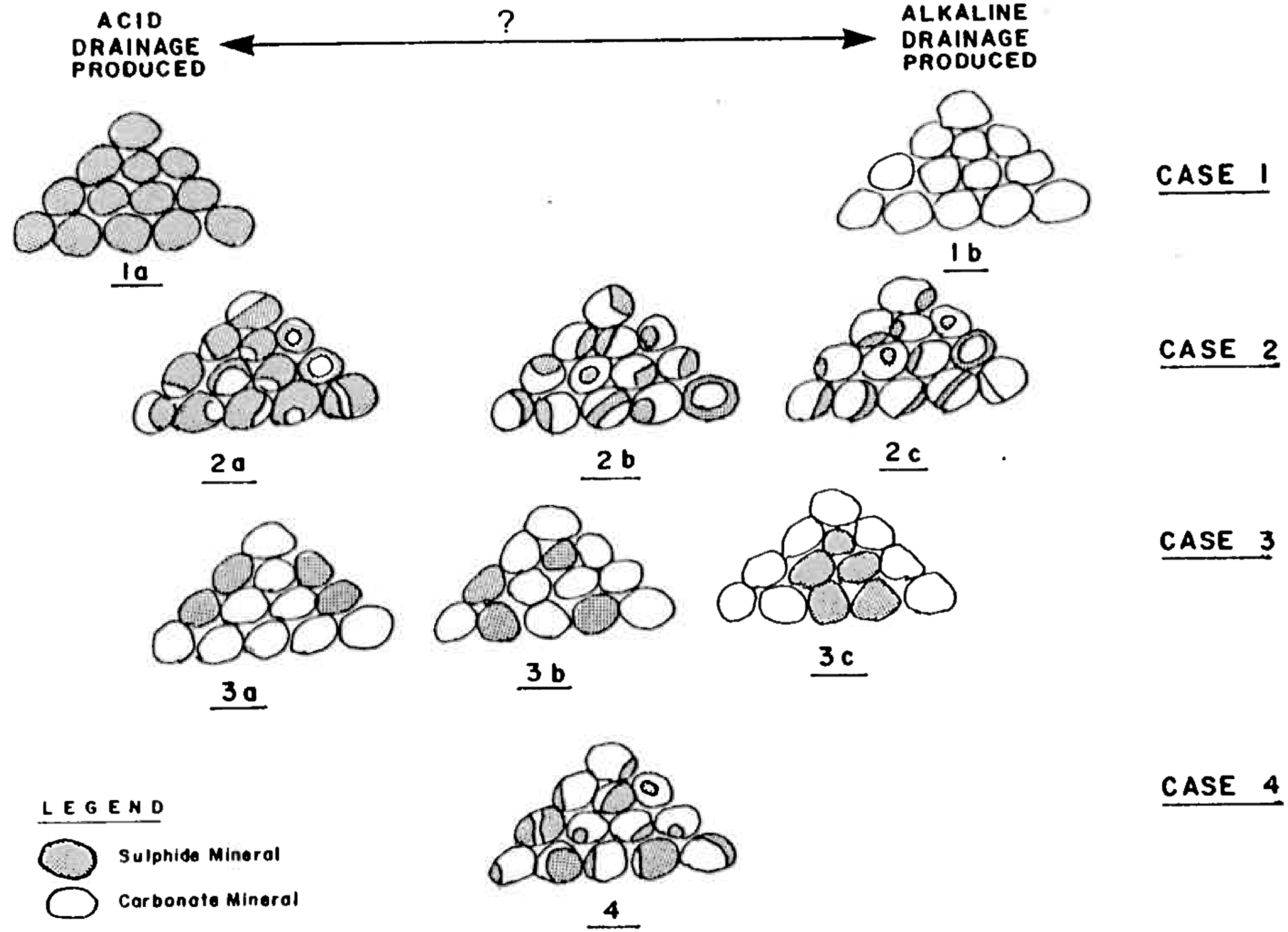


FIGURE 6 UNCERTAINTY IN AMD PREDICTION

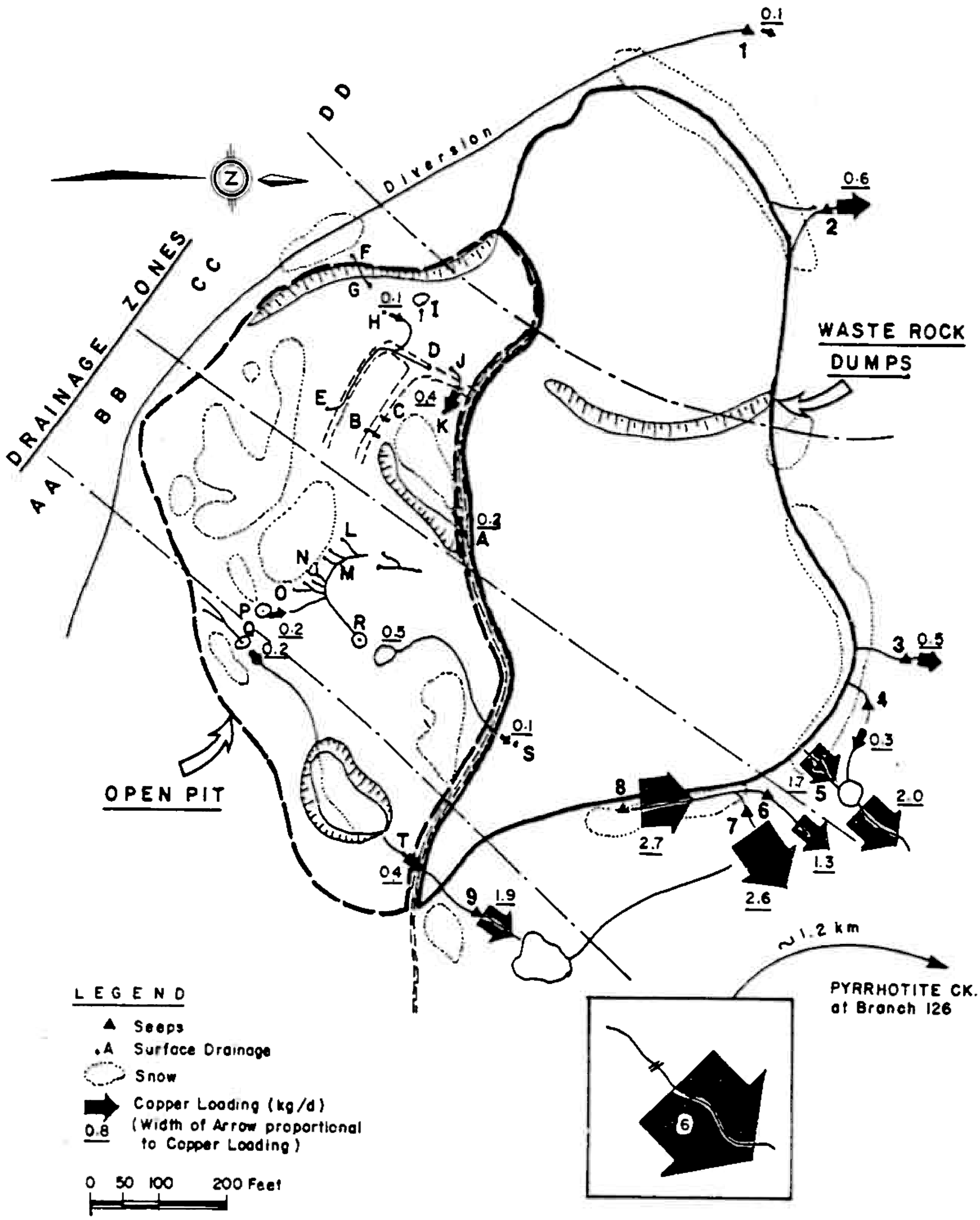


FIGURE 7 METAL MINE OPEN PIT AND WASTE ROCK DUMP - SURFACE DRAINAGE AND SEEPAGE COPPER LOADINGS

APPENDIX B: First Pages of Selected Water Quality Standards

United States
Environmental Protection
Agency

Office of Water
Regulations and Standards
Criteria and Standards Division
Washington, DC 20460

EPA 440/5-84-031
January 1985

PB85-227023

Water

EPA

Ambient Water Quality Criteria for

Copper - 1984

(1978) reported the results of various field studies with poorly defined or experimentally confounded exposure conditions. Papers by Baudouin and Scoppa (1974), Dodge and Theis (1979), Evans (1980), Furmanska (1979), Muramoto (1980, 1982), and Verma, et al. (1980) contain too few experimental details to allow interpretation of the results. Bringmann and Kuhn (1982) cultured Daphnia magna in one water and conducted tests in another water. Smith and Heath (1979) only reported results graphically. Shcherban (1977) did not report usable results, and Brkovic-Popovic and Popovic (1977a,b) used questionable dilution water. Data were not used if mortality in the controls was too high (Ho and Zubkoff, 1982; Hullsom, 1983; Watling, 1981, 1982, 1983). High control mortalities occurred in all except one test reported by Sauter, et al. (1976). Control mortality exceeded 10% in one test by Mount and Norberg (1984). The 96-hr values reported by Buikema, et al. (1974a,b) were subject to error because of possible reproductive interactions (Buikema, et al. 1977). Bioconcentration factors could not be calculated from the data of Anderson and Spear (1980a).

Summary

Acute toxicity data are available for species in 41 genera of freshwater animals. At a hardness of 50 mg/L the genera range in sensitivity from 16.74 µg/L for Ptychocheilus to 10,240 µg/L for Acroneuria. Data for eight species indicate that acute toxicity decreases as hardness increases. Additional data for several species indicate that toxicity also decreases with increases in alkalinity and total organic carbon.

Chronic values are available for fifteen freshwater species and range from 3.873 µg/L for brook trout to 60.36 µg/L for northern pike. Fish and

CANADIAN WATER QUALITY GUIDELINES

prepared by the
Task Force on Water Quality Guidelines
of the
Canadian Council of Resource
and
Environment Ministers

March 1987

Table of Contents

Pollution Control Objectives

for

The Mining, Smelting and Related Industries

of

British Columbia

1979

TABLE V Objectives for the Discharge of Final Effluents to Marine and Fresh Waters

Parameter (mg/L dissolved in effluent unless otherwise stated) (5)	Range	
Total suspended solids (1) (2)	25	75
Total dissolved solids	2500	5000
Toxicity (96-hour LC 50 static bioassay) (3)	100%	80%
pH (pH Units)	6.5 - 8.5	6.5 - 10
Radioactivity: (6)		
Gross Alpha pCi/L	10	100
Radium^{226} pCi/L (dissolved in effluent passing through a 3 μm filter)	less than 10	
Specific elements and compounds:		
Aluminum (Al)	0.5	1.0
Ammonia (as N)	1.0	10.0
Antimony (Sb)	0.25	1.0
Arsenic (as trivalent As)	0.05	0.25
Arsenic (total dissolved)	0.10	1.0
Cadmium (Cd)	0.01	0.1
Chromium (Cr)	0.05	0.3
Cobalt (Co)	0.5	1.0
Copper (Cu)	0.05	0.3
Cyanide (as CN)	0.1	0.5
Fluoride (F)	2.5	10.0
Iron (Fe)	0.3	1.0
Lead (Pb)	0.05	0.2
Manganese (Mn)	0.1	1.0
Mercury (Total) (Hg) (4)	Nil	0.005
Molybdenum (Mo)	0.5	5.0
Nickel (Ni)	0.2	1.0
Nitrite/Nitrate (as N)	10.0	25.0
Phosphate (Total P biologically available in effluent)	2.0	10.0
Selenium (Se)	0.05	0.5
Silver (Ag)	0.05	0.5
Uranyl (UO_2)	2.0	5.0
Zinc (Zn)	0.2	1.0
Oil and Grease (Total)	10.0	15.0

(1) Not applicable to approved direct discharge of tailing solids.

(2) Variances may be allowed during periods of excess runoff.

(3) Bioassay on salmonid species.

(4) Natural background concentration will be assessed.

(5) Analysis for Total Elements in tailings may be required prior to and during operations and the Director would give consideration to this information when issuing a permit.

(6) To apply to operations where the objective is not the mining of radioactive ores.

federal register

Friday
November 28, 1980

Part V

Environmental Protection Agency

Water Quality Criteria Documents; Availability

[illegible]MENTAL PROTECTION [illegible]CY

[illegible]1623-3]

[illegible] Quality Criteria Documents; [illegible]ailability

AGENCY: Environmental Protection Agency.

ACTION: Notice of Water Quality Criteria Documents.

SUMMARY: EPA announces the availability and provides summaries of water quality criteria documents for 64 toxic pollutants or pollutant categories. These criteria are published pursuant to section 304(a)(1) of the Clean Water Act.

AVAILABILITY OF DOCUMENTS: Summaries of both aquatic-based and health-based criteria from the documents are published below. Copies of the complete documents for individual pollutants may be obtained from the National Technical Information Service (NTIS), 5285 Port Royal Road, Springfield, VA 22161, (703-487-4650). A list of the NTIS publication order numbers for all 64 criteria documents is published below. These documents are also available for public inspection and copying during normal business hours at: Public Information Reference Unit, U.S. Environmental Protection Agency, Room 2404 (rear), 401 M St., S.W., Washington, D.C. 20460. As provided in 40 CFR Part 2, a reasonable fee may be charged for copying services. Copies of these documents are also available for review in the EPA Regional Office libraries.

Copies of the documents are not available from the EPA office listed below. Requests sent to that office will be forwarded to NTIS or returned to the sender.

1. Acenaphthene, PB81-117269.
2. Acrolein, PB81-117277.
3. Acrylonitrile, PB81-117285.
4. Aldrin/Dieldrin, PB81-117301.
5. Antimony, PB81-117319.
6. Arsenic, PB81-117327.
7. Asbestos, PB81-117335.
8. Benzene, PB81-117293.
9. Benzidine, PB81-117343.
10. Beryllium, PB81-117350.
11. Cadmium, PB81-117368.
12. Carbon Tetrachloride, PB81-117376.
13. Chlordane, PB[illegible]-117384.
14. Chlorinated benzenes, PB81-117392.
15. Chlorinated ethanes, PB81-117400.
16. Chloroalkyl ethers, PB81-117418.
17. Chlorinated naphthalene, PB81-117426.
18. Chlorinated phenols, PB81-117434.
19. Chloroform, PB81-117442.
20. 2-chlorophenol, PB81-117459.
21. Chromium, PB81-117467.
22. Copper, PB81-117475.
23. Cyanides, PB81-117483.
24. DDT, PB81-117491.
25. Dichlorobenzenes, PB81-117509.
26. Dichlorobenzidine, PB81-117517.
27. Dichloroethylenes, PB81-117525.
28. 2,4-dichlorophenol, PB81-117533.
29. Dichloropropanes/propenes, PB81-117541.
30. 2,4-dimethylphenol, PB81-117558.
31. Dinitrotoluene, PB81-117566.
32. Diphenylhydrazine, PB81-117731.
33. Endosulfan, PB81-117574.
34. Endrin, PB81-117582.
35. Ethylbenzene, PB81-117590.
36. Fluoranthene, PB81-117608.
37. Haloethers, PB81-117616.
38. Halomethanes, PB81-117624.
39. Heptachlor, PB81-117632.
40. Hexachlorobutadiene, PB81-117640.
41. Hexachlorocyclohexane, PB81-117657.
42. Hexachlorocyclopentadiene, PB81-117665.
43. Isophorone, PB81-117673.
44. Lead, PB81-117681.
45. Mercury, PB81-117699.
46. Naphthalene, PB81-117707.
47. Nickel, PB81-117715.
48. Nitrobenzene, PB81-117723.
49. Nitrophenols, PB81-117749.
50. Nitrosamines, PB81-117756.
51. Pentachlorophenol, PB81-117764.
52. Phenol, PB81-117772.
53. Phthalate esters, PB81-117780.
54. Polychlorinated biphenyls (PCBs), PB81-117798.
55. Polynuclear aromatic hydrocarbons, PB81-117806.
56. Selenium, PB81-117814.
57. Silver, PB81-117822.
58. Tetrachloroethylene, PB81-117830.
59. Thallium, PB81-117848.
60. Toluene, PB81-117855.
61. Toxaphene, PB81-117863.
62. Trichloroethylene, PB81-117871.
63. Vinyl chloride, PB81-117889.
64. Zinc, PB81-117897.

FOR FURTHER INFORMATION CONTACT: Dr. Frank Gostomski, Criteria and Standards Division (WH-585), United States Environmental Protection Agency, 401 M Street, S.W., Washington, D.C. 20460, (202) 245-3042.

SUPPLEMENTARY INFORMATION:

Background

Pursuant to section 304(a)(1) of the Clean Water Act, 33 U.S.C. 1314(a)(1), EPA is required to periodically review and publish criteria for water quality accurately reflecting the latest scientific knowledge:

(A) on the kind and extent of all identifiable effects on health and welfare including, but not limited to, plankton, fish, shellfish, wildlife, plant life, shorelines, beaches, esthetics, and recreation which may be expected from the presence of pollutants in any body of water, including groundwater, (B) on the concentration and dispersal of pollutants, or their byproducts, through biological, physical, and chemical processes, and (C) on the effects of pollutants on biological community diversity, productivity, and stability, including information on the factors affecting rates of eutrophication and rates of organic and inorganic sedimentation for varying types of receiving waters.

EPA is today announcing the availability of criteria documents for 64 of the 65 pollutants designated as toxic under section 307(a)(1) of the Act. The document on TCDD (Dioxin) will be published within the next month after review of recent studies. Criteria for the section 307(a)(1) toxic pollutants being published today will replace the criteria for those same pollutants found in the EPA publication, *Quality Criteria for Water*, (the "Red Book.") Criteria for all other pollutants and water constituents found in the "Red Book" remain valid. The criteria published today have been derived using revised methodologies for determining pollutant concentrations that will, when not exceeded, reasonably protect human health and aquatic life. Draft criteria documents were made available for public comment (44 FR 15926, March 15, 1979, 44 FR 43660, July 25, 1979, 44 FR 56628, October 1, 1979). These final criteria have been derived after consideration of all comments received.

These criteria documents are also issued in satisfaction of the Settlement Agreement in *Natural Resources Defense Council, et al.* v. *Train*, 8 E.R.C. 2120 (1976), modified, 12 E.R.C. 1833 (D.D.C. 1979). Pursuant to paragraph 11 of that agreement, EPA is required to publish criteria documents for the 65 pollutants which Congress, in the 1977 amendments to the Act, designated as toxic under section 307(a)(1). These documents contain recommended maximum permissible pollutant concentrations consistent with the protection of aquatic organisms, human health, and some recreational activities. Although paragraph 11 imposes certain obligations on the Agency, it does not create additional authority.

The Development of Water Quality Criteria

Section 304(a)(1) criteria contain two essential types of information: (1) discussions of available scientific data on the effects of pollutants on public health and welfare, aquatic life and recreation, and (2) quantitative concentrations or qualitative assessments of the pollutants in water which will generally ensure water

ENVIRONMENTAL PROTECTION AGENCY

[OW-FRL-2871-6]

Water Quality Criteria; Availability of Documents

AGENCY: Environmental Protection Agency.

ACTION: Notice of final ambient water quality criteria documents.

SUMMARY: EPA announces the availability and provides summaries of nine ambient water quality criteria documents and national guidelines for criteria development. These criteria are published pursuant to section 304(a)(1) of the Clean Water Act. These water quality criteria may form the basis for enforceable standards.

Availability of documents: This notice contains: (1) Summaries of nine documents containing final ambient water quality criteria for the protection of aquatic organisms and their uses, (2) a summary of changes in the document entitled "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" (which is an updated and revised version of the Guidelines previously published at 45 FR 79341; November 28, 1980), and (3) responses to public comments on the Guidelines. Copies of the complete criteria documents and the revised Guidelines may be obtained from the National Technical Information Service (NTIS), 5285 Port Royal Road, Springfield, VA 22161 (phone number ((703) 487-4650). A list of the NTIS publication order numbers for all 10 documents is published below. These documents are also available for public inspection and copying during normal business hours at: Public Information Reference Unit, U.S. Environmental Protection Agency, Room 2404 (rear), 401 M Street SW., Washington, D.C. 20460. As provided in 40 CFR Part 2, a reasonable fee may be charged for copying services. Copies of these documents are also available for review in the EPA Regional Office libraries. Copies of the documents are *not* available from the EPA office listed below. Requests sent to that office will be forwarded to NTIS or returned to the sender.

1. Ambient Water Quality Criteria for Ammonia—EPA 440/5-84-001; NTIS Number PB85-227114
2. Ambient Water Quality Criteria for Arsenic—EPA 440/5-84-033; NTIS Number PB85-227445
3. Ambient Water Quality Criteria for Cadmium—EPA 440/5-84-032; NTIS Number PB85-227031
4. Ambient Water Quality Criteria for Chlorine—EPA 440/5-84-030; NTIS Number PB85-227429
5. Ambient Water Quality Criteria for Chromium—EPA 440/5-84-029; NTIS Number PB85-227478
6. Ambient Water Quality Criteria for Copper—EPA 440/5-84-031; NTIS Number PB85-227023
7. Ambient Water Quality Criteria for Cyanide—EPA 440/5-84-028; NTIS Number PB85-227460
8. Ambient Water Quality Criteria for Lead—EPA 440/5-84-027; NTIS Number PB85-227437
9. Ambient Water Quality Criteria for Mercury—EPA 440/5-84-026; NTIS Number PB85-227452
10. Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses. NTIS Number PB85-227049.

FOR FURTHER INFORMATION CONTACT: Dr. Frank Gostomski, Criteria and Standards Division (WH-585), U.S. Environmental Protection Agency, 401 M Street SW., Washington, D.C. 20460, (202) 245-3030.

SUPPLEMENTARY INFORMATION:

Background

Section 304(a)(1) of the Clean Water Act (33 U.S.C. 1314(a)(1)) requires EPA to publish and periodically update ambient water quality criteria. These criteria are to reflect the latest scientific knowledge on the identifiable effects of pollutants on public health and welfare, aquatic life, and recreation.

EPA has periodically issued ambient water quality criteria, beginning in 1973 with publication of the "Blue Book" (Water Quality Criteria 1972). In 1976, the "Red Book" (Quality Criteria for Water) was published. On November 28, 1980 (45 FR 79318) and February 15, 1984 (49 FR 5831), EPA announced the publication of 65 individual ambient water quality criteria documents for pollutants listed as toxic under section 307(a)(1) of the Clean Water Act.

Today EPA is announcing the availability of nine individual water quality criteria documents which update and revise certain criteria previously published in the "Red Book" and in the 1980 ambient water quality criteria documents. The criteria documents for ammonia and chlorine replace criteria previously published in the 1976 "Red Book." The criteria documents for arsenic, cadmium, chromium, copper, cyanide, lead, and mercury replace the aquatic life criteria previously published in the 1980 ambient water quality criteria documents. Draft criteria documents were made available for public comment on February 7, 1984 (49 FR 4551). These final criteria have been derived after consideration of all comments received.

Dated: July 19, 1985.

Edwin C. Johnson,

Acting Assistant Administrator for Water.

Appendix A—Summary of Water Quality Criteria

1. Ammonia

Freshwater Aquatic Life

The procedures described in the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" indicate that, except possibly where a locally important species is very sensitive, freshwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of ammonia does not exceed the recommended criterion more than once every three years on the average and if the one-hour average concentration does not exceed the recommended criterion more than once every three years on the average.

The recommended exceedence frequency of three years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to ammonia exceeds the criterion. Stressed systems, for example one in which several outfalls occur in a limited area, would be expected to require more time for recovery. The resilience of ecosystems and their ability to recover differ greatly, however, and site-specific criteria may be established if adequate justification is provided.

To protect freshwater aquatic life, the criteria for ammonia (in mg/liter un-ionized NH_3) are based upon ambient water temperature and pH with one-hour and four-day average concentrations provided. Criterion concentratons for the pH range 6.5 to 9.0 and the temperature range 0 °C to 30 °C are provided in the following tables. Total ammonia concentrations equivalent to each un-ionized ammonia concentration are also provided in these tables. There is limited data on the effect of temperature on chronic toxicity. EPA will be conducting additional research on the effects of temperature on ammonia toxicity in order to fill perceived data gaps. Because of this uncertainty, additional site-specific information should be developed before these criteria are used in wasteload allocation modelling. For example, the chronic criteria tabulated for non-salmonids at temperatures much below

20 °C are less certain than those tabulated at temperatures near 20 °C. Where the treatment levels needed to meet these criteria below 20 °C may be substantial, use of site-specific criteria is strongly suggested. Development of such criteria should be based upon site-specific toxicity tests.

The use of criteria in designing waste treatment facilities requires the selection of an appropriate wasteload allocation model. Dynamic models are preferred for the application of these criteria. Limited data or other factors may make their use impractical, in which case one should rely on a steady-state model. The Agency recommends the interim use of 1Q5 or 1Q10 for criterion maximum concentration (CMC) design flow and 7Q5 or 7Q10 for the criterion continuous concentration (CCC) design flow in steady-state models for unstressed and stressed systems respectively. The Agency acknowledges that the CCC stream flow averaging period used for steady-state wasteload allocation modelling may be as long a 30 days in situations involving POTW's designed to remove ammonia where limited variability of effluent pollutant concentration and resultant concentrations in receiving waters can be demonstrated.

In cases where low variability can be demonstrated, longer averaging periods for the ammonia CCC (e.g., 30-day averaging periods) would be acceptable because the magnitude and duration of exceedences above the CCC would be sufficiently limited. These matters are discussed in more detail in the Technical Support Document for Water Quality Based Toxics Control (U.S. EPA, 1985).

(1) ONE-HOUR AVERAGE CONCENTRATIONS FOR AMMONIA*

pH	0 C	5 C	10 C	15 C	20 C	25 C	30 C
A. Salmonids or Other Sensitive Coldwater Species Present							
Un-ionized Ammonia (mg/liter NH_3)							
6.50	0.0091	0.0129	0.0182	0.026	0.036	0.036	0.036
6.75	0.0149	0.021	0.030	0.042	0.059	0.059	0.059
7.00	0.023	0.033	0.046	0.066	0.093	0.093	0.093
7.25	0.034	0.048	0.068	0.095	0.135	0.135	0.135
7.50	0.045	0.064	0.091	0.128	0.181	0.181	0.181
7.75	0.056	0.080	0.113	0.159	0.22	0.22	0.22
8.00	0.065	0.092	0.130	0.184	0.26	0.26	0.26
8.25	0.065	0.092	0.130	0.184	0.26	0.26	0.26
8.50	0.065	0.092	0.130	0.184	0.26	0.26	0.26
8.75	0.065	0.092	0.130	0.184	0.26	0.26	0.26
9.00	0.065	0.092	0.130	0.184	0.26	0.26	0.26
Total Ammonia (mg/liter NH_3)							
6.50	36	33	31	30	29	20	14.3
6.75	32	30	28	27	27	18.6	13.2
7.00	28	28	25	24	23	16.4	11.6
7.25	23	22	20	19.7	18.2	13.4	9.5
7.50	17.4	16.3	15.5	14.9	14.6	10.2	7.3
7.75	12.2	11.4	10.9	10.5	10.3	7.2	5.2
8.00	8.0	7.5	7.1	6.9	6.8	4.8	3.5
8.25	4.5	4.2	4.1	4.0	3.8	2.8	2.1
8.50	2.6	2.4	2.3	2.3	2.3	1.71	1.28
8.75	1.47	1.40	1.37	1.38	1.42	1.07	0.83
9.00	0.86	0.83	0.83	0.86	0.81	0.72	0.58
B. Salmonids and Other Sensitive Coldwater Species Absent							
Un-ionized Ammonia (mg/liter NH_3)							
6.50	0.0091	0.0129	0.0182	0.026	0.036	0.051	0.051
6.75	0.0149	0.021	0.030	0.042	0.059	0.084	0.084
7.00	0.023	0.033	0.046	0.066	0.093	0.131	0.131
7.25	0.034	0.048	0.068	0.095	0.135	0.190	0.190
7.50	0.045	0.064	0.091	0.128	0.181	0.26	0.26
7.75	0.056	0.080	0.113	0.159	0.22	0.32	0.32
8.00	0.065	0.092	0.130	0.184	0.26	0.37	0.37
8.25	0.065	0.092	0.130	0.184	0.26	0.37	0.37
8.50	0.065	0.092	0.130	0.184	0.26	0.37	0.37
8.75	0.065	0.092	0.130	0.184	0.26	0.37	0.37
9.00	0.065	0.092	0.130	0.184	0.26	0.37	0.37
Total Ammonia (mg/liter NH_3)							
6.50	36	33	31	30	29	29	20
6.75	32	30	28	27	27	26	18.6
7.00	28	28	25	24	23	23	16.4
7.25	23	22	20	19.7	19.2	19.0	13.5
7.50	17.4	16.3	15.5	14.9	14.6	14.5	10.3
7.75	12.2	11.4	10.9	10.5	10.3	10.2	7.3
8.00	8.0	7.5	7.1	6.9	6.8	6.6	4.9
8.25	4.5	4.2	4.1	4.0	3.9	4.0	2.9
8.50	2.6	2.4	2.3	2.3	2.3	2.4	1.81
8.75	1.47	1.40	1.37	1.38	1.42	1.52	1.18
9.00	0.86	0.83	0.83	0.86	0.91	1.01	0.82

*To convert these values to mg/liter N, multiply by 0.822.

Environment Canada
Environmental Protection Service

Environnement Canada
Service de la protection de l'environnement

Metal Mining Liquid Effluent Regulations and Guidelines

Règlements et directives sur les effluents liquides des mines de métaux

Regulations, Codes and Protocols
Report EPS 1-WP-77-1

Water Pollution Control Directorate
April 1977

Règlements, codes et méthodes d'analyse
Rapport EPS 1-WP-77-1

Direction générale de la pollution des eaux
Avril 1977

Canada

METAL MINING LIQUID EFFLUENT REGULATIONS AND GUIDELINES

RÈGLEMENTS ET DIRECTIVES SUR LES EFFLUENTS LIQUIDES DES MINES DE MÉTAUX

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